ADAPTING BIORETENTION FOR STORMWATER TREATMENT IN XERIC CLIMATES

by

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STATEMENT OF DISSERTATION APPROVAL

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ABSTRACT

Bioretention as a green approach to urban stormwater management has gathered a great deal of attention by researchers over the last 10 years, and implementation is becoming more widespread. However, much of the research and implementation of bioretention has occurred in mesic sites that receive over 100 cm of annual precipitation (such as Prince George's County, MD, and North Carolina) and a need has been identified to study these systems in other climates. To address this need, a new bioretention design is proposed for stormwater treatment facilities in water-limited climates based on research that describes biogeochemical processes such as vegetation evapotranspiration (ET) and nutrient cycling in water-limited ecosystems. The nutrient removal of this design was tested over 1 year and compared to the performance of the more commonly implemented wetland bioretention design and a media filter with no plants. Also, a ^{15}N isotopic label was added to each treatment to verify that plants directly participate in the removal of nitrogen from stormwater inputs.

The results of these studies demonstrate that a bioretention system designed to closely match arid ecosystem hydrology that includes the use of upland plants does remove more total nitrogen and phosphorus than the no-plant media filter, but the more commonly used wetland community removed the most total nitrogen and phosphorus of the tested designs. The added nitrogen label was identified in all vegetation in both the upland and wetland communities. Forty-six percent of the added ^{15}N label was recovered in the effluent of the control cell within 1 month of the addition of the label; 21% and 7%

of the added label was recovered in the same time period from the upland and wetland treatments, respectively. In conclusion, the proposed design does protect receiving waters from nutrient loading associated with stormwater runoff from urban landforms, but increasing planting densities by two or three times and expanding the palate of vegetation used may improve this performance.

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I would like to thank my wife for motivating me to return to school and for all of her support throughout the journey. I would also like to thank my parents and my in-laws for their help in taking care of us while we tried to take care of the twins over the last year. This research and dissertation could not have been completed without their support.

All of my advisors have played a key role in the development of this project. Dr. Burian was instrumental in the research development phase; Dr. Goel facilitated access to analytical equipment and lab; Dr. Bowling provided critical (and highly constructive) input in the design of my labeling experiment and in the final analysis of my data. Last and most important, Dr. Pomeroy and Dr. Hultine have been supportive, enabling, and encouraging throughout the entire 4 years, and I could not imagine completing this journey without either of them. While Dr. Ehleringer is not formally on my committee, he has played an instrumental role in facilitating the interdisciplinary collaborations that my project required to be successful through access to the growth site, access to SIRFER, and by encouraging the interdisciplinary nature of water resources research.

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Here are a few of the general lessons I have learned. The research I conducted during my PhD straddled the fields of ecosystem ecology and civil engineering to address

the design challenge of ecological stormwater mitigation. As a result, I have divided my expertise between two traditionally isolated fields. The most challenging aspect of my interdisciplinary approach was that I was required to communicate complex concepts between engineers and ecologists. In order to do this successfully, I had to form a broad knowledge base that spanned both disciplines, and I have had to synthesize this base into new concepts that could be implemented in the context of stormwater management. While I have become well versed in the fundamental concepts of both disciplines, I have not gained a deep understanding of any specific modeling approach, laboratory procedure, or biological process. Because of the recent emphasis in interdisciplinary approaches to addressing research questions within the scientific community, I believe that the extensive knowledge and peer base I have developed will suit me well as I advance in my academic career.

I realize I have taken a path less traveled. I hope that the relationships I have facilitated between civil engineering and ecosystem sciences help to pave the way so that future students can follow my lead and continue to apply ecological principles to improve the sustainability of civil engineering. But more specifically, here are a few points I hope others can learn from:

- Using 3 g of 99.8 atom% ^{15}N NH_4NO_3 is probably much more label than necessary for my research.
- You can attract a lot of volunteers if professors offer extra credit for participation.
- Science has to look good: Aesthetics of a research site are especially important to establish credibility, especially if stakeholders will interface with the site.

- Poster presentations are academia’s stick to motivate grad students to do better science.
- You can’t write a good journal article in 1 month.
- Have other people read your writing. Then have them read it again.
- Breaking off pieces of your research for others to develop is very difficult but can be very rewarding to see others succeed.
- There is no substitute for face time with advisors, collaborators, facilities managers, and administrators – also, maintaining professional relationships is very hard without a personal foundation.
- Caring for infant twins is hard; my in-laws are incredible people.

CHAPTER 1

INTRODUCTION

1.1 Background

1.1.1 Urbanization as a threat to surface waters

Urbanization is cited by the U.S. Environmental Protection Agency (U.S. EPA) as the greatest cause of impairment to surface water quality (U.S. EPA, 2010b). In the 20th century, urbanization increased the world's metropolitan population from 220 million to 2.8 billion (United Nations Population Fund, 2009) contributing to significant changes in land surface characteristics. The extent of urbanization has been suggested as the most dominant factor altering the water budget of an area (Claessens, Hopkinson, Rastetter, & Vallino, 2006). Urbanization and its associated stormwater infrastructure have a wide range of impacts on regional hydrology through removal of indigenous vegetation, grading of land surfaces, compaction of pervious surfaces, and construction of impervious surfaces (Burian & Pomeroy, 2010; Novotný & Brown, 2007).

The populations of Arizona, Nevada and Utah are expected to double in the next 20 years; California, Colorado, and Idaho are all expected to grow over 50% in the same period (U.S. Census Bureau, 2005). Stormwater runoff from this projected urban growth in desert regions will have profound impacts on the already delicate water cycle by increasing urban runoff volume and event frequency, resulting in increased channel erosion and sedimentation (Hollis, 1975; Konrad & Booth, 2005; Paul & Meyer, 2001; Walsh, Fletcher, & Ladson, 2009). Once degraded by unmanaged stormwater, opportunities to rehabilitate riparian ecosystems in xeric climates are often limited by water availability and other resources, thereby increasing the time and expense required to restore damaged ecosystems back to pre-disturbance conditions (Gasith & Resh, 1999; Schwinning, Belnap, Bowling, & Ehleringer, 2008; S. G. Whisenant, 1999).

A recent National Research Council (NRC) report (2008) emphasizes that stormwater runoff from the built environment is one of the great challenges of modern water pollution control. Stormwater runoff is a principal contributor to water quality impairment of waterbodies throughout the United States. Nutrients, specifically, are documented as a cause for impairment for approximately 15% of the 44,000 impaired water bodies in the United States (U.S. EPA, 2010a). Nutrient levels in stormwater can periodically be high and produce large mass discharges of nitrogen (N) (Burian, Streit, McPherson, Brown, & Turin, 2001; Burton & Pitt, 2010; U.S. EPA, 2010a, 2012). N sources include aerosols dissolved in precipitation, dry atmospheric deposition, soils and fertilizers washed off of impervious surfaces, and animal waste (Burton & Pitt, 2010). Poor air quality from particulate dust and the burning of fossil fuels in heavily populated basins common in the desert Southwest are increasing dissolved N levels in precipitation (Burian et al., 2001; Galloway et al., 2003; Pataki et al., 2006; Taylor, Fletcher, Wong, Breen, & Duncan, 2005). As populations in the desert Southwest grow and natural lands are replaced by suburbs, runoff volume and event frequency are increasing, increasing the amount of nutrients transported from terrestrial to aquatic systems (Buchanan & Honey, 1994; Hollis, 1975). Urbanization increases the amount of N in stormwater, paves over the natural ecosystems that previously utilized the N deposited on the landscape, and conveys the N-enriched stormwater directly to receiving waters.

Most freshwater ecosystems are N limited (Schade, Marti, Welter, Fisher, & Grimm, 2002). Adding anthropogenic N to these systems can lead to increased algal growth that directly causes eutrophication and/or anoxic and hypoxic dead zones that choke out aquatic macrofauna (Field et al., 1998; Lee, 2002; Novotny & Olem, 2003; Novotny & Witte, 1997; U.S. EPA, 2010a). Some of the detrimental effects of eutrophication include lower dissolved oxygen, choked flows, increased water temperature, and increased sedimentation (Diaz & Rosenberg, 2008; Galloway et al., 2003; Howarth, 1988; Novotny & Witte, 1997; Rabalais, Turner, & Wiseman Jr, 2002). Additionally, increasing nutrient availability in waterways that were historically nutrient limited can create opportunities for invasive species such as *Tamarix ssp* (tamarisk) and *Elaeagnus angustifolia* (Russian olive) that have the capacity to increase transpiration losses out of waterways to the atmosphere. (Ehleringer, 2010; Hultine et al., 2009; Hultine, Bush, & ; Hultine, Jackson, Burtch, Schaeffer, & Ehleringer, 2008; Konrad & Booth, 2005; Pataki, Bush, Gardner, Solomon, & Ehleringer, 2005; Rickey & Anderson, 2004; Stromberg, Tiller, & Richter, 1996). Each of these unintended consequences of untreated stormwater runoff further compounds water resources availability in systems where demand often exceeds local supply. Investment in treatment technologies capable of reducing the harmful effects of urban stormwater runoff is needed to maintain the hydrological integrity of water resources and facilitate future growth projection in the desert Southwest.

1.1.2. Bioretention used to mitigate urban stormwater

Low Impact Development (LID) stormwater management includes facilities that replicate natural ecosystem functions to protect infrastructure and natural ecosystems from the negative hydrological effects of urbanization. Green infrastructure (GI), a part of LID, is comprised of the interconnected networks of natural and constructed ecological systems within, around, and between urban areas (Tzoulas et al., 2007). A major focus of LID and GI initiatives is using plants within small engineered ecosystems to reduce pollutant loading to receiving waters and to infiltrate and transpire captured water.

Bioretention facilities are among the most common GI stormwater approaches currently implemented. Bioretention utilizes soils and both woody and herbaceous plants to remove pollutants from stormwater runoff (U.S. EPA, 2013). Bioretention has been demonstrated to reduce peak flows of stormwater runoff and nutrient loading to receiving waters (BMPDatabase.org, 2012; Bratieres, Fletcher, Deletic, & Zinger, 2008; Chen, Peltier, Sturm, & Young, 2013; Collins et al., 2010; Davis, 2007; Dietz & Clausen, 2005; Hatt, Fletcher, & Deletic, 2009; Hunt, Jarrett, Smith, & Sharkey, 2006; Hunt, Smith, Jadlocki, Hathaway, & Eubanks, 2008; Kim, Seagren, & Davis, 2003; Li & Davis, 2009; Prince George's County, 2002). Currently, a number of bioretention design guidelines are available as references for planners and designers (Prince George's County, 2002; U.S. EPA, 2013). These guidelines focus on bioretention design for mesic systems, or systems that receive 750 to 2000 mm of rain each year, and address traditional stormwater engineering approaches such as facility sizing and hydraulics design.

In their stormwater report, the National Research Council (NRC) also highlighted that while performance characteristics have been established for many stormwater management measures, additional research is needed on the relevant hydrologic and water quality processes across different climates and soil conditions (2008). The aforementioned research and design recommendations do not provide guidelines sufficient to assist designers of LID and GI in xeric climates, where stormwater management challenges associated with urbanization are expected to increase the most in the coming decades.

1.2 Problem statement

Despite the effort committed to studying treatment capacity in bioretention cells and the parallel effort committed to quantifying ecological processes related to nutrient and water cycles in urban systems, the two disciplines have progressed separately. To the engineer, the bioretention cell is a black box that improves water quality with little understanding as to how the system functions (Davis, Hunt, Traver, & Clar, 2009), and to the urban ecologist, the urban landscape is a subject of study, with few venues to apply findings that will improve ecological processes in urban areas. The techniques developed by biologists to quantify ecosystem N exchanges and water utilization in recent years could greatly enhance the engineer's ability to design and implement LID GI stormwater infrastructure that aid in the protection of a community from stormwater (Davis et al., 2009).

Previous research on bioretention engineering demonstrates a continuing focus of the physical processes of LID with ecological processes considered secondarily. However, recent work in the field of ecology suggests that ecological processes likely control all aspects of the unit processes driving treatment within bioretention (Barbour &

Billings, 2000; Lucas & Greenway, 2008; Paungfoo-Lonhienne et al., 2008; Treseder, 2004; Whiteside, Treseder, & Atsatt, 2009). Understanding the mechanisms of how volume and nutrients are treated within bioretention is critical to maximizing the sustainability of water resources management in the desert Southwest and in other water-limited regions.

The EPA has compiled design guidelines for bioretention in temperate, mesic climates, and simply states that "design modifications may be necessary to implement LID in arid or cold climates" (U.S. EPA, 2013). Given the predictions of rapid population growth in the desert Southwest, learning to mitigate negative impacts of urban growth is critical to protecting our unique ecosystems. Opportunity for ecological remediation is more limited in arid systems because many plants perform poorly or die if water is limited. Because of this, bioretention design, including hydrologic and ecological, and aesthetic components of a facility, requires greater consideration than bioretention for more mesic climates. There is an immediate need to better understand how bioretention functions hydraulically and ecologically under xeric precipitation patterns and climate extremes common to the desert Southwest.

1.3 Research objectives

In order to successfully implement LID GI stormwater technologies, stormwater engineers must gain a better understanding of the ecosystem processes that drive treatment in LID GI. The goal of this research is to address the current gap between ecosystem sciences and stormwater engineers by using research tools and knowledge developed by ecosystems scientists to design LID GI. Under the broad scope of applying ecological concepts to improving stormwater LID GI design, three specific hypotheses guided this research:

- Hypothesis 1: plants drive nitrogen removal in bioretention, even under the high nutrient loading conditions expected from urban landscapes to bioretention facilities.
- Hypothesis 2: bioretention facilities without vegetation (media filters) may physically filter larger constituents from stormwater, but dissolvable nutrients such as NO_2^-, NO_3^-, and $NH4^+$ will wash through the facility untreated.
- Hypothesis 3: In bioretention cells with plants, influent stormwater nitrogen will be immobilize biologically or abiotically during a storm event, biologically mineralized to NO_3^-, then taken up by plants. These processes will reduce nutrient loading to receiving waters even when plants are dormant, because roots and the associated microbial networks will store the influent N until conditions are favorable for above-ground growth.

To test these hypotheses, three research efforts were carried out. First, a review of stormwater engineering, rangeland restoration ecology, and plant physiology literature was synthesized into a design recommendation for bioretention in xeric climates. Second, bioretention test cells were constructed following these design recommendations, and the nutrient removal capacity of this design was tested over 1 year. Each cell that was tested comprised of different plant communities, including an arid upland community, a wetland community, and a media-only cell with no plants. Third, a ^{15}N isotopic label was added to the test cells as part of the 1-year nutrient removal study to test if plants assimilate stormwater N into biomass.

1.4 Dissertation overview

The details of the methods of research and results obtained by these methods are presented in the following chapters. First, in Chapter 2, a design recommendation for the implementation of bioretention in xeric climates is synthesized from a review of stormwater engineering, rangeland restoration ecology, and plant physiology literature. This chapter was published in the Journal of American Water Resources in December of 2012 (Houdeshel, Pomeroy, & Hultine, 2012). Next, Chapter 3 describes the nutrient treatment capacity of this design over 1 year and compares it to the treatment achieved by a media filter with no vegetation and a wetland vegetation community commonly recommended for use in bioretention in mesic climates. Chapter 4 describes the methods and results from a ^{15}N labeling experiment in which a NO_3NH_4 source enriched with ^{15}N was added to each test garden described in Section 1.3, then recovered in plant tissue and in the garden effluent. Chapters 3 and 4 are in production for journal submission, but have not been submitted to date. Chapter 5 summarizes the conclusions of each previous chapter, describes future research topics, and recommends improvements to the proposed design for bioretention in xeric climates in Chapter 2.

CHAPTER 2

BIORETENTION DESIGN FOR XERIC CLIMATES BASED ON ECOLOGICAL PRINCIPLES

2.1 Abstract

Bioretention as a green approach to urban stormwater management has gathered a great deal of attention by researchers over the last 10 years, and implementation is becoming more widespread. However, much of the research and implementation of bioretention has occurred in mesic sites that receive over 100 cm of annual precipitation (such as Prince George’s County, MD, and North Carolina) and a need has been identified to study these systems in other climates. The arid southwestern United States is the fastest growing region in the country, and there is a need to establish design recommendations for bioretention to control stormwater from expanding development in this ecologically harsh region. To initiate the discussion about the use of bioretention in arid and semi-arid climate, a review of the ecological constraints on plants in arid and semi-arid systems is combined with design recommendations of bioretention in more mesic climates and USEPA Storm Water Management Model simulations to synthesize ecologically based design recommendations (including regional plant selection recommendations) for bioretention in arid climates.

2.2 Introduction

Acceptance and implementation of low impact development (LID) approaches to stormwater management has increased dramatically in recent years (Collins et al., 2010; Davis et al., 2009). Green infrastructure (GI), a part of LID, is comprised of the interconnected networks of natural and constructed ecological systems within, around, and between urban areas (Tzoulas et al., 2007). The expansion of these practices can be attributed to the many organizations that invested early in the LID and GI movement in both implementation and investigation of the costs and benefits of these approaches (Davis, 2007; Davis et al., 2009; Hunt et al., 2006; Prince George's County, 2002; U.S. EPA, 2012) These efforts have focused largely on LID and GI approaches to stormwater management in mesic climates, or climates that receive 750 mm to 2000 mm (30 in to 80 in) precipitation per year, to incorporate wetland remediation approaches to stormwater treatment. However, little work has taken place to modify these concepts to xeric (arid and semi-arid) climates (Davis et al., 2009). Implementation of GI in the xeric western United States is of utmost importance because this region is experiencing the greatest urban growth in the United States and ecological resilience is low given limited precipitation inputs and high evaporative demand (Belnap, 1995; Claessens et al., 2006; Schwinning et al., 2008; U.S. Census Bureau, 2005; U.S. EPA, 2010b; S. Whisenant, 1999).

Bioretention is a GI practice that utilizes engineered ecosystems to store, treat, and infiltrate precipitation that falls on developed impervious surfaces. Bioretention maximizes water storage in a specifically designed garden, or series of gardens, so water can be infiltrated into the ground or transpired by plants as it would have prior to development of impervious areas (Davis et al., 2009; Hsieh & Davis, 2005; 2000). Water is captured, treated, and in most cases, infiltrated near the area of precipitation. The known stormwater management benefits of bioretention include reducing runoff rates and volumes from urban areas known to accelerate erosion in receiving waters; reducing pollution transported from the urban landscape into fragile aquatic habitat; reducing the need for expensive stormwater conveyance systems; and flood control during large precipitation events (Brix, 1993; Davis et al., 2009; Novotný & Brown, 2007; Roesner, Bledsoe, & Brashear, 2001; Zhang, Seagren, Davis, & Karns, 2011). Bioretention creates an opportunity to expand green space in urban settings, which makes a city more attractive and may act as a local carbon sink (Pataki et al., 2006). Additionally, if bioretention is installed as an alternative to traditional landscaping, implementation of bioretention and other GI stormwater management approaches may relieve emerging stress on regional water supply in xeric locations by creating an attractive no-irrigation landscaping alternative.

Currently there is a conspicuous lack of information available in the stormwater management literature addressing bioretention in xeric climates (National Research Council, 2008). Therefore, in order to initiate dialogue addressing guidelines for bioretention in xeric regions, this paper combines designs of bioretention in mesic climates with literature describing plant ecophysiology, arid land ecology, wildland restoration for arid regions, and hydrologic modeling. Design guidelines are then synthesized to create an ecologically based stormwater management system capable of thriving in harsh western climates while simultaneously treating urban runoff and utilizing stormwater as the primary irrigation source.

2.3 The need for bioretention facilities in xeric urban landscapes

The extent of urbanization has been suggested as the most dominant factor altering the water budget at local and regional scales (Claessens et al., 2006). Locally, urbanization is stated by the U.S. EPA to be the greatest cause of impairment to surface water quality (U.S. EPA, 2010a). Regionally, urban spread in xeric climates increases water demand by increasing population and increasing the area of land irrigated for lawns and gardens (Eriksson, Auffarth, Henze, & Ledin, 2002).

To continue addressing the challenges of managing stormwater runoff associated with urbanization, the USEPA has recently initiated national rulemaking to reduce stormwater discharges from new development and redevelopment and make other regulatory improvements to strengthen its stormwater program (U.S. EPA, 2011) . As part of this rulemaking, GI is being emphasized for its ability to reduce the magnitude of water cycle modification in urban areas, as well as its ability to reduce pollutants in stormwater runoff. The National Resource Council has also emphasized that stormwater control measures that utilize engineered ecosystems, including LID, to harvest, infiltrate, and evapotranspire stormwater, are critical to reducing the volume and pollutant loading of small storms (National Research Council, 2008).

Bioretention utilizes soils and plants to remove pollutants from stormwater runoff (Davis et al., 2009). Currently, a number of bioretention design guidelines are available as references for planners and designers (Prince George's County, 2002; U.S. EPA, 2012). These guidelines focus on bioretention design in mesic systems, and address traditional stormwater engineering approaches such as facility sizing and hydraulics design. The NRC also highlighted the need to expand study of relevant hydrologic and water quality processes across different climates and soil conditions (National Research Council, 2008). The aforementioned LID and GI guides do not provide appropriate information to assist designers in solving the Stormwater management challenges that accompany the forecasted urbanization in xeric climates.

The challenge of mitigating the negative effects of urbanization on surface waters through GI is further compounded in xeric climates because the native ecosystems are less resilient to recover from anthropogenic influence (Belnap, 1995; Schwinning et al., 2008) . Additionally, ecological remediation opportunities are limited by water availability in xeric regions (Barbour & Billings, 2000; Heady & Child, 1994; Knapp, Briggs, Hartnett, & Collins, 1998; S. Whisenant, 1999). States dominated by xeric climates have the fastest growing populations in the country (U.S. Census Bureau, 2005); this growth will have profound impacts on the water cycle at the local and regional scale.

To compound the difficulties of managing water for large, growing populations in sensitive xeric climates, Global Climate Models (GCMs) have predicted that the Colorado River will suffer a reduction in stream flow due to climate forcing caused by anthropogenic inputs of CO_2 and other greenhouse gasses to the atmosphere (T. Barnett et al., 2004; T. P. Barnett & Pierce, 2009; N. S. Christensen & Lettenmaier, 2007; U.S. Buereau of Reclamation, 2007). The Colorado River provides water to 27 million people and drives significant sectors of the economies of Colorado, Utah, Arizona, Nevada, southern California, and Mexico (T. P. Barnett & Pierce, 2009). Consumptive diversions have prevented almost all flow from reaching the Sea of Cortez since the beginning of the construction of the Hoover Dam in the 1930s (Carriquiry & Sánchez, 1999). These regional predictions of rapid growth and reduced water availability emphasize the need to manage stormwater to protect ecosystems stressed from a changing climate and to reduce regional water demand by managing stormwater as a resource.

Over half of the water use within major urban centers in northern Utah and southern California is irrigation of landscaping (Eriksson et al., 2002; Salt Lake City, 2013). In xeric climates, implementing GI in place of traditional landscaping provides an opportunity to reduce water demand per capita, thus allowing more growth under constrained resource availability. An integrated approach to stormwater management and water supply in xeric climates allows protection of the region's scarce surface waters by reducing the threat of physical and chemical damage to waterways from urban runoff and the demand these waters must satiate.

2.4 Proper plant selection for LID facilities in xeric climates

In xeric climates, water is a limited resource. The vast majority of the Desert Southwest, including southern California, Nevada, Utah, Arizona, New Mexico and western Texas receive less than 380 mm of rain fall each year. This creates a unique challenge as few species suggested for use in bioretention in more mesic climates can survive the dry conditions common in the arid West. Looking to the extensive work that has been done in ecological restoration can close the gap on this challenge by providing a framework for plant selection. Ecological restoration is a discipline that focuses on how to reestablish plants in order to repair damaged hydrological functions in wildlands (Heady & Child, 1994; Holechek, Pieper, & Herbel, 1995; Knapp et al., 1998; S. Whisenant, 1999) Recommendations for plant selection for LID in xeric climates are limited, but recommendations for plant selections for ecological restoration in xeric climates are abundant. Plant selection for Ecological Restoration is based on matching physiological plant traits with site conditions to repair damaged hydrologic functions in wildlands (Heady & Child, 1994; Holechek et al., 1995; Knapp et al., 1998; S. Whisenant, 1999). The common focus on renovating the hydrologic cycle shared between the use of bioretention for urban stormwater management and ecological restoration enables designers of bioretention to draw from the extensive work done by restoration ecologists to restore predevelopment hydrology to urban landscapes.

Within xeric climates of the western United States, two distinct precipitation patterns drive equally distinct vegetative communities (Figure 2.1). The Great Basin and Intermountain West, encompassing the Salt Lake City, UT, Boise, ID, and Denver, CO urban centers are “cool” deserts, where precipitation typically falls in the winter or spring as snow (Barbour & Billings, 2000). Plants depend on soils to store moisture until the growing season when temperatures are warm enough to allow plant activity. Growth is rapid in spring and quickly decreases for many shallowly-rooted species as soils dry. Coastal Southern California experiences the same pattern of precipitation delivery except that average winter temperatures are well above freezing. Conversely, Arizona, western Texas and New Mexico and southern Utah are in regions that are considered “warm” deserts, or deserts that receive the majority of their precipitation as rain in phase with the growing season (Barbour & Billings, 2000). The Mojave Desert in Eastern California and southern Nevada is transitional, as this region is affected by both of the above patterns. Las Vegas is the largest urban area in this climate regime.

Plants in xeric climates have adapted to surviving in low-water environments by evolving mechanisms that control water transport within the plant. Plant water transport is driven by water potential (Ψ) gradients through the soil-plant-atmosphere continuum. Water flow through the plant is initiated as the stomata (small pores on the leaf surface that allow gas exchange) open, allowing water to evaporate from the moist leaf to the drier atmosphere, thereby reducing leaf Ψ below soil Ψ (Elfving, Kaufmann, & Hall, 1972; Zimmermann, 1983). Low atmospheric Ψ evaporates water from the stomata lens, which pulls water from the soil into the roots, through the xylem to the leaf surface. If this pathway is broken by cavitation, most plants lose the ability take up water from the soil even if soil moisture becomes adequate to allow transport, and the plant dies. Plants can control leaf surface area characteristics and the rate of water flow by controlling the aperture of the stomata, allowing more or less water to evaporate from the leaf to maintain a suitable Ψ gradient (Sperry, Hacke, Oren, & Comstock, 2002; Tyree & Sperry, 1989).

In xeric climates, plants are stressed by both the limited amount of moisture in soils and by the limited amount of moisture in the atmosphere. Low soil moisture combined with low atmospheric water content increases a plant's risk of cavitation of liquid water in the conduits of plants (i.e., the entry of gas bubbles into conduits that normally transport water to the leaves) as a result of extreme Ψ gradients between the leaf-atmosphere interface and the root-soil interface. Freezing of the water in the xylem is another stress plants in xeric climates are exposed to that can lead to cavitation. Ice crystals can act to catalyze the formation of bubbles in the low pressures in the xylem, or the formation of ice within a vessel can concentrate dissolved gas beyond the saturation point, creating bubbles that cause cavitation.

Distinct native plant communities have evolved to maximize productivity in each desert region. These communities are physiologically adapted to minimize the greatest risk of xylem cavitation and subsequent leaf desiccation and plant mortality in their respective region. In addition to the different patterns of precipitation delivery and related ψ stresses described above, sensitivity to cold has also influenced plant distribution across xeric landscapes. Cold desert species are exposed to frost damage because of colder temperatures when soil moisture is available in winter and spring. One common strategy to limit frost damage in winter is to shed leaves, but this limits photosynthetic capacity in early spring when temperatures are commonly warm enough for photosynthesis but freezing is still common. Other strategies employed by evergreen shrubs such as *Artemisia tridentata* (sagebrush) and *Cercocarpus ledifolius* (curl-leaf mountain mahogany) include concentrations of phenols and salts in the leaf to lower the freezing temperature of water; insulating the xylem with thick bark to reduce the risk of ice formation, decreasing xylem diameter and other architectural techniques that compromise hydraulic conductivity for resiliency if freezing damage does occur, and re-filling the xylem through positive root pressure in the spring (Sperry et al., 2002; Tyree & Sperry, 1989). The development of these strategies allows plants in cold deserts to photosynthesize early in the growing season when water is available then close down water transport through the dry months, thus avoiding the risk of cavitation due to extreme Ψ gradients in summer. Mojave Desert species can “green up” or produce new leaves with adequate temperature and soil moisture any time of year, and drop their leaves when the soil dries. In all cases, the plant avoids cavitation by developing mechanisms that allow photosynthesis when water is available and protect against the potentially harmful temperatures that accompany water availability.

Bunchgrasses are common in arid and semiarid ecosystems, and are known to have high transpiration rates over short growing seasons. The growing season of a grass is described as warm season or cool season, indicating geographic range and physiology. Cool season grasses are most productive in spring when temperatures are moderate and soil moisture is abundant because enzymatic and biochemical limits reduce photosynthetic efficiency per water use in hot weather. Warm season grasses utilize a more evolved photosynthetic pathway and are most productive when high temperatures are coupled with summer precipitation. When growth conditions deteriorate, bunchgrasses drop their seeds and go dormant until either temperature or soil moisture is again optimal for growth.

Root structure is also varied among western desert plant species in different temperature regimes. Two rooting patterns are most common: phreatophytes, or plants with large, deep tap roots to access groundwater sources year round, or plants with shallow and extensive, wide spreading root networks (Barbour & Billings, 2000; Holechek et al., 1995; S. Whisenant, 1999). Rooting depths of phreatophytes, including *Chrysothamnus nauseous* (rubber rabbitbrush), *Atriplex confertifolia* (shadscale), *Quercus gambelii* (scrub oak), and *Prosopis glandulosa* (mesquite), are known to exceed 30 meters, including a recorded depth in excess of 50 meters (Canadell et al., 1996); roots of cool season shrubs are commonly four to nine times the above-ground biomass

(Fernandez & Caldwell, 1975; Jackson et al., 1996; Rodin & Bazilevich, 1967). Phreatophytes thrive by utilizing deep soil water other plants cannot access in climates where prolonged droughts and seasonal dry periods in summer or shallow salt accumulations are common. Desert shrubs that cast extensive root networks through shallower soils, such as *L. tridentata* (creosote bush), *Pinus edulis* (piñon pine) and *Arctostaphylos sp* (Manzanita), quickly capture and utilize small precipitation events during the growing season (Brisson & Reynolds, 1994; Kummerow, Krause, & Jow, 1977; West, Hultine, Jackson, & Ehleringer, 2007). Some shrubs including *A. tridentata* (sagebrush), *Prosopis velutina* (velvet mesquite), and *Juniperous occidentalis* exhibit deep taproots and shallow, extensive root networks (Hultine, Scott, Cable, Goodrich, & Williams, 2004; Miller, Eddleman, & Kramer, 1990; Miller, 2005). Shrubs with deep tap roots and shallow root networks such as *A. tridentata* and *P.velutina* have been shown to lift water from deep, saturated soils to shallow, dry soils at night when the plant does not need the water to drive photosynthesis or gas exchange, making water available to transpire the following day by either itself or neighboring plants such as bunchgrasses (Hultine et al., 2004; Richards & Caldwell, 1987).

Directly related to root growth is the presence of Arbuscular Mycorrhizal Fungi (AMF) in soils. As root length increases, opportunities for interactions with AMF increase (Treseder & Turner, 2007), increasing the ability to absorb nutrients and help sustain plants in xeric systems (Gianinazzi-Pearson, 1996; Requena, Perez-Solis, Azcón-Aguilar, Jeffries, & Barea, 2001; Tao & Zhiwei, 2005). Soil structure and infiltration rates are also improved by AMF. The fungi exude glomalin, a sticky protein that improves soil stability by holding soil aggregates together during wetting and drying cycles (Wright, 1996).

Green infrastructure and wildland restoration both strive to restore natural hydrology to a compromised site. However, GI has a great advantage in achieving this goal because the facility is engineered and many variables can be controlled. Soil texture, moisture regimes, and land use are often all out of the control of the restoration ecologist. But the engineer can dictate plant selection by choice of growth media, can adjust moisture amounts by adjusting drainage size to supply water to the site, and can erect permanent barriers such as curbing to protect the site from trampling. A basic understanding of plant water relations and physiological plant traits can greatly increase the opportunity for a successful GI installation.

2.5 Design recommendations

Currently, a number of bioretention design guidelines are available as references for planners and designers (Prince George's County, 2002; U.S. EPA, 2012). Reviewing these references demonstrates that to date, designing LID bioretention has concentrated on mesic systems and addresses traditional stormwater engineering approaches such as facility sizing and hydraulics design. In addition to these parameters, selecting appropriate plants for a bioretention can enhance facility performance by promoting natural ecological processes and play an equally critical role in the success or failure of the facility. Plant selection suggestions for mesic landscapes are provided in currently available reference materials (Prince George's County, 2002; U.S. EPA, 2012), but are insufficient to assist designers who are not familiar with the ecological intricacies of xeric climates.

Based on physiological traits and differences in water use, a mixture of bunchgrasses and shrubs will maximize the functional treatment benefits offered by plants within a bioretention facility. Grasses can demonstrate extensive root growth up to 0.6 m and can regrow up to six sets of roots to this depth per growing season (Knapp et al., 1998). The regrowth of roots creates many small channels for water rushing onto the surface of a bioretention facility to rapidly infiltrate through the topsoil layer to the storage layer, which minimizes ponding time and maximizes capture efficiency. Grass roots also form an extensive net that interfaces with AMF, forming a dense web stabilizing soils and filtering water as it flows through. Deep-rooted shrub roots can access deep-water pockets that are unavailable to grasses (Canadell et al., 1996; Knapp et al., 1998; Richards & Caldwell, 1987; Wilcox, Breshears, & Turin, 2003). Shrubs have fine roots that turn over to improve infiltration and perennial roots that do not turn over that can grow to great depths. Select shrubs can root through the bioretention cell, encouraging infiltration into the native soils below. Canadell et al. (1996) report that many shrubs in xeric climates grow roots exceeding 5 m, and can grow through many types of media. The process of hydraulic lift facilitated by a wide range of deeply-rooted plant species may serve to irrigate bunchgrasses with shallower root systems that cannot directly access deep, seasonally stored water. Mimicking nature by combining deep-rooting shrubs with extensive, shallow rooting grasses should provide the maximum hydraulic stormwater function and drought tolerance potential for a bioretention garden.

Plant roots play critical roles in the ability of bioretention to infiltrate, transpire water, and absorb nutrients in all soil textures. Increased root growth increases infiltration by creating macropores as roots grow and turn over (Knapp et al., 1998; Whisenant, 1999). Sandier soils encourage more expansive root growth because as average soil particle size increases the soil's physical ability to store water and nutrients decreases, forcing the plants to mine deeper into the soil to find water and nutrients (Cuevas & Medina, 1986; Nagarajah, 1987; Silver et al., 2000). Infiltration rates through coarser media are high without plants, and while less root growth may occur in fine soils, plants have a greater effect on infiltration rates in fine soils. Undeveloped, fine soils with low organic content have low inherent infiltration rates (Wilcox, Sbaa, Blackburn, & Milligan, 1992). Plant roots also provide hosts for AMF, further improving soil structure. Fine-textured soils that are well-developed through the addition of organic matter and established macropores can exhibit infiltration rates much greater than bulk mineral soil of the same texture.

Selecting appropriate plants for use in bioretneiton in xeric climates must be addressed regionally because stresses to plants are unique to each desert region. A summary of plant species, ecological traits, and appropriate regions of use in bioretention are given in Table 2.1 and shown in Figure 2.2. In general, warm season bunchgrasses should be planted in concert with locally native shrubs and evergreens in warm deserts, and a mixture of warm season and cool season bunchgrasses should be planted with locally native shrubs and evergreens in cool deserts. Commercial plant availability can dictate plant selection; however, proper planning can allow suppliers to order desired species from growers if the demand for a particular species is high.

Spring or summer plantings in all regions will require intensive weekly irrigation during the first year of establishment to help root systems develop sufficiently to access deeper pockets of soil moisture. After the first year, the plants suggested in Table 2.1 will survive and grow in each plant's recommended region without supplemental irrigation. The plants suggested are slow-growing species under natural moisture regimes. Because of the low growth rates, leaf turn over and maintenance are expected to be low. Trimming bunchgrasses to a height of 10 cm each winter will promote new shoot growth the next growing season.

2.6 Physical design parameters for bioretention in xeric climates

Before GI can be used to replicate and restore natural hydrology to a site, the natural hydrology of that site must be well understood (Whisenant, 1999). In warm deserts, precipitation falls onto highly permeable sands and is rapidly returned to the atmosphere by evapotranspiration or infiltrated into groundwater storage and produces very little overland flow for small events, but high-intensity events that saturate shallow soils frequently cause flash flooding (Barbour & Billings, 2000; Herschman, 2008). In cold deserts, very little precipitation falls during the growing season, but because of snow storage, infiltration, and localized groundwater storage, moisture is often available to plants throughout the hot summer (Barbour & Billings, 2000; Ehleringer, Phillips, Schuster, & Sandquist, 1991; Knapp et al., 1998; S. Whisenant, 1999).

As snow melts during warm periods in the winter and spring, a great deal of water is infiltrated into the soils. Spring rains fall on the wetted soil and rapidly infiltrate to feed local aquifers and almost no runoff is produced (Wilcox, Rawls, Brakensiek, & Wight, 1990; Wilcox et al., 1992). Water slowly percolates to feed base flow in nearby streams or is locally stored in pockets of deep soil water and available to the native, deep-rooted plants through much of the summer (Donovan & Ehleringer, 1994; Linton, Sperry, & Williams, 2002). Fall precipitation provides water for seed germination before winter and establishment of new plants in spring.

After development of land, impervious surfaces prevent precipitation from infiltrating where it lands. Bioretention is intended to provide a pathway for precipitation that falls on vast impervious surfaces to infiltrate into the groundwater system at designated points. In order to accomplish this hydraulically, short-term storage must be engineered to allow a large volume of water to infiltrate over a relatively small footprint. Modifying the design recommendation from Hsieh and Davis (2005) so that a storage layer of gravel or expanded shale media replaces the “filtration layer” consisting of sand and sandy loam soils is an efficient way to achieve this temporary storage space. Precipitation runoff can then be routed to the gravel storage reservoir and then slowly infiltrated into the native soils below, where small pockets of underground storage will naturally form. Appropriately selected native plants can root through the gravel storage reservoir and into the native soils to access these small pockets through the summer months. The storage layer will be oxygen limited when saturated and should promote denitrification before infiltration (Brown & Hunt, 2011; Lucas & Greenway, 2010). Deep-rooting shrubs can provide carbon to microbes below the storage level, promoting nutrient immobilization as water infiltrates below the storage layer

Mulch is commonly prescribed as a soil covering of bioretention because of its ability to sorb pollutants from stormwater (Davis et al., 2009; Dietz & Clausen, 2005; Hsieh & Davis, 2005). However, mulch requires frequent maintenance and replacement. In xeric climates, mulch becomes sun-faded and loses its aesthetic quality, then must be disposed of and replaced because the dry conditions do not provide an environment that promotes decomposition (Sue Pope, personal correspondence; 2010). Decorative gravel is often twice the cost to install compared to bark mulch. However, 4 cm to 10 cm of cobble or gravel does not need to be replaced and does not require clean-up or maintenance after a large flood event because it does not float. Further, light-colored rock covering can increase the albedo, or solar radioactive reflectance, of a site and decrease surface temperatures relative to bark mulch-covered areas, reducing water demand for the plants (Montague & Kjelgren, 2004). In spite of the higher cost of installation, using gravel as a top layer reduces maintenance, and therefore lessens the whole life cost of the facility (Houdeshel, Pomeroy, Hair, & Moeller, 2010).

Researchers have expressed concern that nutrient-rich topsoil installed in bioretention leaches nutrients into surface waters (Davis et al., 2009; Dietz & Clausen, 2006; Hunt et al., 2006). Many plants native to xeric climates are adapted to a wide range of soils with high infiltration rates maintained by ecological influences and low nutrient content (Barbour & Billings, 2000; Whisenant, 1999; Wilcox et al., 1990, 1992). Therefore, a sandy loam topsoil of low nutrient content, similar to the recommendations summarized in Davis et al. (2009), is recommended for bioretention in xeric climates. The native soils excavated from the site are likely sufficient to grow locally native plants. Using in-situ soils reduces costs associated with hauling and reduces transport of required resources to the site, improving the sustainability of the project. If in-situ soils are high in clay content, mixing sand or top-soil with the native soils may be preferred to improve infiltration, but upon establishment, the engineered ecosystem is expected to maintain high infiltration rates even in high-clay soils (Wilcox et al., 1992).

Sizing the storage layer appropriately is crucial to the functionality and cost of the facility. As size increases, so do costs associated with excavation and imported fill materials (Houdeshel et al., 2010). If a storage layer in a cold desert is undersized, the facility cannot infiltrate enough runoff in the spring to sustain plants through the summer. This is less of a concern for warm deserts because the rain falls during the growing season and plants transpire the moisture as soon as it falls. However, the storage layer should not be reduced because warm deserts often experience larger storms at greater intensities than cold deserts (Herschman, 2008) and downstream physical and ecologic impacts of increased flow rates and volumes as a result of urbanization on receiving waters has been well documented (Brix, 1993; Davis et al., 2009; Emerson & Traver, 2008; Hollis, 1975; Roesner et al., 2001). The physiological restraints of plants, rather than cost, should decide storage layer design for xeric regions. Many shrubs and trees adapted to arid climates are known to root through thin gravel layers to access deeper spoil moisture. However, increasing the depth of the storage layer may restrict the ability of some species to successfully root through the storage layer. Because of this, a standardized storage layer depth of 0.6 m is recommended for both warm deserts and cold deserts to maximize storage efficiency and to best facilitate plant performance.

Based on these concepts, we recommend a bioretention design modified from Hsieh and Davis (2005) that includes a gravel storage layer to maximize storage capacity instead of sand. Our recommendation for an unlined bioretention cell includes, from the bottom up, a 0.6 m gravel storage layer, a 0.5 m topsoil layer, weed barrier, and a 0.03 m to 0.10 m decorative gravel on top (Figure 2.3). The storage layer provides short-term storage volume during and after precipitation and/or melting events to allow infiltration of a large drainage area over a small footprint. The topsoil layer provides a medium for plants to establish during the 1st year, and to develop an extensive web of roots to facilitate AMF that will capture and store nutrients that flood the site. The weed barrier acts to reduce evaporative losses and prevent unwanted weeds that can rapidly deplete soil moisture content (Mack, 1981). Light-colored decorative gravel is prescribed here instead of mulch to reduce maintenance, fortify the site against damage during flooding, and reduce albedo. Mixtures of sizes, colors, and textures can be used to achieve a desired appearance or architectural objective. Large boulders can also be placed within the facility and curbing can be placed around the facility to protect vegetation against trampling.

2.7 Bioretention sizing recommendations

Bioretention and other LID approaches to stormwater management are typically sized to capture small, frequent rain events (Davis et al., 2009) because research suggests capturing the initial flush from an impervious surface can greatly reduce pollutant loading to surface waters (Davis et al., 2009) and because small, frequent floods cause more damage to streams than large, infrequent floods (Hollis, 1975). However, the Technical Guidance on Implementing the Stormwater Runoff Requirements for Federal Projects under Section 438 of the Energy Independence and Security Act of, 2007 (EISA) requires that federal projects “manage on-site the total volume of rainfall from the 95th percentile storm or managing on-site the total volume of rainfall based on a site-specific hydrologic analysis” to meet predevelopment hydrology (U.S. EPA, 2011). The 95th percentile “storm” is 20 mm over one day for Salt Lake City, UT and 25.4 mm over 1 day for Phoenix, AZ, according to the methods recommended by USEPA (2011) and explained by Hirschman and Kosco (2008).

In order for urban water managers in xeric climates to understand which measure of control is most appropriate, the natural hydrology of each site must be evaluated. A generalization can be made from a study by Wilcox et al. (1990) that measured the annual runoff from two predevelopment sites in southeastern Idaho and Arizona. The Idaho site was characteristic of a Great Basin sagebrush grassland (cold desert) averaging 240 mm of precipitation annually over 20 years and the Arizona site was characteristic of a Sonora Desert shrubland (warm desert) averaging 267 mm of rainfall annually over 7 years. Wilcox et al. (1990) found that during the research period the cold desert produced an average 2 mm of runoff per year and the warm desert produced an average of 20 mm of runoff each year (Wilcox et al., 1990).

Once the natural hydrology of a region is understood, a system can be engineered to mimic the natural processes driving this hydrology. Based on the physiological needs of regionally appropriate plants, we recommend a 0.6 m depth for the bioretention storage layer. Given a constant storage depth, a garden area to drainage area (GA:DA) relationship can be developed for various storm depths if the precipitation to runoff relationship (P:R) is defined.

Many LID design references that are currently available suggest that bioretention sizing be based on results from a single design storm that targets a statistically determined storm depth over a given time (North Carolina State University, 2011; Prince George's County, 2002). However, long-term simulations are now being encouraged as more appropriate (Sitler & Clark, 2011).The results of both approaches are compared in order to recommend an appropriate DA:GA ratio for xeric climates. The TR-55 method expresses ground surface conditions by a unitless curve number, where highly impervious, smooth surfaces receive a number close to 100 (concrete pavement = 98) and pervious, rough surfaces where runoff is slow are assigned lower numbers (healthy meadow = 30), then predicts runoff as a function of precipitation inputs, drainage area, and ground surface condition (U.S. Department of Agriculture, 1986). The TR-55 method was used to calculate GA:DA relationships for a warm desert site and a cold desert site because the EISA requires that a bioretention facility be sized to treat the 95th percentile "storm," and the TR-55 method is referenced as the appropriate method for this type of analysis in bioretention design guidelines (North Carolina State University, 2011; Prince George's County, 2002). Results and assumptions used in the TR-55 calculations are given in Table 2.2.

Continuous modeling should be used in addition to single storm event modeling to verify that a bioretention facility is sized appropriately to satisfy site management goals (Sitler & Clark, 2011). In order to compare the long-term results of bioretention facilities sized according to the EISA requirements, the USEPA Storm Water Management Model 5.0 (SWMM) was used to conduct a 20-year continuous simulation (1990 to 2010) for a Great Basin site and a Sonora Desert site. A storage unit with infiltration and an overflow outlet was used to model the bioretention units. Precipitation data from Salt Lake City, UT (NOAA station 427598) was used to represent cold desert precipitation and guide design parameters; precipitation data from Phoenix, AZ (NOAA station ID 026481) were used to represent warm desert precipitation and guide design parameters. Model parameters used to simulate the bioretention cells for the warm desert and cold desert are given in Table 2.3. Results are reported in Table 2.4. From this analysis, the recommendations for sizing a bioretention facility according to the 95th percentile storm match predevelopment hydrology for Salt Lake City, UT because pre-development hydrology produces little to no surface runoff. However, if the goal of the facility is to capture 95% of the annual postdevelopment runoff, then the TR-55 method will over-size the facility. The long-term model shows that capturing the 95th percentile storm captures 95% of the annual runoff for Phoenix, AZ, but does not match the 20 mm of annual runoff measured by Wilcox et al. (1990). In order to achieve an annual runoff of 20 mm, the DA:GA must be no greater than 9:1 and would capture 98% of total annual runoff.

2.8 Conclusion

The survey of bioretention, plant physiology, and wildland restoration supported by leaf gas exchange measurements and hydrologic modeling exercises presented here suggest that if bioretention system design is ecologically based, then bioretention can be utilized to mitigate negative effects of urban stormwater runoff and reduce per-capita water demand by providing a zero-irrigation alternative to traditional landscaping. Addressing both stormwater runoff and easing demand for regional water resources will benefit the fragile desert ecosystems that surround the fastest growing population centers in the country.

More research is needed to measure transpiration and evaporation in bioretention cells in all climates, but given the high vapor pressure deficit of xeric climates, evaporation rates likely account for a significant loss in bioretention systems. Knowing evapotranspiration rates over the course of the growing season can help to optimize facility sizing to more precisely supply the water needed to sustain plants that provide stormwater treatment. The carbon budgets of these systems are also of concern. More research is needed to confirm that the wetting and drying experienced by bioretention in xeric climates are net carbon sinks and not contributing to global climate change driven by increased atmospheric CO_2 or N_2O levels. When these relationships are quantified at the garden scale, models can be developed to predict urban effects on carbon sequestration, water savings, and stream health improvements. With a better understanding of how bioretention might affect these regional issues, water resources managers can better decide the appropriate rate and scale to which bioretention should be implemented in xeric systems.

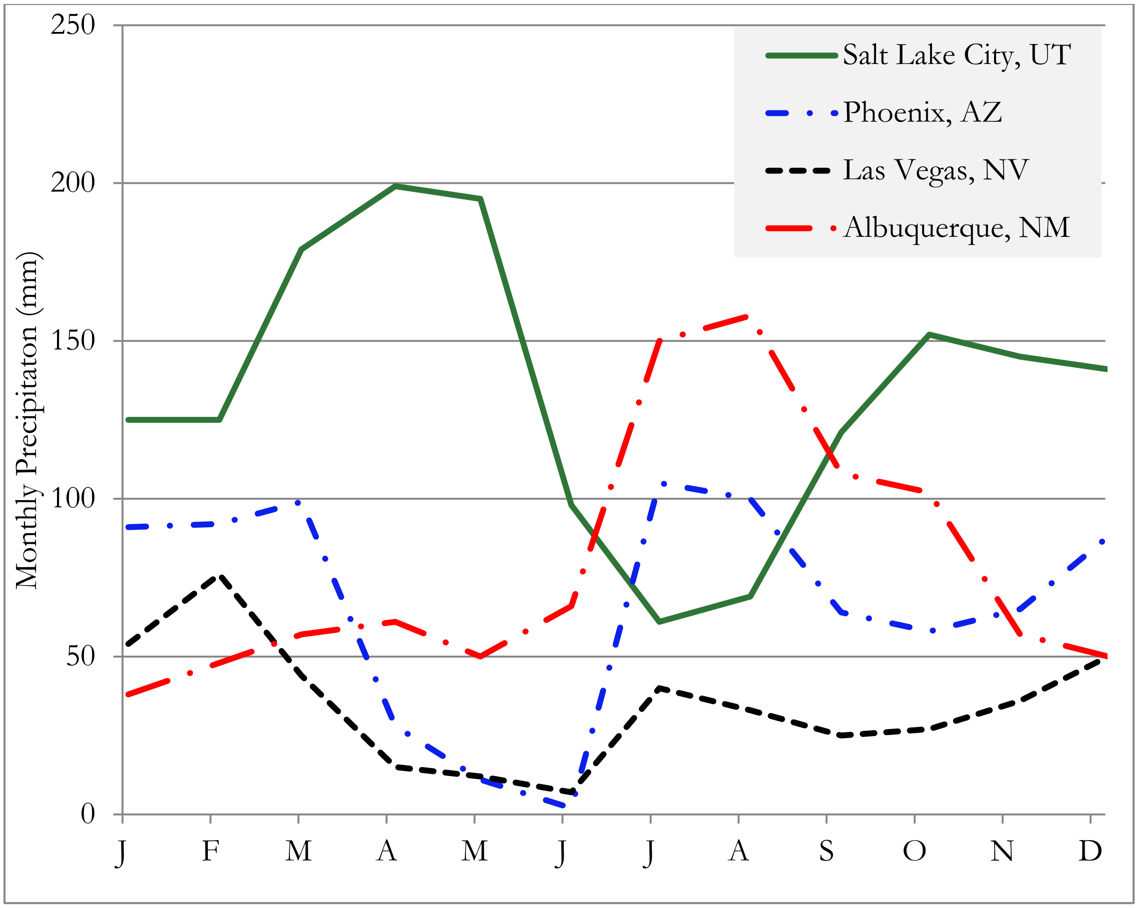

Figure 2.1. Precipitation patterns for four arid to semi-arid western cities. Salt Lake City receives the majority of precipitation in the winter and early Spring where Phoenix and Albuquerque receive most of their precipitation in late summer. Las Vegas receives some precipitation as summer monsoons in summer, but also receives some precipitation from large frontal winter storms with no consistent wet season.

Table 2.1 Recommendations for plants to be used in bioretention in arid climates. Recommendations are based on literature review and favorable plant traits. Rooting pattern codes signify: E = shallow, extensive; P = phreatophyte; B = bulb. Under the Form column: G = bunchgrass; S = shrub, T = tree, F = perennial flowering forbs. Under the Regions column: 1 Basin and Range (Salt Lake City, UT, Boise, ID, Denver, CO); 2. Mojave (Las Vegas); 3. Warm Deserts (Phoenix, AZ) and 4. Coastal Southern California (Anaheim, CA, San Diego, CA).

Species name	Common Name	Form	Rooting Pattern	AMF Host	Region
Schizachyrium scoparium	Little bluestem	WG	E	Likely	1,2,3,4
Bouteloua gracilis	Blue gramma	WG	E	Likely	1,2,3,4
Sorghastrum nutans	Indiangrass	WG	E	Likely	1,2,3,4
Pascopyrum smithii	Western wheat grass	CG	E	Likely	1
Pseudoroegneria spicata	Bluebunch wheat grass	CG	E	Likely	1
Rosa woodsii	Wood rose	S	E	Likely	1,2,3,4
Rhus Aromatica	Fragrant sumac	S	E	Yes	1,2,3,4
Fallugia paradoxa	Apache plume	S	E	Likely	1,2,3
Chrysothamnus nauseosus	Rubber rabbitbrush	S	P	Likely	1,2,3
Atriplex canescense	Four-winged saltbrush	S	P	Likely	1,2,3
Juniperus osteosperma	Utah juniper	T	E and P	Yes	1,2,3
Cercocarpus ledifolius	Curly mahogany	S, T	P	Likely	1,2
Larrea tridentata	Creosote	S	E	Likely	2,3
Artemisia tridentata	Sagebrush	S	E and P	Yes	1,2
Cercocarpus montanus	Mountain mahogany	S, T	P	Likely	1,3
Eschscholzia californica	California poppy	F	E and P	Unknown	1,4
Epilobium angustifolium	Fireweed	F	E	Yes	1,4
Baileya multiradiata	Desert marigold	F	P	Unknown	2,3
Eschscholzia glyptosperma	Desert poppy	F	P	Unknown	2,3
Tulipia sp.	Tulips	F	Bulb	Unknown	1
Arctostaphylos glauca	Bigberry manzanita	S, T	E	Likely	4
Solidago californica	California goldenrod	S	E	Likely	4
Delphinium bicolor	Low larkspur	F	E	Unknown	1

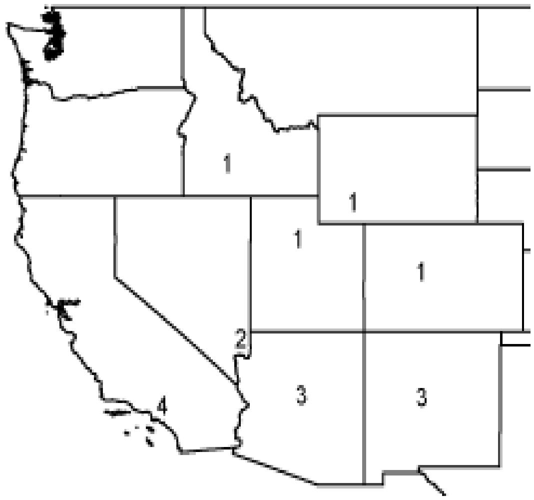

Figure 2.2. Map displaying relative geographic locations of recommended regions in Table 3.1.

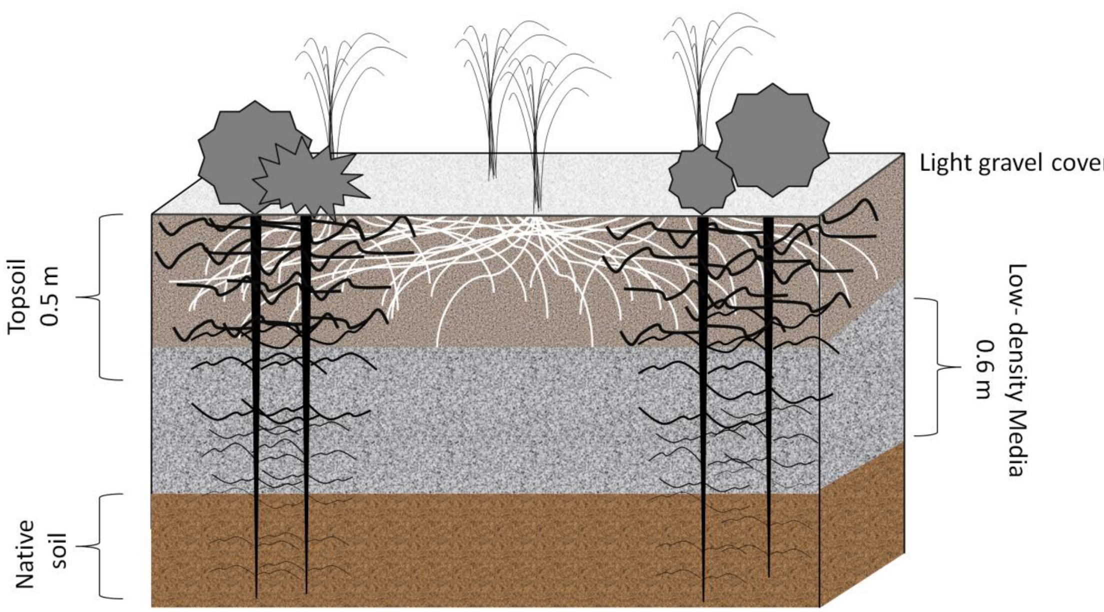

Figure 2.3. Design recommendation for bioretention in arid climates. Regionally native bunchgrasses and shrubs planted in 0.66 m topsoil over 0.66 m of low-density fill with light-colored gravel on top of a weed barrier to protect the surface from erosion damage. Dark roots represent deep-rooted shrubs; white roots represent bunchgrass roots.

Table 2.2. When calculating the required storage volume for a given runoff depth (Q) , the area of the bioretention cell scales linearly with drainage area. If design parameters of the bioretention cell are constant (media depth, vertical sides of storage layer, and homogenous porosity of storage media) garden area (GA) can be expressed as a percentage of drainage area (DA) for a given Q.

	Phoenix, AZ	Salt Lake City, UT
95th percentile "storm" (mm)	25.4	20
Calculated Q from TR 55 (mm)	20	15.2
Storage layer Depth (m)	0.2	0.2
Porosity of storage layer (%)	40	40
DA:GA	12:1	16:1

Table 2.3. Modeling parameters used to simulate long term performance of the EISA stormwater management regulations for warm (Phoeniex, AZ) and cool (Salt Lake City, UT) deserts. Infiltration parameters used for the warm and cool desert models were taken from the results of Wilcox et al. (1990) and Wilcox et al. (1992), respectively.

	Phoenix, AZ	Salt Lake City, UT
GA as % of DA	8.4 %	6.3 %
Functional Storage layer Depth (m)	0.08	0.08
K_s of storage unit	24.1	30
S_f of storage unit	79	72
Initial deficit	0.15	.015

Table 2.4. Results from Continuous SWMM results to measure long-term performance of a bioretention cell sized according to EISA guidelines for Salt Lake City, UT. Infiltration parameters for bioretention facilities represent ungrazed sagebrush grassland as measured by Wilcox et al., (1992).

	Phoenix, AZ			Salt Lake City, UT		
Annual Average Runoff to Bioretention (m^3)		1997			3862	
GA as % DA	11	8.4	7	6.3	5	4
Annual average capture (%)	98.4	95.5	92.6	99.8	97.2	94.2

CHAPTER 3

EVALUATION OF THREE VEGETATION TREATMENTS IN BIORETENTION GARDENS IN IN A SEMI-ARID CLIMATE

3.1 Abstract

Bioretention has become a well-established tool to reduce the transport of nutrients from impervious urban landscapes to sensitive riparian and estuarine habitats in mesic climates. However, bioretention in arid and semi-arid climates receiving less than 500 mm of annual precipitation has not been well tested. Nutrient removal performance was evaluated by building three identical 10 m^2 bioretention test cells with different vegetation communities: 1) an irrigated wetland, 2) an unirrigated upland vegetation community, and 3) a no-vegetation control. Synthetic stormwater was added to each cell to replicate the runoff from an average precipitation year falling on a 220 m^2 impervious surface in Salt Lake City, UT that contained 1.69 mg/l total nitrogen (TN), 0.28 mg/l NO_3, and 0.21 mg/l PO_4. Over the 1-year study period starting January 1, 2012, the average event mean concentration (EMC) for the effluent from the wetland treatment was 0.77 mg/l total nitrogen (TN), 0.23 mg/l NO_3, and 0.10mg/l PO_4. The average effluent EMC from the upland treatment was 1.43 mg/l TN, 0.87 mg/l NO_3, and 0.09 mg/l PO_4. Average effluent EMC from the no-vegetation control treatment was 3.15 mg/l TN, 2.5 mg/l NO_3, and 0.11 mg/l PO_4. This resulted in net nutrient reduction of PO_4 by all

treatments and net reduction of TN in the upland and control treatments. Net reduction of NO_3 was only achieved in the wetland treatment. The presence of plants did not appear to influence PO_4 treatment in this study. The wetland treatment retained more TN and NO_3 than the other treatments. However, this improved treatment came at the cost of over 12,000 liters (3,200 gallons) of irrigation to sustain this vegetation through the hot, dry summer. Increasing planting densities or integrating greywater treatment may improve nutrient removal in bioinfiltration gardens planted with upland vegetation in water-limited climates.

3.2 Introduction

Riparian ecosystems in arid and semi-arid regions are hotspots for biodiversity in areas that otherwise lack diversity and productivity. Because riparian ecosystems act as sinks for nutrients, pollution, and other materials, they are at an increased risk to changes on the landscape, especially the expansion of heavy urban land use. Opportunities to re-establish these ecosystems after they are degraded are often limited by water availability and other resources, thereby increasing the length of time between degradation and recovery (Gasith & Resh, 1999; Schwinning et al., 2008; Whisenant, 1999). The fasted growing populations in the United States are in the most arid climates (U.S. Census Bureau, 2005). The expanding urbanization in the arid West is replacing natural lands with impervious surfaces that increase urban runoff volume and event frequency (Hollis, 1975; Konrad & Booth, 2005; Paul & Meyer, 2001; Walsh et al., 2009). Poor air quality from particulate dust and the burning of fossil fuels in heavily populated basins common in the arid West increases dissolved N levels in precipitation (Buchanan & Honey, 1994;

Burian et al., 2001; Galloway et al., 2003; Pataki et al., 2006; Taylor et al., 2005). Fertilizers used on lawns and petrochemical residues found in urban systems can also increase nitrogen (N) and phosphorus (P) levels in streams so that these nutrients are no longer limiting to primary production (Burton & Pitt, 2010; Eriksson et al., 2002; Hultine et al., 2008; Schade et al., 2002). Receiving waters down-stream from rapidly growing population centers are at a serious risk of erosion, eutrophication, and invasion from nonnative species; each of these consequences of untreated stormwater runoff further compounds water resources availability in systems where demand often exceeds local supply (Hultine et al., 2009; Hultine et al., 2010; Hultine et al., 2008; Konrad & Booth, 2005; Pataki et al., 2005; Rickey & Anderson, 2004; Stromberg et al., 1996).

Bioretention is a form of Low Impact Development (LID) that collects stormwater runoff from impervious surfaces in a specially designed garden built to maximize ecological treatment of nutrients and other pollutants (Davis et al., 2009; U.S. EPA, 2013). Bioretention was first implemented as a stormwater control device in mesic climates (that receive 750 to 2000 mm of annual precipitation), and has been demonstrated to reduce peak flows of stormwater runoff and nutrient loading to receiving waters (BMPDatabase.org, 2012; Bratieres et al., 2008; Chen et al., 2013; Collins et al., 2010; Davis, 2007; Dietz & Clausen, 2005; Hatt et al., 2009; Hunt et al., 2006; Hunt et al., 2008; Kim et al., 2003; Li & Davis, 2009; Prince George's County, 2002). These systems utilize various designs of media layering and outlet controls to facilitate ecological N and P immobilization during storm events by wetland vegetation communities (Bratieres et al., 2008; Brown & Hunt, 2011; Davis et al., 2009; Henderson, Greenway, & Phillips, 2006; Hsieh & Davis, 2005; Hsieh, Davis, & Needelman, 2007;

Hsieh, Davis, & Needelman, 2007; Lucas & Greenway, 2008, 2010). However, in spite of requests from federal agencies and other watershed protection advocates (Davis et al., 2009; Transportation Research Board, 2013; U.S. EPA, 2013), little research has been conducted to determine how well these systems could improve water quality in arid and semi-arid climates.

The biggest challenge to designing bioretention in arid and semi-arid climates is sustaining vegetation through the long hot and dry periods that characterize these regions. The bioretention designs and vegetation communities recommended for mesic climates are not sustainable in arid and semi-arid climates. The use of irrigation for stormwater management in these regions is contrary to the goals of LID. Water use often exceeds local water supply such that most urban areas in the western United States import large volumes of water through interbasin transfers. Piping water out of one watershed and into another to irrigate ornamental landscaping dramatically alters natural hydrological processes at the regional scale.

Houdeshel et al. (2012) proposed design guidelines for bioretention stormwater treatment facilities in water-limited climates based on studies that describe biogeochemical processes such as vegetation evapotranspiration (ET) and nutrient cycling in water-limited ecosystems. These guidelines include the use of regionally native upland vegetation adapted to water-limited climates is recommended instead of wetland vegetation that has substantially higher water demands and for the routing of stormwater to a subgrade gravel storage layer instead of allowing stormwater to pond on the surface of bioretention facilities. Houdeshel et al. (2012) demonstrated the hydrological functionality of their suggested design, but the nutrient treatment performance of using upland vegetation in bioretention has not been tested.

The purpose of this study was to quantify N and P treatment by bioretention in a semi-arid climate. We compared the treatment capacity of the bioretention design recommended by Houdeshel et al. (2012) with a wetland vegetation community suggested for use in bioretention in more mesic climates and against a media-only system without plants. We predicted that a wetland community would achieve better N and P reduction than an upland shrub and bunchgrass community, but at a cost of requiring substantial supplemental irrigation.

3.3 Methods

3.3.1 Site description

Three bioretention cells were constructed in 2010 to test the removal capacity of different vegetation communities on N and P in stormwater at the Green Infrastructure Research Facility on the University of Utah campus in Salt Lake City, UT. Each cell was sized based on recommendations for bioretention design in semi-arid climates from Houdeshel et al. (2012) to capture 95% of annual runoff from a 220 m^2 (2400 ft^2) parking lot. The vegetation treatments tested were 1) a media treatment without plants, which will be referred to as the “Control” treatment, 2) an upland native community that did not require irrigation in semi-arid climates, or the “Upland” treatment, and 3) a wetland community that required irrigation, or the “Wetland” treatment. The vegetation planted in the Upland and Wetland treatments are listed in Table 3.1. Because Lucas and Greenway (2008) suggested that N treatment may be a function of ecological community establishment, each site was allowed to establish for 15 months before the experiment began. The site is at 1,480 m (4,850 ft) above sea level and averages less than 400 mm of precipitation annually. The precipitation pattern is characterized by snowy winters, cool and rainy springs, and extended dry periods with little precipitation throughout the hot, dry summer.

The type of media and drain configurations of bioretention units have been demonstrated to significantly affect N removal performance (Brown & Hunt, 2011; Davis et al., 2009; Hsieh & Davis, 2005; Hsieh et al., 2007; Lucas & Greenway, 2010). To address this, media layers and drain configurations of the three tested cells were identical. Each cell contained a 0.95 cm (3/8”) expanded shale reservoir (porosity = 0.45) with a storage capacity of 2,800 liters (740 gallons) and was lined with an impermeable barrier to facilitate effluent capture for nutrient analysis. Soil texture in the top 0.6 m of the cell was composed of 63% sand, 23% silt, and 14% clay. The final soil mix contained 0.8 g TN/kg soil; soil P or C was not tested. Each cell had a middrain 0.6 m from the bottom of the cell and a drain pipe below the gravel layer at the bottom of the cell for effluent collection and drainage. In this study, the middrain was closed and effluent was only collected from the lower underdrain. Effluent flow rate was kept constant for all three cells throughout the study to replicate the measured infiltration rate of a near-by bioinfiltration garden described by Stephen (2012). As a result, the gravel storage layer drained in approximately 24 hours, depending on the seasonally-adjusted volume of stormwater added.

A weather station was located on site and daily maximum and minimum temperatures were collected using a Vaisala HMP 45 AC humidity and temperature probe

(Vaisala, Woburn, MA, USA), placed approximately 2 m above the ground surface. Temperature was measured every 30 seconds and stored as daily maximum and minimum temperatures by a Campbell CR10X-2M data logger (Campbell Scientific, Logan, UT, USA).

3.3.2 Synthetic storm treatment

The site experiences high variability of annual precipitation. To assure that the study was representative to the local climate and not an abnormally wet or dry year, synthetic storms were created by mixing tap water with a nutrient source in 2,080 liter tanks, which were then applied to the cells in a manner that replicated average monthly precipitation patterns for Salt Lake City, UT. The number of storms and size of storms applied in each month, starting January 1, 2012, are shown in Table 3.2. The storm sizes and frequencies were calculated from hourly precipitation records from 1990 to 2010 at the National Oceanic and Atmospheric Administration (NOAA) Salt Lake City, UT, Airport weather station (Station ID 42-759800). Storm depth was calculated by dividing the monthly average precipitation by the monthly average number of storms; storm runoff volume was calculated by multiplying the monthly storm depth by a 220 m^2 parking lot. Storms were defined as precipitation events greater than 1.0 mm with an interstorm period of 6 hours (Iowa Department of Natural Resources, 2009).

Irrigation was applied to the Wetland treatment in addition to the storm runoff simulations. The wetland cell was irrigated an average of three times per week, with an average delivery of approximately 300 liters (80 gallons) per irrigation event for 13 weeks from the beginning of June through August. Irrigation was stopped in September under the assumption that the storm simulation schedule would provide sufficient water to sustain the wetland plants during the cooler month, although some evidence of water stress was apparent in *Typha sp.*, *Phragmities sp.*, and *S.exigua* species.

Each runoff event was simulated with a synthetic stormwater blend targeting the average values of N and P concentrations found in urban runoff by Burian et al. (2001), Taylor et al. (2005), Dietz and Clausen (2005), Hunt et al. (2006), and Sharkey (2006) in Table 3.3. Target N and P concentrations were achieved by mixing SuperSoil™ Garden Amender soil amending mulch (Rod McLellan Co, Marysville, OH) with tap water at a ratio of 1 kg mulch per 500 liters of tap water in a 2080 liter tank. Average influent nutrient concentrations for the entire study period are shown in Table 3.3.Turbulence is high in the tank during filling and complete mixing of N and P was verified in preliminary tests. Because some variability is expected, influent was tested for actual N and P levels during each synthetic storm.

3.3.3 Effluent collection and nutrient analysis

To test the prediction that effluent from the Wetland treatment will contribute the least amount of N and P to receiving waters, effluent from one synthetic storm was collected from each cell and tested for NO_3, NO_2, total nitrogen (TN), NH_4, and PO_4 once per month over 1 calendar year. Organic N (ON) was calculated by: ON = TN-(NO_3 + NO_2 + NH_4). An ISCO auto-sampler (Teledyne-ISCO, Lincoln, NE) was connected to a custom-made tipping bucket flow gauge by a PRX 11500 reed switch (HIS sensing, Chickasha, OK) to collect 250 ml samples from the effluent in 40 liter increments. A 1” pvc pipe drained each cell to the tipping bucket, which tipped after 2.2 liters drained into the bucket. The bucket dumped the collected water into a funnel that drained to the bottom of a 300 ml beaker so that the water from each tip flushed out the water previously in the beaker. This insured the auto-sampler was collecting a discrete sample every 40 liters. The ISCO auto-sampler was programmed to draw a 200 ml sample every 18 tips. The three cells were flooded simultaneously for the majority of the study. Equipment failure occasionally required that one storm be run on the subsequent day. On these occasions, similar atmospheric conditions occurred on both days storms were synthesized to ensure comparison between flood events was appropriate. Synthetic stormwater influent and effluent samples were collected and filtered with a 450 μm glass filter on the day of collection and refrigerated to minimize species transformation; all analysis was completed within 24 hours of filtration.

NH_4 was tested using the TNT™ 830 ULR NH_4 method that is EPA-certified to be accurate between 0.015 and 2.0 mg/l (Hach™ Company, Loveland, CO). The method utilizes a colorimetric analysis that tests NH_4 directly and was measured on a Hach 6500 spectrophotometer. The TN analysis was performed using Hach™ persulfate digestion method 10208, accurate between 1mg/l – 16 mg/l, and measured using a Hach 6500 spectrophotometer (Hach™ Company, Loveland, CO). NO_3, NO_2, and PO_4 were analyzed on a Metrohm 881 Compact IC (Metrohm, USA, Riverview, FL) capable of measuring 0.002 mg/l for the three ions tested. Two stock check standards were established at the beginning of the study and run with samples each month to verify that all instrumentation remained accurate.

Total effluent mass (TEM) from the treatment cells to receiving waters, flow-weighted event mean concentration (EMC), and total mass reduction (TMR) were calculated for one storm from each treatment cell each month for TN, ON, NO_3, NO_2, NH_4, and PO_4. To calculate TEM, nutrient concentration of each sample was multiplied by 40 liters then summed (samples were collected in 40 liter increments). The flow-weighted EMC was then calculated by dividing the TEM by the total effluent volume. TMR was calculated by subtracting the total effluent mass from the total influent mass. Monthly TEM and TMR was then estimated by multiplying results for the tested storm event by the number of storms simulated in that month (Table 3.2); yearly TEM and TMR are reported as the sum of each monthly value.

Due to equipment malfunction, only one synthetic storm was administered in June as described in Table 3.2. The remaining two storms were adjusted to match the target July storm volumes. To compensate for this in the TEM calculations, the EMC for May was multiplied by six storms instead of five. Influent loading for all three June storms were summed and included in the annual TMR calculations. No outflow was produced in the second two June storms, and TEM is reported as zero for June.

3.3.4. Statistical analysis

Effluent EMCs were highly variable throughout the year. To understand this variability, regression models were used to describe patterns in this variability. The two variables tested for influence on EMC for all nutrients were cell age and atmospheric temperature. Cell age was chosen to determine if nutrient removal increased or if the cells were becoming saturated over the course of the experiment. Exponential regressions were used to explore relationships between these variables and monthly EMC because any rate of change observed throughout the study must, over the long term, either: 1) approach zero and level off, asymptotically approaching 100% TMR, or 2) flush all nutrients stored in the system, causing effluent mass to asymptotically approach 0% TMR.

Trends in TN, ON, NO_3 , and PO_4 EMC were analyzed using ANOVA to identify if nutrient removal improved or declined as a function of study duration. Because the study started in January, month of study also corresponds to the month of the year. The correlation between TN, ON, and NO_3 and atmospheric daily maximum were also analyzed using ANOVA. The correlation between PO_4 and atmospheric daily minimum were also analyzed using ANOVA. Strength of correlation was determined by the least squared method, producing an r^2 value for each exponential regression analysis.

3.4 Results

3.4.1 Mass reduction

The Wetland treatment provided the greatest TMR for all nutrients tested. Half of the 200 g of TN added to the system were retained, reducing the TN mass loading to receiving waters by 50%. A 31% mass reduction in NO_3 was observed in the effluent from the Wetland treatment (Figure 3.1). Conversely, both the Control and Upland demonstrated negative reductions, or net leaching, of NO_3. 60% of the PO_4 added to the Wetland was retained over the study period.

In the Upland treatment, TMR was observed for all nutrients except NO_3, of which 53 g were exported when only 32 g were added. In spite of the net leaching of NO_3, the Upland treatment still removed 9% of the influent TN. On the other hand, the Upland treatment reduced PO_4 loading by 50%, retaining 10 g of the 20 g that were added throughout the study period. In the Control treatment, mass reduction was achieved for ON, NH_4, and PO_4 with 40%, 78%, and 35% TMR, respectively. However, the Control treatment leached 63% more TN than was added because the removal of ON and NH_4 was negated by the six-fold export of NO_3 (Figure 3.1).

The variation of TN mass reduction between each treatment was driven by the variation of NO_3 concentrations in the effluent of each cell. The contribution of NO_3 to the annual average EMC was 0.18 mg/l representing 25% of TN in the Wetland treatment, 0.87 mg/l representing 62% of TN in the Upland treatment, and 2.5 mg/l representing 77% of TN in the Control treatment (Figure 3.2). Little difference was seen between the annual average ON EMC among treatments. The annual average ON EMC for the Wetland, Upland, and Control treatments were 0.51mg/l, 0.50 mg/l, and 0.63 mg/l, respectively. Effluent NH_4 and NO_2 contributed very little to annual average TN EMC, only representing 3% of TN in the Control treatment, 4% of TN from the Upland treatment, and 5% of TN from the Wetland treatment (Figure 3.2). The TN concentrations from all events were strongly correlated to NO_3 concentrations (r^2 = 0.95, p = 2.2 x 10^{-16}, F-statistic = 578) than for ON (r^2 = 0.28, p = 0.0008, F statistic = 13.7), supporting that the variation in TN is better explained by variation in NO_3 than by ON.

3.4.2 Time and temperature as controlling variables of EMC

The measured monthly PO_4 EMC decreased exponentially with age in the Wetland treatment (r^2 = .84, p = 0.0002), Control treatment (r^2 = .78, p = 0.0007), and Upland treatment (r^2 = .61, p = 0.007) (Figure 3.3). The correlation between the PO_4 EMC from the Upland treatment and age were less significant because the January 2011 PO_4 EMC was much lower than the PO_4 EMC from the Upland treatment for the following 4 months and for the PO_4 EMC from the Wetland and Control treatments from the same month (Figure 3.3).

Temperature was only observed to explain the monthly variability in TN and NO_3 EMC in the Upland treatment (Figure 3.4). TN and NO_3 EMC decrease exponentially ($r^2 = 0.53$, $p = 0.02$ and $r^2 = 0.73$, $p < 0.01$, respectively), or TN and NO_3 removal increases, as temperature increases (Figure 3.4). For all treatments, ON effluent concentrations were stable throughout the study and did not correlate with time or temperature for any treatment, indicating the correlation between TN and temperature in the Upland treatment was driven by the lower release of NO_3 at higher temperatures. NO_3 EMC in the Control and Wetland treatments did not appear to be influenced by the range of temperatures experienced during this study; similarly, temperature was not shown to have an effect on PO_4 EMC in any of the three treatments. The r^2 and p values for all relationships described are given in Table 3.4. Analysis of PO_4 EMC is shown against minimum daily temperature because the correlations were stronger and more significant than using daily maximum temperatures in the same analysis; likewise, the relationships between N treatment and daily maximum temperature are shown because they were stronger than the same analysis conducted with daily minimum temperature.

3.5 Discussion

3.5.1 Relative vegetation community performance

The Wetland treatment retained the most nutrients during the study period, thus providing the most protection for receiving waters from nutrients associated with urban stormwater runoff. However, in the climate where the study was conducted, the Wetland treatment required significant supplemental irrigation to sustain the vegetation, a management approach that does not align with the overall goals of LID. The Upland treatment that required no supplemental irrigation during the study demonstrated a net reduction in TN and PO_4. However, the Upland leached more NO_3 than was added. All treatments demonstrated net removal for NH_4, ON, and PO_4 over the study period. This can likely be explained by abiotic sorption. The expanded shale media used in this study is negatively charged, facilitating sorption of positively charged cation molecules such as NH_4 and ON. Likewise, media columns without plants have been shown to reduce PO_4 mass loading from stormwater (Hsieh & Davis, 2005; Hsieh et al., 2007). It appears the synthetic stormwater additions made during this study did not saturate the sorptive capacity of the media used.

3.5.2 Effluent nitrate likely a function of microbial activity

Soil microbes are known to drive major processes in the nitrogen cycle including assimilation, mineralization, and denitrification. In order to drive these processes, microbes need suitable soil temperature and adequate water and carbon. Soil moisture sensors within each cell showed that the soil seldom froze below 20 cm (data not presented), indicating that, while microbial activity is often correlated with temperature, microbial activity in the tested cells was seldom prohibited by freezing. Each cell received the same amount of water in the winter, spring, and fall. However, the Wetland treatment received supplemental irrigation in the summer and NO_3 EMC's were most similar between the nonirrigated Upland and irrigated Wetland treatments during this period. These patterns indicate that if microbes were indeed driving NO_3 removal, as suggested by previous studies (Bratieres et al., 2008; Brown & Hunt, 2011; Chen et al., 2013; Lucas & Greenway, 2008, 2010), then the availability of carbon to soil microbes was likely the mechanism driving the observed difference in NO_3 removal among the three treatments. While primary productivity was not directly measured, the above-ground biomass was observed to be much greater in the Wetland treatment than the Upland treatment. Similarly, the irrigated wetland vegetation also likely allocated more carbon to below-ground biomass, providing fuel to facilitate microbe-driven N cycling.

If soil microbial nutrient treatment is indeed limited by vegetation primary productivity in our test cells, then vegetation density may directly correlate to nutrient treatment. The Upland treatment cell was designed to mimic vegetation densities found in natural semi-arid upland systems. However, natural systems are water limited, and previous research indicates that the large impervious catchment areas provide ample water for native upland vegetation in bioinfiltration system (Houdeshel & Pomeroy, under review). It may be possible to improve nutrient treatment by an upland community by increasing vegetation density of upland vegetation well beyond a natural system to more closely match the primary productivity occurring in the Wetland treatment.

<u>3.5.3 Phosphorus treatment performance</u>

All cells showed that effluent PO_4 EMC declined over the study period, with little difference between treatments. Acknowledging that the lack of replication in this study prohibits quantitative comparisons, the results in Figure 3.3 suggest that short-term (one year) PO_4 treatment is independent of vegetation productivity or vegetation community type. Hsieh and Davis (2005) and Hsieh et al. (2007) demonstrated that PO_4 effluent concentrations varied greatly with different media configurations in column studies without plants, although Henderson et al. (2006) and Lucas and Greenway (2008) show that plants reduced PO_4 effluent concentrations relative to similar media without plants. The media in the systems tested here were all identically configured. Because the results of the three treatments tested here were similar, PO_4 removal was likely a function of the media properties. The cells tested here occupied 5 m^3 of soil. Henderson et al. (2006) and Lucas and Greenway (2008) found vegetation to significantly improve PO_4 treatment in much smaller soil volumes. Our results seem to confirm that media selection is more important than the presence of plants for PO_4 treatment. However, the systems tested here contained much greater soil volumes and similar influent phosphorus loading compared to those tested by Henderson et al. (2006) and Lucas and Greenway (2008). It is likely that in this study the abiotic sportive capacity of the media in the Control treatment was not saturated, and that the sportive capacity of the smaller soil volumes without vegetation used by Henderson et al. (2006) and Lucas and Greenway (2008) did reach saturation and so demand by plants contributed significantly to treatment. A longer-duration study that inputs enough PO_4 to saturate the sportive capacity of the Control treatment may demonstrate that vegetation does contribute to PO_4 removal similar to Henderson et al. (2006) and Lucas and Greenway (2008).

3.5.4 Treatment design and applications to water-limited environments

Design recommendations for water-limited environments by Houdeshel et al. (2012) encourage the use of bioinfiltration, or gardens that are allowed to infiltrate without under-drains. This experiment was conducted on three cells that were lined and under-drained. However, the outflow through the underdrain was regulated to replicate the hydraulic retention time of adjacent bioinfiltration gardens. Therefore, in this experiment the bioretention test cells functioned much like lysimeters that were lined to facilitate effluent sampling from drain ports rather than under-drains. One concern of bioinfiltration gardens is the leaching of contaminants to groundwater, especially where water tables are shallow. The experimental design allowed water to be collected and tested from the bottom of the gravel storage reservoir, providing insight to the quality of water that would have infiltrated below these facilities.

The impervious liner used in this study to collect effluent water samples may have reduced net primary production of the Upland treatment by changing the ecohydrology of the vegetation used in this treatment. The shrubs selected for the Upland treatment tested are known to construct root systems much deeper than the gravel storage layer in the design tested. In a setting without a liner, these shrubs could have accessed infiltrated soil water below the storage reservoir throughout the summer. Benefits to this may have included increased primary productivity by the shrubs and hydraulic lift (i.e., passive redistribution of deep soil water to drier soils near the surface through roots), which could increase bunchgrass primary productivity and extend favorable soil conditions for nutrient cycling by soil microbes during the dry summer months (Richards & Caldwell, 1987). The liner in the Upland treatment tested may have artificially reduced the NO_3 removal capacity of the system tested here by reducing net primary productivity. The liner did not likely affect nutrient removal in the Wetland treatment because this treatment was irrigated, making water available to the plants through the summer. Future studies should be designed in a way that allows the upland vegetation to access soil water below the gravel storage layer. This may require deeper test cells with a layer of soil underneath the gravel layer, or water sampling ports could be integrated into unlined bioretention gardens. Each of these approaches increases the complexity of the systems being tested, and would require careful thought before implementation. Vegetation density should also be considered as a treatment variable when testing the nutrient treatment capacity of upland vegetation in bioretention.

The Upland treatment resulted in a net export of NO_3, while the Wetland treatment was a net NO_3 sink. Research suggests that soil microbes are responsible for driving nutrient removal, (McClain et al., 2003; Stutz & Morton, 1996; Tao & Zhiwei, 2005; Whiteside, Digman, Gratton, & Treseder, 2012), and that many soil microbes depend on carbon resources fixed and transported below ground by vegetation (Gianinazzi-Pearson, 1996). Because of these relationships, increasing net primary productivity by plants in the upland treatment will likely increase soil microbial biomass by providing more below-ground carbon. This should increase NO_3 demand by the soil microbial biomass in the Upland treatment by increasing below-ground carbon resources needed to fuel microbial denitrification, thus improving NO_3 removal.

3.5.5 Nonpotable irrigation sources needed to optimize watershed protection

As tested, the effluent from the Wetland treatment contained less N and P than the effluent from the Upland treatment. But this treatment came at a cost of about 12,000 liters (3,300 gallons) of potable water that was applied over a 13-week period during the warmest months of the study. Signs of water stress were observed in the plants, indicating that the irrigation applied (1,500 $l \cdot m^{-2} \cdot w^{-1}$) was the minimum amount of water that could have sustained this vegetation community under the experienced climate. If the design of bioretention for xeric climates cannot be improved to achieve the water quality performance demonstrated by the Wetland treatment, a sustainable irrigation source must be identified to maximize watershed protection from urban stormwater.

We consider irrigating stormwater treatment facilities with potable water to be unpalatable in arid and semi-arid climates. However, expanding to waste water sources that could be treated by bioretention facilities is a compelling design question. This research supports further investigation of integrated uses of bioretention for both stormwater management and on-site greywater treatment from adjacent buildings. This integration of bioretention and greywater treatment could provide a water source to sustain wetland communities to maximize stormwater treatment potential and reduce the regional demands on wastewater treatment plants. Integrated stormwater and wastewater treatment is ideally suited for semi-arid, cold desert climates because storm events are infrequent, and extended drying periods likely degrade bioretention performance by reducing soil conductance, infiltration, and nutrient treatment capacity. A constant supply of greywater could provide needed nutrients and water for plants and microbes.

3.6 Conclusions and design recommendations

This research suggests that wetland vegetation in bioretention can be highly effective at reducing nutrient loading to receiving waters in arid climates, but at the cost of requiring substantial supplemental irrigation. Although the regionally native upland plant community captured less nutrient mass, it did not require supplemental irrigation, and may therefore provide greater ecosystem services than the wetland vegetation. Perhaps increasing vegetation density to two or three times greater than a natural upland ecosystem may increase nutrient treatment capacity without causing decreases in plant performance or survival during the summer. The expanded shale used in this study was likely responsible for the high PO_4 removal by all three treatments. This study should be extended to evaluate the long-term performance of the vegetated and nonvegetated treatments; likewise, comparing the PO_4 treatment capacity of different media of the same porosity as the expanded shale used here, such as gravel or pumice, should be conducted to determine the value of the increased cost associated with the expanded shale. Finally, the concept of combining stormwater and wastewater treatment has proven challenging in mesic climates at large scales, but integrating greywater and stormwater treatment in arid climates at the site scale may create a new direction for urban water sustainability.

3.7 Acknowledgments

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Table 3.1. List of prominent species found in the Upland and Wetland treatments. Alfalfa was naturally recruited into the garden as a weed, and is not recommended for use in bioretention. However, because it is a known nitrogen fixer, its presence is important and reported.

Upland Garden		
Species name	**Common Name**	**Form**
Schizachyrium scoparium	Little Blue Stem	Bunchgrass
Bouteloua gracilis	Buffalo Grass	Bunchgrass
Sorghastrum nutans	Indiangrass	Bunchgrass
Amelanchier utahensis	Utah Serviceberry	Shrub
Cercocarpus ledifolius	Curl-leaf mahogany	Shrub, Evergreen
Cercocarpus montanus	Mountain mahogany	Large Shrub
Artemisia cana	Silver sage	Shrub
Wetland Garden		
Species name	Common Name	Form
Juncus effuses	Common rush	Rush
Dactylis glomerata	Oarchardgrass	Bunchgrass
Typha sp.	Cattail	Bunchgrass
Phragmities sp.	Phragmites	Bunchgrass
Salix exigua	Coyote willow	Shrub, Tree
Medicago sativa	Alfalfa	Forb

Table 3.2. Synthetic storm protocol. The average storm volume represents the volume of water multiplied by a 220 m^2 impervious area for the given average storm depth. No abstractions are included.

Month	Average Number of Storms	Average Monthly Rainfall (cm)	Average Storm Depth (cm)	Average storm volume (liter)
January	5	3.33	0.66	1480
February	5	3.25	0.65	1440
March	5	4.39	0.88	1950
April	6	5.23	0.87	1940
May	5	4.39	0.88	1950
June	3	2.72	0.91	2020
July	2	0.09	0.05	100
August	2	1.37	0.69	1520
September	3	2.64	0.88	1960
October	4	3.48	0.87	1930
November	4	3.30	0.83	1890
December	5	2.84	0.57	1260
Total Annual Rainfall		37.03		

Table 3.3. Summary of nutrient values in stormwater runoff reported in previous studies. The average of these values was used as the target nutrient concentration for the applied synthetic stormwater.

Study	Location	TN	ON	TKN	NH_4	NO_x	DON	TP
Taylor et al. (2005)	Melburn, Aust.	2.13	1.1	1.39	0.29	0.74	0.6	
Hunt et al. (2006)	N. Carolina	1.35	0.57*	0.8	0.23	0.4		0.11
Burian et al. (2001)	s. California				0.37	1.42		
Dietz and Clausen (2006)	Connecticut	1.2	0.5	0.4	0.04	0.3		0.015
Sharkey (2006)	N. Carolina	1.89	1.41*	1.67	0.26	0.23		0.29
	Averages	1.64	0.80	1.07	0.24	0.62	0.60	0.14
	Measured average synthetic stormwater concentrations	1.69	1.0		0.43	0.29		0.21

*Indicates calculated values not reported by the initial study.

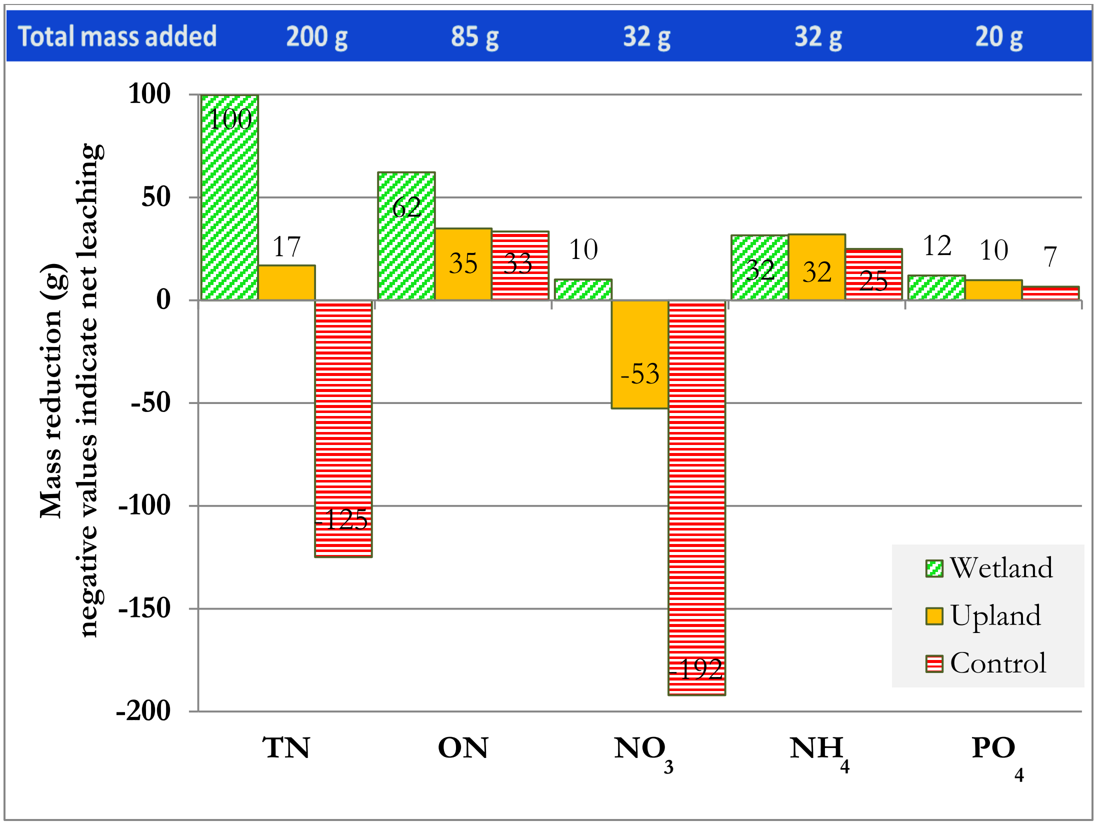

Figure 3.1. Total Mass Reduction (TMR) achieved by each treatment for each tested nutrient over the 1-year study. Negative values indicate net export, where effluent mass was greater than influent mass. Organic N (ON) is calculated as ON = TN – (NH_4 +NO_2 + NO_3). All other nutrients were directly measured as described in the Methods section.

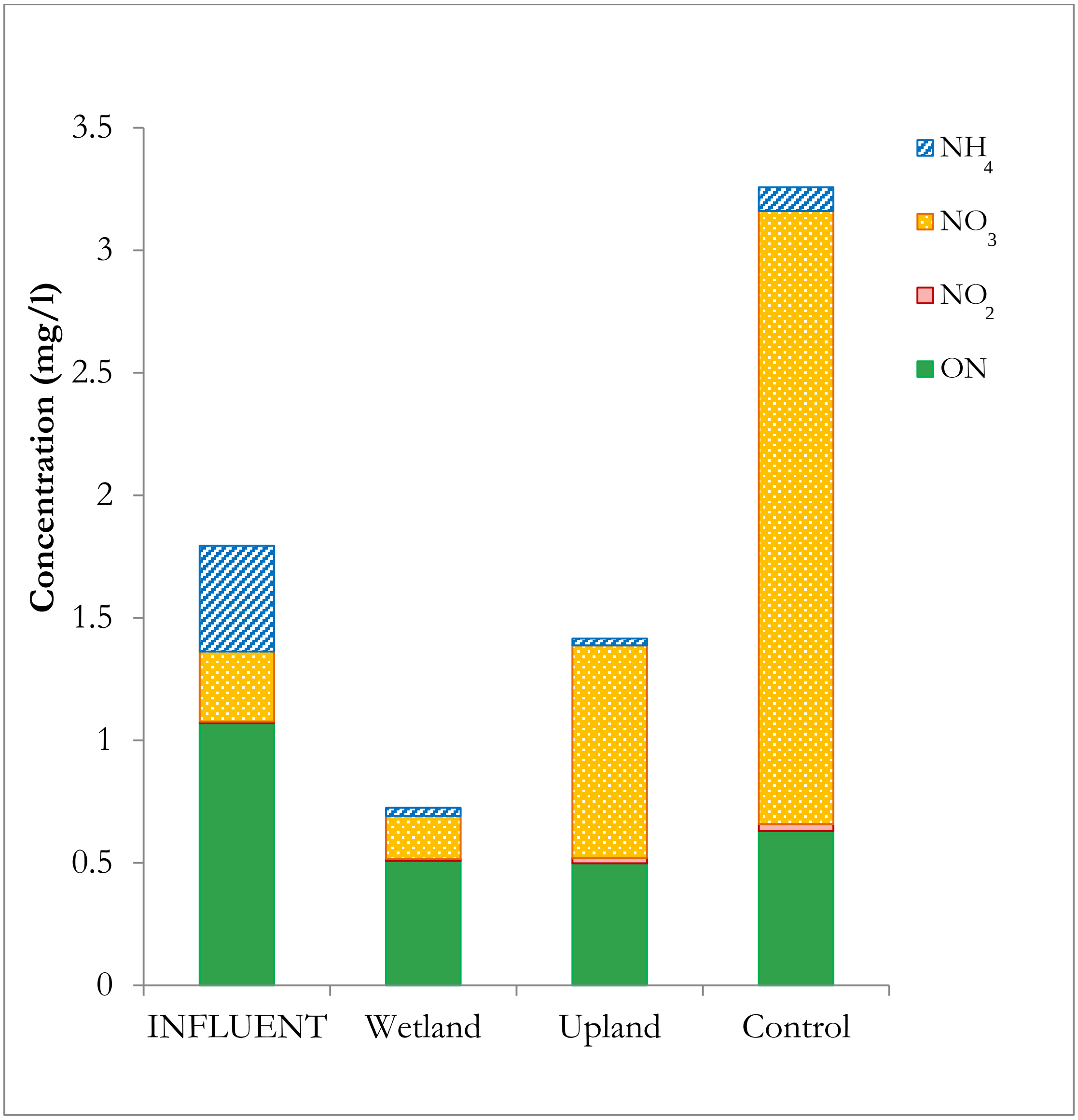

Figure 3.2. Cumulative contribution of each tested N species to average TN EMC for the synthetic stormwater influent and effluent from each treatment.

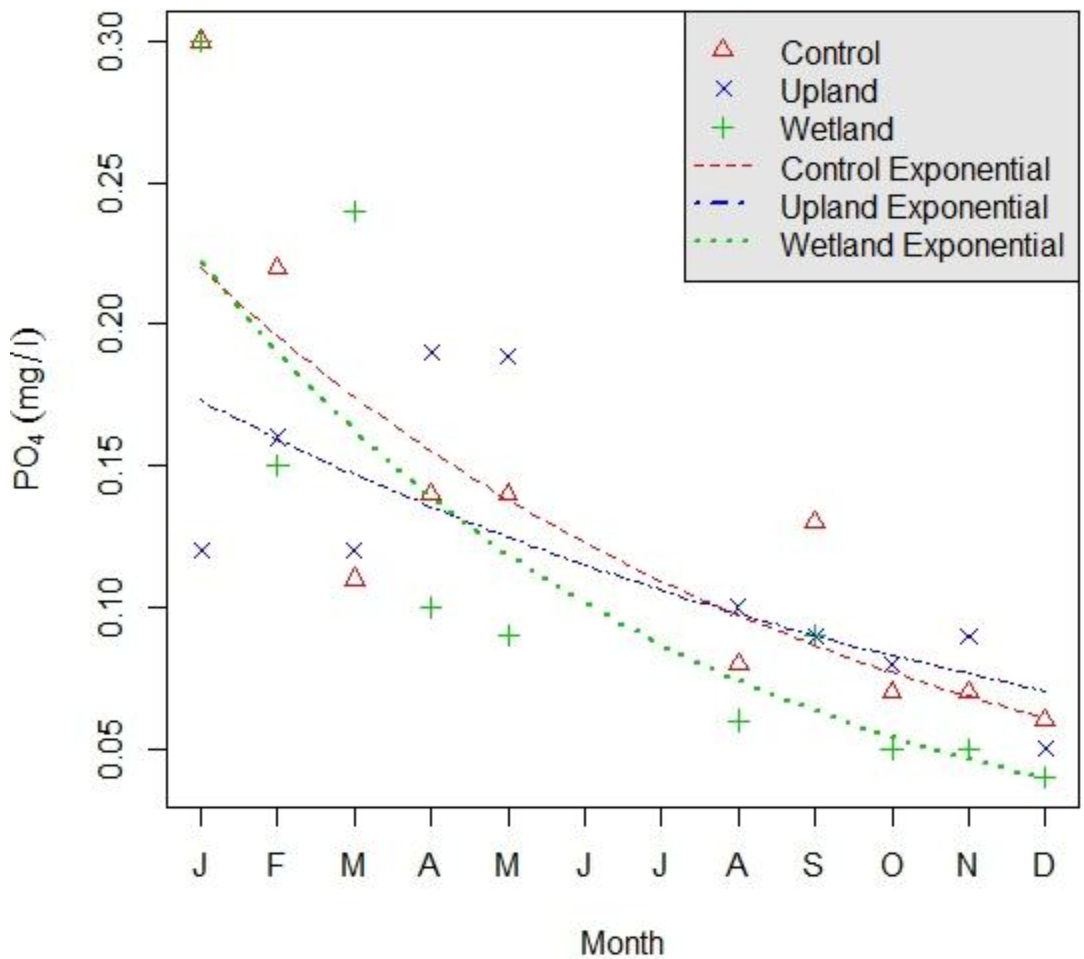

Figure 3.3. Effluent PO_4 concentrations for each treatment over the duration of the study. EMR improved in all gardens over time. The study started January 1.

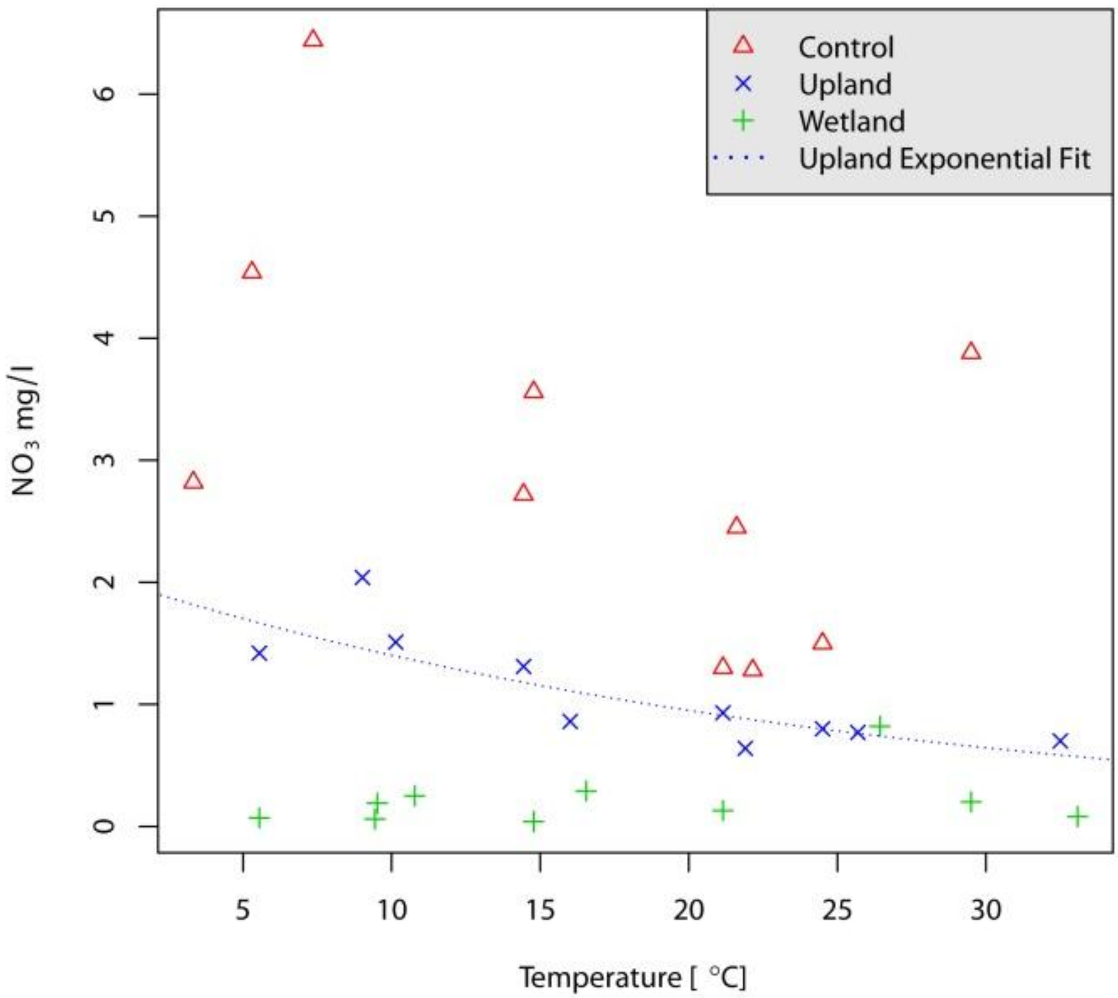

Figure 3.4. Monthly NO_3 effluent concentrations compared to daily maximum atmospheric temperature on the day of the synthetic storm application. Temperature was only observed to explain the observed variability in NO_3 EMC in the Upland treatment.

Table 3.4. Results of the ANOVA analysis for exponential regression fits for the relationships described. Highlighted values are significant at p>0.05 and F>5.3.

		Vs. Month			Vs. Temp		
		r^2	*T*-test *P*	*F*	r^2	*T*-test *P*	*F*
PO_4	Control	0.78	0.0007*	28.40*	0.01	0.44	0.67
	Wetland	0.84	0.0002*	42.46*	0.01	0.77	0.09
	Upland	0.61	0.007*	12.54*	0.00	0.86	0.03
		r^2	*T*-test *P*	*F*	r^2	*T*-test *P*	*F*
NO_3	Control	0.01	0.76	0.10	0.28	0.12	3.10
	Wetland	0.25	0.14	2.64	0.09	0.41	0.76
	Upland	0.08	0.43	0.69	0.73	0.001*	21.96*
		r^2	*T*-test *P*	*F*	r^2	*T*-test *P*	*F*
ON	Control	0.30	0.10	3.35	0.05	0.55	0.38
	Wetland	0.26	0.13	2.81	0.19	0.21	1.89
	Upland	0.30	0.10	3.51	0.02	0.72	0.14
		r^2	*T*-test *P*	*F*	r^2	*T*-test *P*	*F*
TN	Control	0.05	0.58	0.34	0.26	0.13	2.79
	Wetland	0.41	0.47	5.48	0.04	0.57	0.35
	Upland	0.20	0.20	1.94	0.53	0.02*	9.11*

CHAPTER 4

TRACKING NITROGEN THROUGH ENGINEERED BIORETENTION ECOSYSTEMS

4.1 Abstract

The capacity of vegetated bioretention systems to remove nitrogen from stormwater has been well demonstrated. However, the processes by which vegetation facilitates removal is not well understood. In order to verify that plants assimilate stormwater nitrogen into biomass, a ^{15}N-enriched synthetic stormwater blend was added to three bioretention gardens with three different vegetation treatments in May 2012 as part of an annual synthetic runoff regime modeled to represent an average precipitation year in Salt Lake City, UT. The different vegetation treatments were a wetland, an upland bunchgrass-shrub community, and a media-only treatment without plants. The ^{15}N label was clearly identified in both the upland and wetland vegetation, and the relative amount of the ^{15}N label recovered from the effluent was greater from the media-only treatment than the vegetated treatments. The results from this study suggest that the upland bunchgrass and shrub community are approaching a nitrogen saturation threshold, which may be delayed by increasing planting densities in future designs.

4.2 Introduction

Stormwater runoff from the built environment is one of the greatest challenges of modern water pollution control. Nitrogen (N) loading from urban stormwater runoff is a principal contributor to the impairment of rivers, lakes, estuaries, and coastal waters (U.S. EPA, 2012). Most freshwater and marine ecosystems are N limited (Elser et al., 2007), and adding anthropogenic N to these systems can lead to increased algal growth that causes eutrophication (Elser et al., 2007; Field et al., 1998; Jickells, 1998; Lee, 2002; Novotny & Witte, 1997; U.S. EPA, 2010a). Eutrophication can decrease dissolved oxygen in waterways and suffocate aquatic macro fauna, reduce flows in small and ephemeral streams, increase the temperature of receiving waters, and increase sedimentation in receiving waters (Diaz & Rosenberg, 2008; Galloway et al., 2003; Jickells, 1998; Novotny & Witte, 1997).

Green Infrastructure (GI) is one approach to treating stormwater on site. Bioretention stormwater management systems, a type of GI, are engineered ecosystems that capture and infiltrate or transpire stormwater where precipitation falls. Bioretention systems have been shown to reduce the volume of runoff from a site and to reduce the nutrient loading to receiving waters from urban areas (Davis et al. 2009). Davis et al. (2009) found the treatment of different N compounds in bioretention studies can be highly variable, with total N (TN) reduction reported in all studies cited. The International Best Management Practice (BMP) Database shows that on average, bioretention facilities significantly reduce TN and nitrate (NO_3^-) from runoff. However there is great variation in performance from different facilities, and not all facilities achieve TN and NO_3^- reductions (BMPDatabase.org, 2012). Davis et al. (2009) summarized that while redox reactions are taking place within bioretention, denitrification is not readily occurring, allowing the harmful release of highly soluble NO_3^- to receiving waters. Many hypothesize that designing an anaerobic zone within a bioretention facility is necessary to facilitate denitrification (Brown & Hunt, 2011; Dietz & Clausen, 2006; Hsieh et al., 2007; Hunt et al., 2006; Kim et al., 2003; Lucas & Greenway, 2010). However, Lucas and Greenway (2008) demonstrated that ecologically mature facilities can remove NO_3^- without such zones.

There are many potential N treatment pathways that facilitate stormwater NO_3^-, NH_4^+, and organic N (ON) retention in bioretention facilities including:

- Abiotic sorption of positively charged ON and NH_4^+ to negatively charged soil media
- Abiotic sorption of negatively charged NO_3^- to positively charged organic material
- Biological immobilization, mineralization, and denitrification by microbes
- Plant uptake facilitated by microbial mineralization
- Direct uptake by plants

Column studies replicating bioretention cells without vegetation have shown that NH_4^+ and positively charged ON sorb to media surfaces and negatively charged N ions sorb to positively charged organic material during storm events (Hsieh & Davis, 2005; Hsieh et al., 2007). In natural ecosystems, soil microbes are known to immobilize all forms of N, mineralize ON, and trade NO_3^- and NH_4^+ to plants in exchange for carbon resources between storm events (Gianinazzi-Pearson, 1996; Jackson, Schimel, & Firestone, 1989; Schimel & Bennett, 2004; Tao & Zhiwei, 2005; Whiteside et al., 2009). Also, microbial populations in the soil media can drive denitrification of accumulated NO_3^- pools

(Christensen, Simkins, & Tiedje, 1990; Christensen & Tiedje, 1990; Parkin, 1987). The vegetation may also directly assimilate stormwater NO_3^-, NH_4^+, and ON (Jackson et al., 1989; Paungfoo-Lonhienne et al., 2008; Schimel & Bennett, 2004; Vitousek et al., 1997; Whiteside et al., 2012).

Until now, studies investigating the processes that treat N from stormwater in bioretention gardens have focused largely on inorganic media properties and denitrification driven by extended hydraulic retention times (Brown & Hunt, 2011; Dietz & Clausen, 2005; Hsieh et al., 2007; Hunt et al., 2006; Lucas & Greenway, 2010). While vegetation is often acknowledged as an important component to bioretention (Davis et al., 2009), biological contributions to the processes that treat N have been largely ignored. Bratieres et al. (2008) conducted extensive column studies that measured different nutrient retention of wetland species in a bench-scale test that showed *Carex appressa* (a rush) and *Melaleuca ericifolia* (a native Australian shrub) significantly reduced effluent N mass. However, the study focused on effluent water concentrations and failed to demonstrate the mechanisms driving this reduction, such as uptake into the plants. If mechanisms driving N retention in bioretention systems can be better understood, then engineers should be able to design bioretention gardens to more consistently retain N in all climates.

The interactions that remove nitrogen from stormwater in bioretention are biogeochemical processes, or chemical reactions in which a source material undergoes a biologically driven transformation within a mineral substrate. In the case of a bioretention garden, inorganic N from stormwater runoff is the source material that is biogeochemically transformed into plant biomass that grows in a bioretention garden, denitrified N gasses, and the dissolved N (NO_3^- and NH_4^+) in the water that leaves a garden. The idea of biogeochemical hotspots where accelerated microbial activity increases nutrient cycling rates within soils was first introduced by Parkin (1987) and described in detail by McClain et al. (2003). Bioretention systems can function as engineered biogeochemical hotspots, where vegetation and their associated microbial communities rapidly transform N inputs to less transient N forms, reducing N transport from the urban landscape to adjacent receiving waters.

Stable isotopes can be applied as a tool to trace elements of interest through biogeochemical processes to better understand how sources, pathways, and products interact (Dawson, Mambelli, Plamboeck, Templer, & Tu, 2002; Hultine et al., 2008; Mariotti et al., 1981; Nadelhoffer et al., 1999). By manipulating the isotopic ratio of a source, the isotopic ratios of the resultant products and remaining sources can be measured and pathways can be described. The N cycle is highly complex and can include multiple biogeochemical pathways between sources and sinks, not all of which are well understood (Evans, 2001; McClain et al., 2003). By following an enriched source material through a defined system, the ultimate fate of this source and intermediary processes throughout the system can be studied.

Based on the previous work by civil engineers and ecologists described above, I predict that if vegetation contributes directly to the retention of N from urban stormwater, then it should be possible to add enriched ^{15}N as a component of a synthetic stormwater mix during a simulated runoff event and recover this label in the plant tissue growing in the test bioretention cell. We also predicted that the nitrogen present in the effluent of bioretention systems is the product of nitrification of N sources resident to the garden, and not from influent stormwater from the same flood event. To test this, we applied a 99.8 atom% ^{15}N NH_4NO_3 label to three different bioretention treatment designs and attempted to recover the label 1) in the vegetation growing in each treatment cell, and 2) in the effluent of each cell. First, if the ^{15}N label can be recovered in the new growth of the vegetation in each cell, then the plants in each cell are directly contributing to N treatment by assimilating the influent stormwater N into biomass. Second, if the ^{15}N label can be recovered from the bioretention effluent, then the relative contribution of influent stormwater N to effluent N can be compared.

4.2 Methods

A synthetic stormwater blend enriched with ^{15}N was added to three bioretention gardens of different plant communities in May 2013 in order to trace an N pulse through a bioretention system. New plant growth and the effluent water was sampled and analyzed for $\delta^{15}N$ before and after the addition of the ^{15}N-enriched stormwater to demonstrate the uptake by plants and the presence of the label in the effluent.

4.2.1 Site description

Three bioretention cells of different vegetation communities were established in 2010 at the Green Infrastructure Research Facility on the University of Utah campus in Salt Lake City, UT. The climate at the site is characterized by snowy winters, cool and rainy springs, and extended dry periods with no precipitation throughout hot, dry summers. Each cell was 2.5 m x 4 m with a depth of 1.2 m, consisting of two media layers: a 0.6 m topsoil layer (63% sand, 23% silt, and 14% clay) above a 0.6 m deep expanded shale media layer (average particle diameter was 1 cm). The cells were lined to facilitate the collection and sampling of the water after it percolated through the system. The cells consisted of either 1) a wetland community commonly used in bioretention in mesic climates, 2) an upland native community as prescribed by Houdeshel et al. (2012) that did not require irrigation in semi-arid climates, and 3) a media treatment without vegetation. These treatments will be referred to herein as: the “Wetland” cell; the “Upland” cell; and the no-vegetation “Control” cell. The vegetation types found in the Wetland and Upland cells are listed in Table 4.1. Because Lucas and Greenway (2008) suggest that N treatment may be a function of ecological community establishment, each site was constructed in the fall of 2010 and allowed to establish for 22 months before testing began.

<u>4.2.2 Application of ^{15}N label</u>

The ^{15}N labeling experiment was conducted as part of the year-long nutrient treatment experiment described in Chapter 3. Starting January 1012, runoff events were simulated with synthetic stormwater to mimic an average precipitation year for Salt Lake City, UT Stormwater was synthesized by mixing seasonally adjusted volumes of tap water with soil mulch as a nutrient source in 2,000 liter tanks so that, on average, the influent stormwater contained 1.64 mg/l total N, 0.24 mg/l, and NH_4, 0.62 mg/l NO_3. After mixing, the synthesized stormwater was added to each cell following a Natural Resources Conservation Service Type II design storm pattern. On May 7, 2012, 3 g of 99.8 atom% ^{15}N NH_4NO_3 (Icon Isotopes, Saugerties, NY) were used as the N source instead of the soil mulch, resulting in a final concentration of 0.25 mg/l NH_4^+ and 0.85

mg/l NO_3^-. The enriched ^{15}N label was added only one time in order to estimate the assimilation rate of stormwater N into new plant growth and the residence time of the label in each cell. Plant and effluent water samples were collected and analyzed relative abundance of ^{15}N to demonstrate the presence of the ^{15}N label in each respective product. Isotopic N measurements (δN) are reported here in δN notation with units of ‰ (per mil) as:

$$\delta^{15}N = (R_{sample}/R_{standard} - 1) \times 1000‰ \quad (1)$$

where R is the ratio of ^{15}N to ^{14}N of the sample and standard, respectively. Atmospheric N_2 was used as the standard for δN, so $R_{standard} = 0.00368$ (Dawson et al., 2002; Evans, 2001; Mariotti et al., 1981). Ecological processes associated with nitrification and denitrification can enrich ^{15}N as much as 20‰ to 30‰ (Evans, 2001; Mariotti et al., 1981). To ensure positive identification of the label in the plant tissue and in the effluent over a 9-month study period, the applied dose of $^{15}N - NH_4NO_3$ was calculated to enrich the soil N pool of each cell to δN = 100‰. Each cell contained 7,500 kg of soil, with a measured soil N content of 0.1%, yielding approximately 7,500 g of N in each cell. The background $\delta^{15}N$ for each garden was measured at 7.4‰.

4.2.3 Label recovery in plant tissue

New growth from the vegetation in the Upland and Wetland cells was collected and analyzed for $\delta^{15}N$. New growth was harvested from each plant in the Upland cell on May 3, 2012, 3 days before the addition of the ^{15}N label, and weekly after the addition of the ^{15}N label for 4 weeks. New growth was also sampled and analyzed on August 15. New leaves were easily identifiable at the tips of all shrubs, and soft, green blades were identifiable on all bunchgrasses. The bottom 5 cm of each young grass blade was trimmed and sampled because grass blades grow from the base with the oldest material at the blade tip. Individuals of each species in the Wetland cell were randomly selected and marked. New leaf material was sampled from the marked individuals 3 days before the addition of the ^{15}N label and weekly after the addition of the label for 5 weeks. *Phragmities spp.* an *M. sativa*, were not sampled before the addition of the label because they did not emerge until after the label was introduced. One *Phragmities spp.* emerged on May 28, 20 days after the addition of the label. New leaves were sampled from this individual on May 28 and June 5. On June 25, *M. sativa* was found growing prolifically in the Wetland cell as a weed and allowed to propagate. On June 25, five leaves from four *M. sativa* individuals were combined into two compound samples such that each 10-leaf sample represented two individuals.

The harvested plant tissue from each individual was dried for 72 hours at 90°C and ground. One mg of each tissue sample was analyzed for N isotopic ratios on a Carlo Erba elemental analyzer (Model 1108, Milano, Italy) coupled with a Finnigan MAT delta S isotope ratio mass spectrometer (San Jose, CA, USA) at the Stable Isotope Ratio Facility for Environmental Research (SIRFER) at the University of Utah. Each sample analyzed represented the $\delta^{15}N$ value of the new growth from one individual plant. Samples from different individuals of the same species were then averaged together and reported as the $\delta^{15}N$ value for each species.

4.2.4 Label recovery from bioretention effluent

Effluent $\delta^{15}N$ was evaluated by a three-step process. First, composite effluent samples were collected from each garden during each storm event. Second, heirloom yellow pear tomato plants were grown hydroponically in the effluent composite samples. Last, the leaf tissue from the tomato plants was dried, ground, and analyzed for $\delta^{15}N$. Effluent $\delta^{15}N$ was analyzed in this way from the May 7 (the date the label was added), May 12, May 16, May 21, May 26, June 5, October 15, and November 15 storm simulations.

The effluent was collected from a 3 cm (1.25") pipe that drained the bottom of each cell. A valve was used to regulate outflow so that the gravel storage layer drained completely in 24 h. The collection pipe was routed to a custom-made 3-liter tipping bucket flow gauge. Tips were alternatively directed to a 1,000 liter storage tank, creating a composite sample of each effluent event. The effluent from each treatment from the May 21, October 15, and November 15 storm simulation was analyzed for Inorganic N (IN) mass. The effluent N concentrations from the May 21 storm were assumed to be representative for all storm simulations from May 7 to June 5.

To collect effluent for IN analysis, 24 discrete 250 ml effluent samples (E) were drawn from the effluent in 40 liter increments using an ISCO auto-sampler (Teledyne-ISCO, Lincoln, NE) connected to the tipping bucket flow gauge by a PRX 11500 reed switch (HIS sensing, Chickasha, OK). Each sample was analyzed for NH_4^+-N, NO_2^--N, and NO_3^--N concentrations. NH_4^+-N was tested using the TNT™ 830 ULR NH_4 method that is EPA-certified to be accurate between 0.015 and 2.0 mg/l (Hach™ Company, Loveland, CO). The method utilizes a colorimetric analysis that tests NH_4 on a Hach

6500 spectrophotometer. NO_3^- -N and NO_2^- -N were analyzed on a Metrohm 881 Compact Ion Chromatographer (Metrohm, USA, Riverview, FL) capable of measuring +/- 0.002 mg/l for the ions tested. The total effluent mass of each N species for each storm simulation was calculated by:

$$\text{Total effluent mass (mg)} = \sum E\left(\frac{mg}{liter}\right) x\ 40(liters)\ . \tag{2}$$

Total effluent mass for NO_3^--N, NO_2^--N and NH_4^+-N were then summed and are reported collectively as the total effluent mass of IN (E_m).

Effluent $\delta^{15}N$ was indirectly measured by analyzing leaves from tomato plants grown hydroponically in the bioretention effluent. Four trays of six heirloom yellow pear tomato seedlings (*Solanum lycopersicum)* were grown hydroponically in a 40-liter sub-sample from each composite sample from each storm event (Figure 4.1). Preliminary trials showed that the cotyledon leaves, or the first pair of leaves that emerge from the seed, had the same isotopic signature as the seeds so these leaves were not analyzed. All noncotyledon leaves from one tray were combined into one sample. The samples were dried and ground, then 0.3 mg of the ground samples were analyzed for $\delta^{15}N$ at SIRFER using a Finnigan Delta Plus Advantage mass spectrometer coupled with a duel inlet GC/CP interface (San Jose, CA). When operating in the range of natural $\delta^{15}N$ abundance, the instrument error is less than 0.1 ‰. However, when processing samples that are expected to be enriched as much as 100‰, the error was 1.4‰.

Two compound samples made up of the tomato leaves from one tray were successfully analyzed from each trial for $\delta^{15}N$ and averaged. The tomatoes utilized the dissolved N from the effluent to construct the noncotyledon leaves. It was assumed that the only N source available for the tomato plants to incorporate into leaf material is the dissolved N from the effluent. Therefore, the average $\delta^{15}N$ of the two compound samples is reported as a representative proxy for the $\delta^{15}N$ of the effluent IN (E_δ) after Evans et al. (1996).

4.3 Results

4.3.1 δN recovery from plant tissue

The added ^{15}N label was readily identified in new leaf tissue in both the Upland and the Wetland cells in all species sampled except *M. sativa*. Average $\delta^{15}N$ values for vegetation in the Wetland cell were 2 to 15 times greater than in the Upland cell, indicating that the wetland vegetation assimilated a greater proportion of the influent ^{15}N than the vegetation in the Upland treatment. The highest δN value measured in the Upland cell was 3,810‰ in *S. scoparium*; the highest δN for *C.ledifolious, B. gracilis*, and *A. utahensis* were all between 1,000‰ and 1500‰ (Figure 4.2). These δN values were from new growth sampled on May 21, 14 days after the addition of the label. All data are reported in Appendix C. The two highest δN values measured in the Wetland cell were from the only *Phragmites spp.* individual in the cell during the observation period: δN = 6,300‰ from new growth sampled on June 5, 29 days after the label was added and $\delta^{15}N$ was 5,860‰ in new growth sampled May 28, 21 days after the label was added (Figure 4.3). The second highest δN values of the Wetland species were samples from two clusters of *J. effuses* that appeared to be separate individuals: δN was 4,090‰ and

3,390‰ for the two individuals sampled June 5, 29 days after the label was added, and δN was 3,340‰ and for the two individuals sampled May 15, 8 days after the label was added (Figure 4.3). Only *M. sativa* did not obviously assimilate the ^{15}N label from the storm simulation on May 7 into leaf biomass. δ^{15}N in the two samples representing four *M. sativa* individuals was 8.8‰ and 6.2‰, which was within the δ^{15}N range of the other vegetation tested in the Wetland cell before the label was added.

4.3.2 Storm simulation effluent

The added ^{15}N label was identified in the effluent collected from all three treatments for all eight simulated storm events sampled. The effluent from all three cells was most enriched for the May 7 storm simulation in which the label was added. For this event, the tomato leaves grown in the Upland effluent were most enriched (δ^{15}N = 31,000‰), followed by the tomato leaves grown in the control effluent (δ^{15}N = 19,000‰), with the tomato leaves grown in the effluent from the Wetland showing the smallest enrichment (δ^{15}N = 16,000‰). The total effluent IN mass for the Control cell over the study period (2080 mg IN) was double that of the Upland cell (1107 mg IN), so while the effluent from the Upland cell was more enriched, more of the ^{15}N label washed through the Control cell than the Upland treatment during the storm event in which it was added. Effluent IN mass from the Wetland cell (330 mg) was very low, indicating that almost the entire ^{15}N label that was added was retained within the Wetland treatment.

The reduction of ^{15}N in the effluent of all cells in the 1st month after the label was added fits a power model (r^2 = 0.953, 0.931, and 0.922; p = 3.1x10^{-5}, 0.0001, and 0.0002; ANOVA F = 124.1, 80.5, and 71.2 for the Control, Upland, and Wetland, respectively).

This suggests that if stormwater N is not washed out of the cell in the same storm it is introduced, then it will likely be retained for at least several months under normal hydroclimatological conditions (Figure 4.4). This is further supported by the low effluent $\delta^{15}N$ values ($\delta^{15}N < 200$) for all cells 5 and 6 months after the addition of the label (Table 4.2).

4.4 Discussion

4.4.1 δN recovery from plant tissue

The ^{15}N –enriched nitrogen source applied to the test bioretention gardens during a simulated runoff event was readily taken up by the vegetation in both the Upland and Control treatments. This result demonstrates that plants actively participate in the removal of inorganic N from stormwater in bioretention by assimilating the influent N into plant biomass. While other studies have demonstrated that the presence of vegetation in bioretention cells improves N removal (Bratieres et al., 2008; Lucas & Greenway, 2008, 2010), this is the first demonstration to the author's knowledge that the vegetation actually assimilates stormwater nitrogen in bioretention systems. While this study demonstrates that plants assimilated stormwater N, the pathways for this assimilation are unknown. It is likely that soil microbes are driving the uptake and immobilization of influent N, and that plants are trading carbon resources for N.

M. sativa did not assimilate any of the ^{15}N label into leaf biomass. *M. sativa* is capable of fixing atmospheric N through symbiotic relationships with bacteria (Vitousek et al., 2002). The δN values found in the *M. sativa* leaf tissue are similar to that of the other vegetation in the garden before the ^{15}N label was added. Therefore, *M. sativa* did not assimilate the ^{15}N-enriched NH_4NO_3 either because other organisms immobilized the labeled N before *M. sativa* became active or because *M. sativa* was fixing N from the atmosphere. In an environment without an enriched N pool, the observed $\delta^{15}N$ values of 6.2 and 8‰ would suggest that *M. sativa* may not be acquiring a significant amount of N through fixation. If *M. sativa* was fixing N from the atmosphere, $\delta^{15}N$ values would be expected to be closer to an atmospheric signature of $\delta^{15}N = 0$ (Vitousek et al., 2002). $\delta^{15}N$ values for other species were not collected on June 25. Therefore, it is unknown if the $\delta^{15}N$ values of *M. sativa* were lower than the other vegetation in the cell. δN values similar to the other vegetation would indicate that *M. sativa* was utilizing the same soil N pool as the other species, and that the entire label had been utilized or otherwise made unavailable to vegetation uptake 10 weeks after the label was added.

4.4.2 Nitrogen saturation

Bioretention systems are expected to receive and retain high N inputs from impervious surfaces when implemented in urban environments. When N deposition rates surpass utilization rates in natural ecosystems, N saturation can occur; after N saturation thresholds are surpassed in natural systems, the release rate of IN in streams increases at a disproportionately greater rate than N additions (Aber et al., 1995; Aber, Nadelhoffer, Steudler, & Melillo, 1989; Aber et al., 1998; Stoddard, 1994; Vitousek et al., 1997). For this reason, there is concern that N retention performance of bioretention may change dramatically if these facilities become N saturated.

As IN produced by mineralization and nitrification of resident N sources exceeds the needs of vegetation, the highly dissolvable IN (specifically NO_3^-) will leach out of the cell. Because there was no vegetation in the Control cell, N utilization was expected to be very low. The effluent IN from the Control cell was greater than influent TN, indicating that the system had surpassed the N saturation threshold. Even though more IN left the Control cell than was added during each storm simulation (Figure 4.5), the ^{15}N signature of the effluent measured by the tomato leaves indicates that the cell still retained much of the ^{15}N label. Even for the May 7 storm, $\delta^{15}N$ of the tomato leaves grown in the effluent was only 1/8 of the tomato leaves grown in the 99.8 atom ‰ ^{15}N influent solution. This indicates that much of the effluent IN from this storm simulation was mineralized and/or nitrified N that was resident in the cell before the storm simulation. The $\delta^{15}N$ of the tomato plants grown in the effluent from the Upland cell was greater than the Control cell. However, the mass of IN released by the Upland cell compared to the Control cell was much less. This indicates only a small contribution of the ^{15}N NH_4NO_3 label added on May 7 to the effluent IN from the Upland cell during this period. This indicates some leaching of residual NO_3, meaning that the upland cell may be approaching its saturation threshold. On the other hand, given the high retention of the isotopically labeled N and the low effluent N concentrations, it appears that the Wetland cell did not approach its N saturation threshold during this study.

4.5 Recommendations

In this study, we focused on testing effluent N concentrations to determine the sustainability of N treatment by bioretention facilities. Our results suggest that a multi-year study period that records monthly effluent N concentrations and soil NO_3^- is needed to determine the long-term sustainability of N retention by stormwater treatment systems represented by the cells tested here. It may be preferable to focus on trends in soil and vegetation N because effluent NO_3^- export is not likely a good indication of N accumulating in bioretention facilities as NO_3^- leaching is not expected to increase gradually. NO_3^- export from ecosystems is expected to increase drastically in a step-wise fashion after the N saturation threshold is exceeded (Aber et al., 1995; Aber et al., 1989; Aber et al., 1998; Stoddard, 1994; Vitousek et al., 1997).

This study demonstrated that vegetation in bioretention assimilated stormwater N into biomass. This study was not able to address mechanisms of uptake by the plants, and particularly, the role of soil microbes in N removal from stormwater. A number of ecosystem science techniques could be employed to further understand the N cycle within the tested bioretention treatments, and would be valuable to urban ecologists and stormwater managers to predict the long-term performance of these engineered ecosystems in urban settings. However, the results from this study indicate that in order to best inform the stormwater management community how to improve bioretention N removal, future research should focus on the relationship between different plant species and N retention and the relationship of different vegetation density and N retention. Extensive analysis of below-ground plant biomass, soil, and soil microbes would be required to answer this question. However, describing the nutrient treatment pathways in detail may not provide any practical information from a design and implementation standpoint. Because we have demonstrated that the influent N is assimilated into vegetation biomass, it may be more practical to correlate plant productivity, vegetation biomass and planting densities with nitrogen removal.

Our experimental design does not allow the quantification of the mass of the ^{15N} label that was taken up by each plant or by each species. Because of translocation of N within plants, the entire plant must be harvested and analyzed. However, qualitatively, it does appear that some species such as *Phragmites spp.* and *J. effuses* may assimilate stormwater N more aggressively than other species. Because the ^{15}N label was recovered in all species except *M. sativa*, *Phragmites spp.* and *J. effuses* cannot be described as out-competing the other species for N resources, but because bioretention systems are exposed to very high nitrogen inputs, studies that focus on maximum uptake may be more beneficial to stormwater managers than competition studies.

Another important question with respect to bioretention facilities is the role of denitrification in nitrogen removal from stormwater (Brown & Hunt, 2011; Lucas & Greenway, 2010). There is a risk that incomplete denitrification in bioretention systems may be resulting in the production of N_2O or NO_2 gasses, which are known to be powerful greenhouse gases. Ecosystem scientists have developed methods to capture soil respiration and measure the composition of N-containing gases to determine if denitrification is occurring in soils, and if so, the N gasses that are being produced (Stevens, Laughlin, Burns, Arah, & Hood, 1997). It should be possible to separate N_2 gas from other N-containing gasses and measure the δN of the N_2 gas being produced by these gardens separate from the N_2O and NO_2 gas being produced by these gardens (personal communication, SIRFER). Using an N source with a different δN signature than the soil N pool should allow researchers to determine if the stormwater N that is inundating bioretention facilities is being denitrified, and what species of N gasses are being released back to the atmosphere.

Two components that should be added to future ^{15}N labeling experiments aimed at determining the long-term sustainability of N treatment in bioretention are direct measurements of soil TN, NO_3^-, and δN; and direct measurements of the abundance and δN of gaseous N forms in soil respiration. Direct measurements of soil TN, NO_3^-, and δN over many years will directly demonstrate if N is accumulating in bioretention soils and whether different treatment designs risk approaching an N saturation threshold. If a ^{15}N label is applied to test cells, tracking the δN of the soils will provide insight to the total residence time of N in different bioretention designs. The label that was added in this study was likely stronger than necessary, and a much smaller ^{15}N mass would refine the analysis of ecological processes beyond identifying the terminal pools in which the labels were recovered. Likewise, if a label is not applied, the natural abundance of δN over time may provide insight into the types of biogeochemical processes that are driving the N cycle in each treatment.

The three cells tested here likely represent the three stages of N saturation risk. The Wetland cell is not likely accumulation N and is at little risk of becoming N saturated. The Upland cell is likely accumulating N and is likely to become N saturated at some point in the future. The Control cell is likely beyond the N saturation threshold and is exporting more N than is being introduced to the cell. Research of natural systems suggests that the exportation of IN from bioretention cells will not gradually change to represent the accumulation of N in the soils relative to saturation. However, if the N saturation threshold is crossed, the system could suddenly release much more IN than before the threshold was crossed. Testing soil samples is less expensive and less logistically challenging than monitoring the water quality of bioretention effluent. If a future study can determine the trajectory of N treatment by bioretention facilities based on trends in soil N, then stormwater managers may be able to economically monitor the risk of a bioretention facility becoming N saturated and releasing large amounts of IN to receiving waters.

4.6 Summary

By adding 3 g of 99.8 atom% 15N NH_4NO_3 label to the three different test bioretention treatments, we were able to demonstrate that:

- Plants in the Upland and Wetland cell assimilated stormwater N into leaf biomass.
- δN values were higher in the wetland vegetation than in the upland vegetation, indicating that wetland species are better at protecting receiving waters from stormwater N.
- The Wetland cell appeared to be N limited, the Upland cell appears to be shifting towards N saturation, and the Control cell appears to be beyond the N saturation threshold.
- Some of the retained label may have been denitrified and lost to the atmosphere. Future studies should include methods to try to quantify this flux to help determine the long-term sustainability of different bioretention designs for different climates.
- Export of the added label reduced nonlinearly in all cells, indicating that if N can be retained for the storm event in which it enters, it will likely not be washed out of the facility.

From this study, it appears that wetland ecosystems used to treat stormwater runoff will not become N saturated. However, the design recommended by Houdeshel et al. (2012) may become N saturated over time. More research into adapting this design to investigate if incorporating greater plant densities and a greater variety of species that emerge in early spring can reduce the risk of becoming N saturated. *J. effuses and D. glomerata* are not obligate wetland species and may be able to survive in bioretention facilities in semiarid climates without irrigation. Because there are no plants in the media-only Control cell, N demands are low and the stormwater N additions appeared to saturate the cell's N retention capacity.

4.7 Acknowledgments

This research was conducted under the support of the National Science Foundation under Award Number CBET-1133524 and under the iUTAH EPSCoR Award Number EPS-1209732, and the University of Utah Sustainable Campus Initiative Fund. I thank Austin Orr, Kristianne Sandoval, and Steve Arhart, for their contributions to running synthetic storms in the winter, data collection, monitoring, and site development. We also thank Global Change and Ecosystems Center for continued encouragement of interdisciplinary water resources research at the University of Utah.

Table 4.1. List of prominent species found in the Upland and Wetland treatments. Alfalfa was naturally recruited into the garden as a weed, and is not recommended for use in bioretention. However, because it is a known nitrogen fixer, its presence is important and reported.

Upland Garden			
Species name	**Common Name**	**Form**	**# of individuals sampled**
Schizachyrium scoparium	Little Blue Stem	Bunchgrass	3
Bouteloua gracilis	Buffalo Grass	Bunchgrass	3
Sorghastrum nutans	Indiangrass	Bunchgrass	2
Amelanchier utahensis	Utah Serviceberry	Shrub	2
Cercocarpus ledifolius	Curl-leaf mahogany	Shrub, Evergreen	3
Artemisia cana	Silver sage	Shrub	1
Wetland Garden			
Species name	**Common Name**	**Form**	**# of individuals sampled**
Juncus effuses	Common rush	Rush	2
Dactylis glomerata	Oarchardgrass	Bunchgrass	5
Typha sp.	Cattail	Bunchgrass	4
Phragmities sp.	Phragmites	Bunchgrass	1
Salix exigua	Coyote willow	Shrub, Tree	5
Medicago sativa	Alfalfa	Forb	2

Figure 4.1. Yellow pear tomato seedlings were grown to the size shown in the picture, and then all noncotyledon leaves were analyzed for δN. Forty liters of effluent were recirculated constantly through the growth trays by 0.25 hp motors.

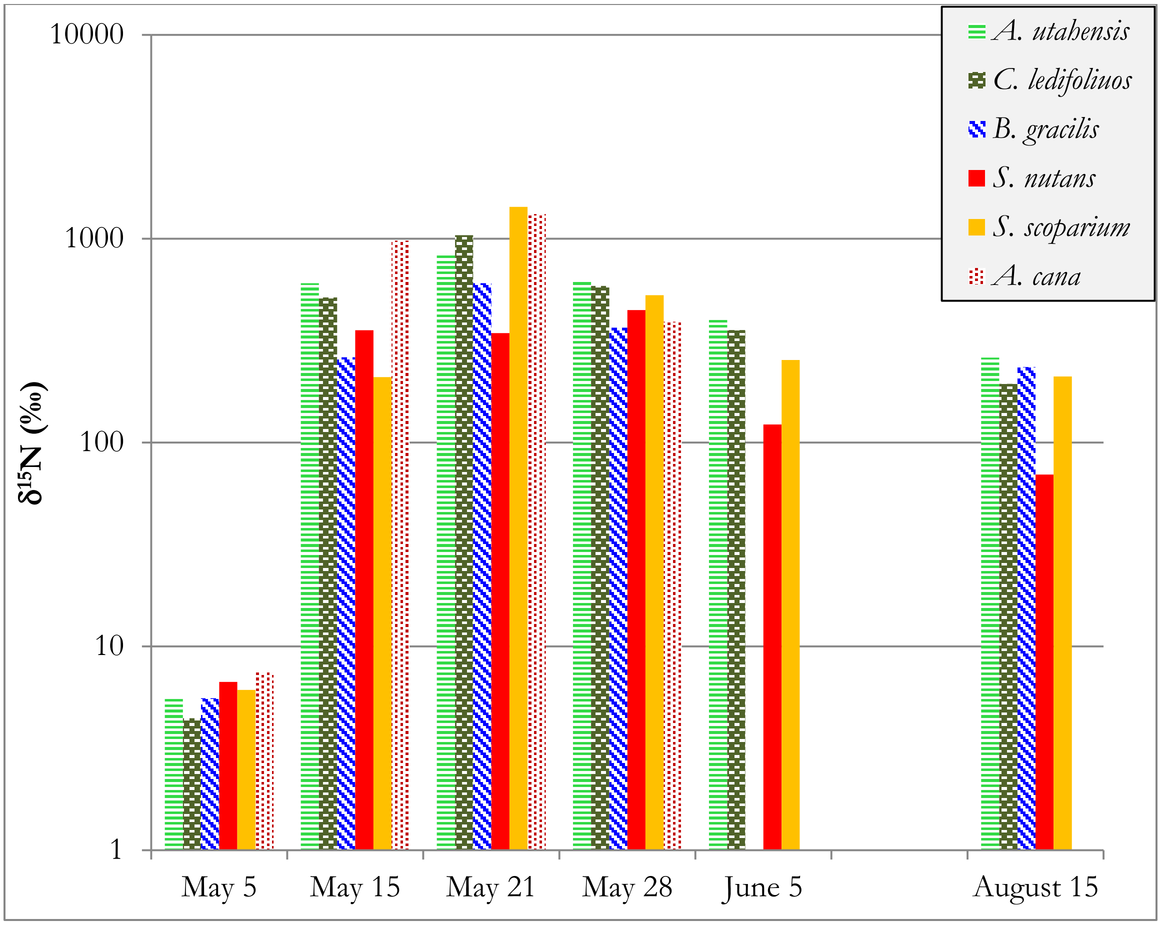

Figure 4.2. Average $\delta^{15}N$ values for the new growth sampled from vegetation in the Upland treatment.

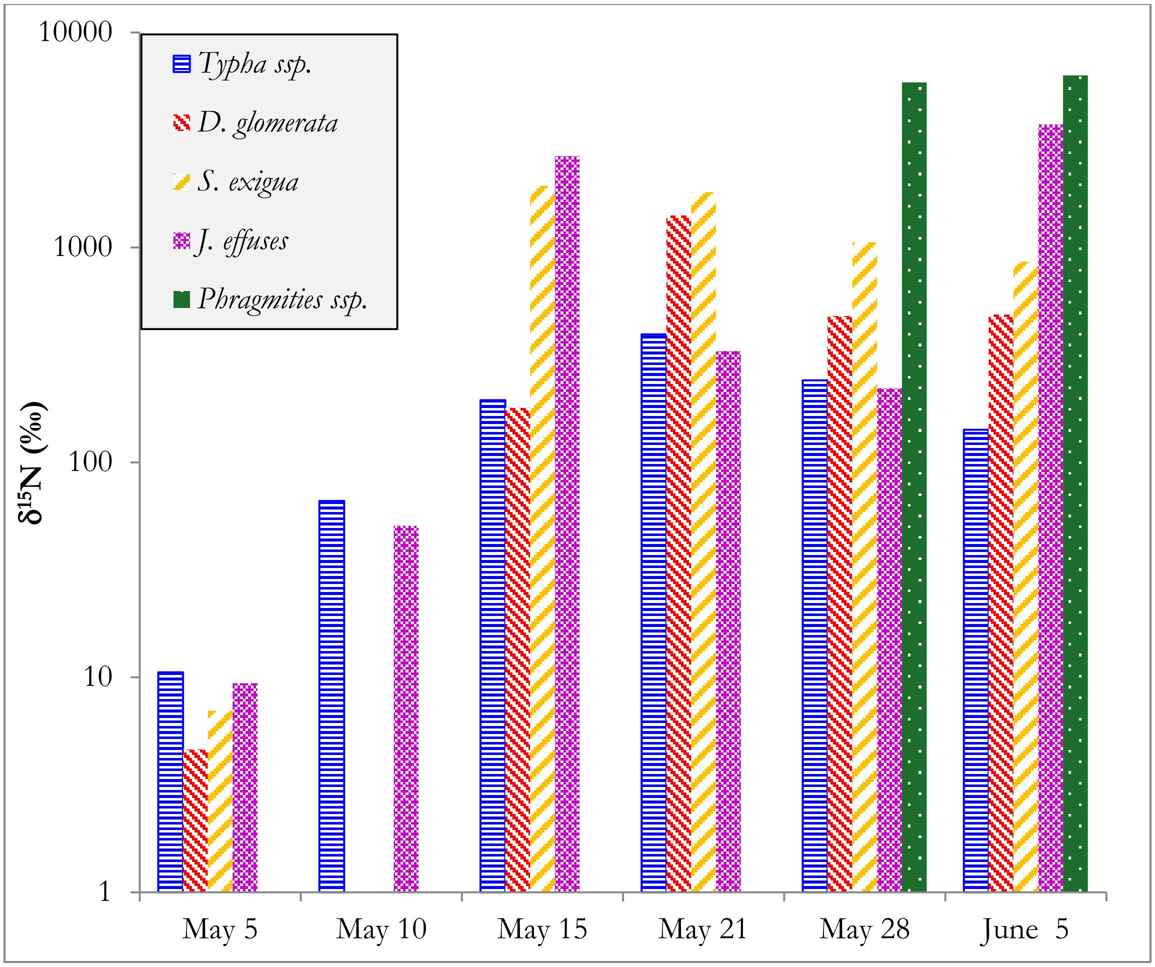

Figure 4.3. Average $\delta^{15}N$ values for the new growth sampled from different species in the Wetland treatment. Samples from the wetland treatement were not collected on August 15; no data were collected for *D. glomerata* or *S. exigua* on May 10.

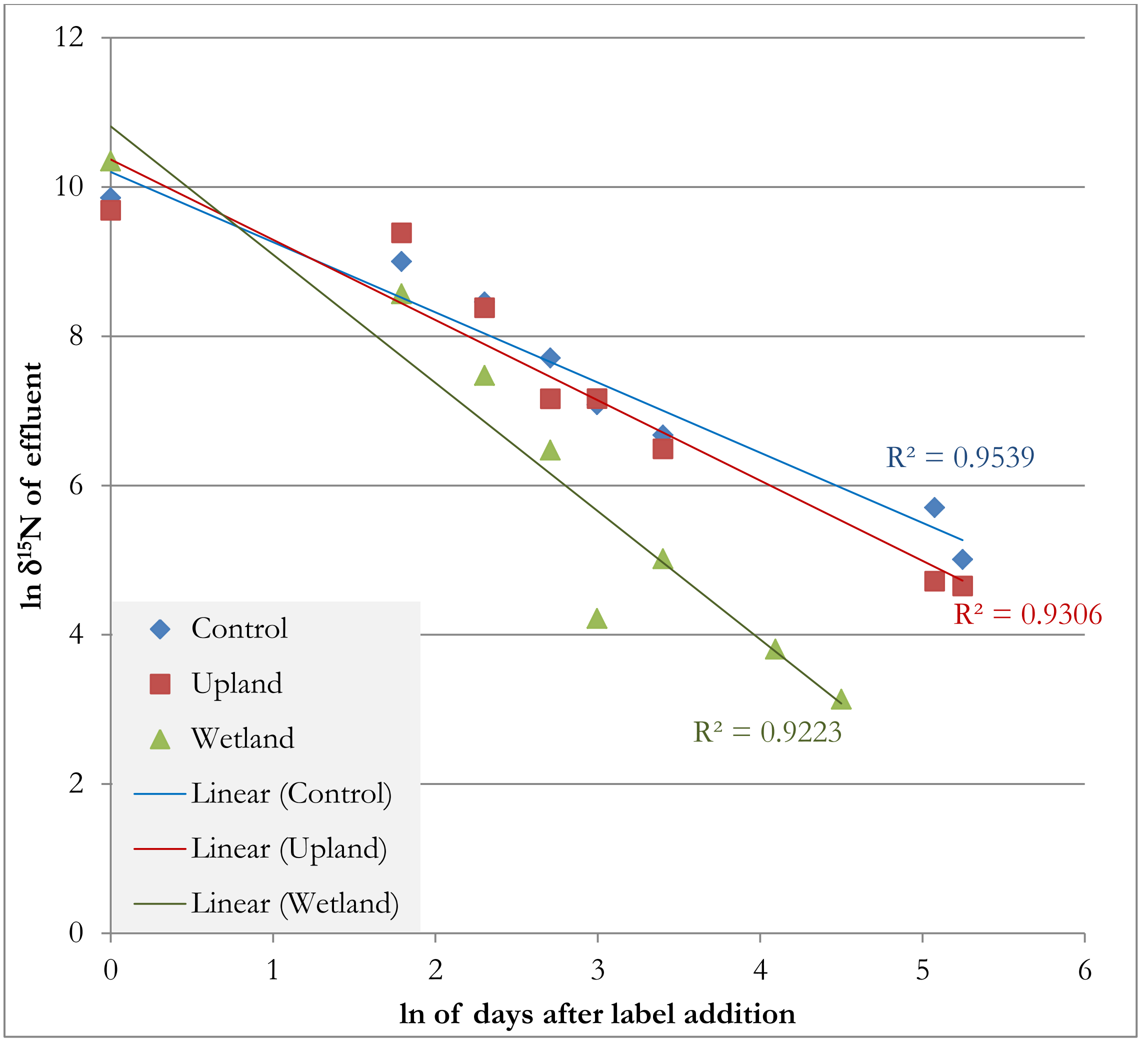

Figure 4.4. Power function of the δN of tomato leaves grown in the effluent from each cell for May 7, May 12, May 16, May 21, May 26, June 5, October 15, and November 15 storm simulations. Original data shown in Table 4. 2.

Table 4.2. $\delta^{15}N$ values from the tomato leaves grown in the effluent from each cell. The influent values reported represent the $\delta^{15}N$ values of tomato leaves grown in the 99.8 atom % ^{15}N, not the mathematically calculated 260,000‰ equivalent.

δN (‰) of tomato leaves grown in effluent

Cell	Influent	5/7	5/12	5/16	5/21	5/26	6/5	10/15	11/15
Control	162,500	19,064	8,122	4,717	2,229	1,192	721	300	150
Wetland	162,500	31,215	5,265	1,764	648	68	151	45	23
Upland	162,500	16,116	11,900	4,369	1,677	1,293	660	112	105

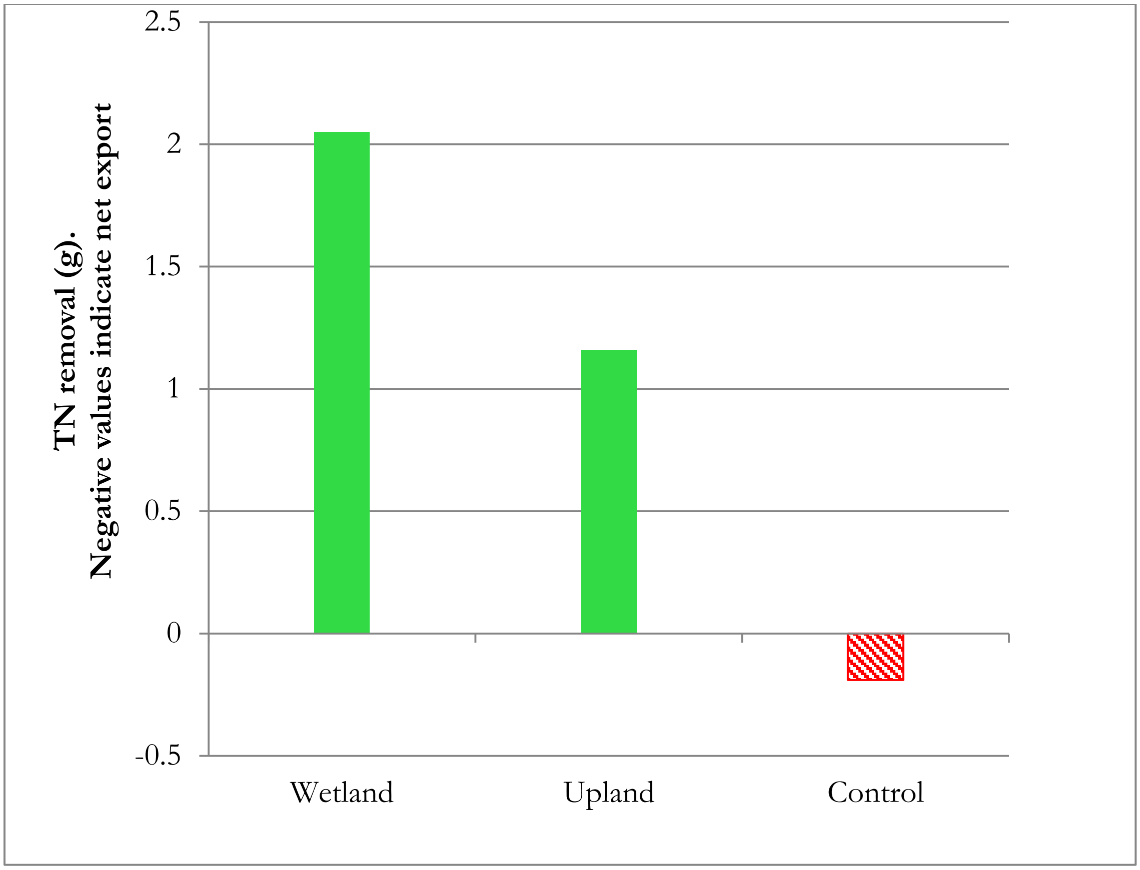

Figure 4.5. Total N retention for each treatment from May 7, 2012 to June 5 of the same year. Negative values for the Control cell indicate that more N was exported than added.

CHAPTER 5

CONCLUSIONS AND RECOMMENDATIONS

At the onset of the research described in this dissertation, there was a conspicuous lack of information available about the use of bioretention as a Low Impact Development stormwater management practice in xeric climates. The goal of the research described herein was to address this gap by applying the research tools and knowledge base developed by ecosystems scientists to design a bioretention system that mitigates the quality and quantity of stormwater runoff from urban landscapes in xeric climates. Based on the preliminary literature review, three hypotheses were developed.

- Hypothesis 1: Plants drive nitrogen removal in bioretention, even under the high nutrient loading conditions expected from urban landscapes to bioretention facilities.
- Hypothesis 2: Bioretention facilities without vegetation (media filters) may physically filter larger constituents from stormwater, but dissolvable nutrients such as NO_2^-, NO_3^-, and $NH4^+$ will wash through the facility untreated.
- Hypothesis 3: In bioretention cells with plants, influent stormwater nitrogen will be immobilize biologically or abiotically during a storm event, biologically mineralized to NO_3^-, then taken up by plants. These processes will reduce nutrient loading to receiving waters even when plants are dormant,

because roots and the associated microbial networks will store the influent N until conditions are favorable for above-ground growth.

To test these hypotheses, three research efforts were carried out. First, a review of stormwater engineering, rangeland restoration ecology, and plant physiology literature was synthesized into a design recommendation for bioretention in xeric climates. Second, bioretention test cells were constructed following these design recommendations, and the nutrient removal capacity of this design was tested over 1 year. Third, a ^{15}N isotopic label was added to the test cells as part of the 1-year nutrient removal study to test if plants assimilate stormwater N into biomass. The general conclusion of these efforts is that the tested bioretention design can sustainably mitigate urban stormwater runoff in xeric climates, but that nutrient treatment capacity may be improved with a better understanding of the behavior of xeric-adapted vegetation in these engineered systems.

Hypothesis 1 was supported by the three research efforts by demonstrating that effluent nutrient mass was greater in the test bioretention cell without vegetation, and that both upland and wetland vegetation assimilated stormwater nitrogen into biomass. Over the 1-year study of an unvegetated bioretention cell, a bioretention cell established with upland vegetation, and a bioretention cell established with wetland vegetation, the upland and wetland test cells both reduced total nitrogen loading to receiving waters, whereas the unvegetated cell leached more total nitrogen to the receiving waters than was added. The ^{15}N label added to each test garden was recovered in both upland and wetland vegetation, which demonstrated that both vegetation communities are assimilating the stormwater nitrogen introduced to these systems. While these studies support that vegetation is driving nitrogen removal in bioretention, longer duration studies are needed to determine if this behavior will continue over extended periods of high nitrogen loading expected from urban landscapes.

Hypothesis 2 was supported by the studies described in Chapters 4 and 5 because the nonvegetated cell tested did remove organic nitrogen over duration of the study, the nonvegetated cell did not remove NO_3^-, and because more of the applied ^{15}N label was recovered from the nonvegetated cell than the vegetated cells. Likewise, this cell demonstrated substantial net leaching of NO_3^- over the study, meaning that more NO_3^- was recovered in the effluent than was added as influent. However, only 50% of the ^{15}N label that was added to the garden was recovered as effluent over 8 months, indicating that half of the added label was "treated" in the garden. The soil media used in all three test cells included a portion of nitrogen-rich mulch. Hypothesis 2 may have been more strongly supported if a low-nitrogen soil media were used instead.

Hypothesis 3 was supported by the high retention of the applied ^{15}N label in the vegetated test cells. The cell established with wetland vegetation retained 93% of the applied label, and the cell established with upland vegetation retained 79% of the applied label. In the May 7 storm simulation event when the label was added, the ^{15}N label only accounted for 12% of the effluent NO_3^- from the cell established with wetland vegetation and 10% of the effluent NO_3^- from the cell established with upland vegetation. The ^{15}N label accounted for 20% of the effluent NO_3^- from the unvegetated cell for the same storm. This indicated that the effluent NO_3^- from the vegetated cells was dominated by mineralized nitrogen sources that were present in the garden before the addition of the label. The effluent NO_3^- concentrations from the upland vegetation community were strongly correlated with atmospheric temperature, such that effluent NO_3^- concentrations were lower when temperatures were hotter. While this does indicate that NO_3^- removal is better in summer when upland plants are active than in winter when plants are mostly dormant, effluent NO_3^- from the unvegetated cell were four times larger than effluent NO_3^- from the vegetated cells in the winter, supporting that the vegetation does play a role in nutrient removal even when plants are dormant.

5.2 Bioinfiltration design for xeric climates

The bioinfiltration design proposed in Chapter 4 differs from a standard bioretention design for mesic climates in three ways. First, the goal of this design is to engineer an ecosystem suitable for upland vegetation that is native to xeric climates instead of wetland species. The life history and physiology of the upland vegetation offers unique design advantages including seasonal dormancy, deep rooting structures, and the ability to regulate water loss through extended hot, dry periods. Second, runoff is routed to an unlined 0.6 m deep subgrade storage layer during a storm event instead of allowing water to pond on the surface of the facility. This is to protect the upland vegetation from inundation on the surface, which can cause anoxic soil conditions that the selected vegetation does not tolerate well. The storage reservoir also provides temporary storage during storm events to allow water to infiltrate to the native soils below, creating a pocket of deep soil water that the deep-rooted shrubs can access in dry periods. The depth of the storage layer was chosen to allow root growth of the selected vegetation to extend through the storage layer while still maximizing storage capacity for a given garden footprint. Third, the depth of the gravel storage layer is prescribed to be constant so that garden area can be scaled linearly with drainage area. This prescription makes the design process straightforward, and allows managers to predict the hydrologic benefits of the design without intensive site analysis.

One of the driving goals of LID is to “restore natural hydrology” of a developed site (U.S. EPA, 2012). In the arid west, this includes capturing stormwater runoff on site during a storm event to prevent physical degradation of receiving waters, reducing the nutrient loading from the urban landscape to receiving waters, and sustaining the plants in the system without supplemental irrigation. The literature review and hydrological modeling efforts described in Chapter 4 indicate that properly designed bioretention can achieve these objectives. The proposed design retains up to the 95th percentile storm of 20 mm (0.78 in) during a storm event. This water passes through the rooting zones of the established vegetation, then infiltrates into the soil below. Deep-rooting shrubs can then access this water through the regionally characteristic hot, dry months in summer, and the selected bunchgrasses grow prolifically in spring, scenes during the summer, and re-emerge with fall rains.

The 1-year nutrient removal study described in Chapter 5 demonstrated that the Upland treatment provided removal of TN, but released more NO_3 in the effluent than was introduced. There are two pathways by which vegetation can facilitate nitrogen removal in bioretention. First, vegetation can assimilate nitrogen into biomass. Second, plants can provide carbon resources to soil microbial communities that drive treatment processes such as denitrification that transform NO_3^- to N-containing gases (i.e., N_2, N_2O, or NO_2). The experiment descried in Chapter 6 clearly demonstrated that vegetation assimilates stormwater nitrogen into aboveground biomass. One of the conclusions reached in Chapter 5 was that soil microbial communities play a role in nutrient removal in bioretention, and that these microbes may be carbon-limited in the Upland treatment. Combining the findings of these two studies suggests that increasing the vegetation in the Upland treatment tested would 1) increase above-ground plant biomass and therefore vegetative N demand, and 2) provide more carbon resources to the soil microbial community. Both of these processes should result in greater N reduction in the Upland cell.

One of the goals of this research was to improve the original design for bioretention in xeric climates described in Chapter 4. As a result of this research, design recommendations in Chapter 4 should be modified to recommend that vegetation adapted to xeric climates be planted at three times the densities found in a natural system, and that a wider palate of vegetation may be employed to further improve nutrient retention. The original design will supply 20 times more water to a bioretention garden than a natural plot of vegetation would receive. The results from Chapter 5 suggest that increased vegetation densities would likely improve nutrient removal. Further, the results from Chapter 6 indicate that some species may assimilate more stormwater N than those selected. For example, *Juncus effuses* assimilated very high δN values following the addition of the ^{15}N label. This species is typically associated with wetlands, but is not an obligate wetland species. There is likely enough water delivered to these systems during rainfall events to sustain *J. effuses* for extended periods during the year, and the life history of the species suggests the above-ground vegetation may go dormant when soil moisture dries below optimal levels.

Different plant species have now been observed in the bioretention cells tested at the Green Infrastructure Research Facility (GIRF) and other bioretention gardens on the

University of Utah campus. From these observations, the plants recommended for use in bioretention in xeric climates listed in Table 4.1 can be strengthened. It appears very important to mix bunch grasses with shrubs to optimize the ecological function of bioretention facilities. The following shrubs are highly recommended for use in bioretention in cold desert climates: *Artemisa cana* (silver sage), *Chrysothamnus nauseosus* (rubber rabbitbrush), and *Rhus aromatic* (fragrant sumac). These shrubs can tolerate temporary inundations and saturated soil conditions, can regulate water loss during dry periods, and are commercially available. From the research described herein, the bunchgrasses recommended for use in bioretention in cold deserts are: *Schizachyrium scoparium* (little bluestem), *Sorghastrum nutans* (Indiangrass), and *Pascopyrum smithii* (western wheatgrass). *S. scoparium* is well established as an ornamental species, is readily available, and has grown well in all test cases. *S. nutans* is also readily available as an ornamental species, and has demonstrated the capacity to senesce midsummer, then vigorously regrow in the fall as precipitation returns. *P. smithii* emerges early in spring and has demonstrated the potential to spread and increase vegetative density as resources are available.

Bioretention systems are designed to require little maintenance. However, performance and aesthetics will likely be improved with pruning. Shrub pruning should be carried out every other year in late fall to minimize damage from snow accumulation, improve aesthetics, and remove N that has been assimilated over the previous years. Bunchgrasses should be trimmed back to 15 cm above ground once in the middle of summer and again in late fall. Trimming the standing dead plant material should improve regrowth the following fall and spring and remove some N from the facility.

5.2.2 Bioretention as part of the urban landscape

The performance of bioretention networks within the urban landscape must be evaluated before wide-spread implementation should be encouraged. The focus of this dissertation is to describe and quantify the unit processes that are occurring within bioretention gardens at the site scale. While there are still components of performance that need to be better understood, this body of research provides the foundational information about nutrient treatment and hydrology that is needed to start to model the effects of bioretention implementation at larger scales. At the neighborhood scale, the distribution patterns and site selection of bioinfiltration may play a larger role in watershed protection than single-unit performance. For example, investing in bioretention retrofits at the top of a watershed will likely have greater system-wide benefits than retrofitting the bottom of the watershed; likewise, regulations on new developments of land that was previously undeveloped will likely have greater impacts than regulations that require retrofitting existing infrastructure. Hydraulically, locating stormwater controls at the top of a watershed would reduce the frequency and intensity of runoff that is propagated through the entire stormwater infrastructure, reduce erosion along channels, and reduce the capacity of water existing infrastructure must accommodate. Geologically, infiltration rates are generally greater and ground water is generally much deeper along the benches of the Salt Lake Valley than on the valley floor. This is especially important during the early stages of bioretention implementation because environmental regulators recognize less risk associated with stormwater percolating 100 m to groundwater than they recognize in areas with shallow water tables (personal communication, Renee Zollinger, Salt Lake City Public Works).

Site selection may also play a large role in water quality protection of receiving waters. Different land uses may vary greatly over small distances. Landscape cover, use by pets, snow removal patterns, and traffic patterns may all dramatically affect stormwater quality generated from a site. Selecting bioretention sites to treat stormwater from areas that are likely to generate greater nutrient runoff could play a significant role in reducing down-stream nutrient loading.

5.2.3 Current barriers to implementation

One component of bioretention implementation is public and institutional perception of the technology. If the implementation of bioretention and other LID practices are going to be studied beyond the site scale, the scientific community must convince institution and utilities managers that installing these systems will benefit them. Because of this, public relations becomes a scientific problem. Careful consideration is required in how to overcome these human components of urban systems. To appropriately address these challenges, engineers and ecologists must collaborate with urban planners, social scientists, and architects to identify the most effective way of developing outreach to inform stakeholders about bioretention to gain public and institutional support.

5.3 Research needs

5.3.1 Nitrogen balance

There is a pressing need to describe the N budget within bioretention that includes N inputs, N outputs, and potential N storage pools. A proposed N budget for bioretention is shown in Equation 1 where stormwater N inputs (N_{in}) must equal N losses to denitrification (N_{dn}), loss through effluent (N_{eff}), assimilation by vegetation (N_{veg}), assimilation into microbial biomass (N_{mb}), and changes in bulk soil storage (N_{soil}).

$$N_{in} = N_{dn} + N_{eff} + N_{veg} + N_{mb} + N_{soil} \qquad \text{Equation I}$$

The research described in Chapter 5 contributed to closing this budget by quantifying N_{eff} for a single input event, and by verifying N_{veg} as a relevant N pool. The experiments described in Chapter 5 showed that the Wetland treatment released 7% of the ^{15}N label added as N_{eff} and the Upland treatment released 21% of the added ^{15}N label as N_{eff}, indicating that 93% and 79% of the added label was either retained as N_{veg}, N_{mb}, or N_{soil}, or lost to the atmosphere as N_{dn} from the Wetland and Upland treatments, respectively. Before it is possible to close the N budget within a bioretention cell, methods must be developed to quantify N_{veg}, N_{mb}, N_{soil}, and N_{dn}.

Removal of the above-ground vegetation for the analysis of total above-ground biomass, %N, and δN should allow future researchers to quantify the mass of an applied ^{15}N label that was assimilated by vegetation. Soils could also be sampled for total N content, NO_3^- content, δN, and the abundance of soil microbial biomass to estimate the mass of an applied label held in the soil matrix. These efforts conducted simultaneously with the quantification of N_{eff} quantifies all terms in Equation 1 except for N_{dn}, meaning that the loss of N from the system through denitrification could be calculated. In addition to this mass balance approach, methods should be developed to directly measure the rate of denitrification in these gardens. Ecosystem scientists have developed methods of detecting denitrification in soils, but these methods are equipment-intensive and highly sensitive to procedural error. However, it may be possible to capture some of the gaseous products of denitrification at the soil surface and identify an added ^{15}N label in these gas samples if stormwater N is indeed being denitrified in these facilities.

5.3.2 Vegetation selection

While the research herein demonstrated that the presence of vegetation improved N removal from stormwater and that all vegetation assimilated stormwater N into biomass, the experimental design was not intended to compare the performance of different species within each treatment. An experiment should be designed to test the N removal and assimilation performance of different species and different assemblies of species in order to prescribe vegetation that maximizes water quality benefits for use in bioretention. Bratieres et al. (2008) conducted this type of study on select wetland plants at a bench scale. However, most species selected for this study were native to Australia and provided little design assistance for engineers in the United States. The paradigm I adopted for my research was to test the performance of my bioretention design for xeric climates against a wetland community and a media-only cell with no vegetation. Moving forward, I would change this paradigm to emphasize replication of variations on the xeric design to improve our understanding of how the selected vegetation responds to the highly altered hydrology of these engineered ecosystems over longer periods. This new paradigm will allow greater understanding of bioretention performance in cold desert climates because observed differences in performance can be supported statistically, whereas the strength of my observations were limited by lack of replication.

APPENDIX A

INFLUENT NUTRIENT DATA

A.1 Influent Concentration notes

1) Organic Nitrogen (ON) was calculated by: ON = TN – (NH_4 + NO_2 + NO_3)
2) In June, July, and September, only one sample was tested and these values were used for all tests for each garden.
3) In October, TN was calculated and not measured: on average, IN was 41% of TN, so October TN was calculated by IN/0.41.
4) Although the same materials were used to make the influent stormwater mixture, the amount of NH_4 decreased substantially in March. This was not noticed until after April, when enough NH_4Cl was added to raise the NH_4 concentration to 5 mg/l NH_4
5) The same brand soil amender we had been using from January through November was no longer available, and so the closest replacement was used. No additional NH_4 was added to this mix, it was just richer in NH_4.

Influent mass in mg measured for each garden for each month.

January	NH_4	TN	NO_2	NO_3	PO_4	ON
Upland	637	5172	0	261	1004	4274
Wetland	784	6921	0	333	1109	5803
Control	666	5444	0	269	950	4508
February						
Upland	802	3463	0	280	519	1515
Wetland	784	6921	0	333	1109	5803
Control	654	3451	0	294	469	2070
March						
Wetland	35	1224	0	296	126	893
Upland	101	1615	0	312	170	1201
Control	69	2413	0	305	174	2039
April						
Wetland	37	1896	0	389	197	1470
Upland	52	2108	0	371	168	1685
Control	52	2050	0	364	236	1634
May						
Upland	1099	2784	42	433	174	1210
Wetland	1099	2784	42	433	174	1210
Control	1099	2784	42	433	174	1210
June	2	112	0	23	12	87
July	63	158	2	25	10	69
August						
Wetland	802	2310	0	459	453	1178
Upland	960	2651	33	473	569	1178
Control	816	2047	34	456	442	1178
September	1240	3424	42	611	735	1522
October						
Wetland	884	3288	0	940	203	1464
Upland	733	3288	0	990	246	4565
Control	761	3404	0	1001	210	1642
November						
Wetland	863	2269	0	918	198	1430
Upland	915	2406	0	973	210	1516
Control	889	2338	0	946	204	1473
December						
Wetland	1662	3739	0	831	198	1246
Upland	1586	4286	0	893	236	1806
Control	1674	4125	0	887	223	1564

Influent concentrations (mg/l) for each nutrient measured for each month.

January	**NH_4**	**TN**	**NO_2**	**NO_3**	**PO_4**	**ON**
Upland	0.5	4.7	0.0	0.2	0.7	3.9
Wetland	0.4	3.5	0.0	0.2	0.7	2.9
Control	0.4	3.7	0.0	0.2	0.6	3.0
February						
Upland	0.5	4.7	0.0	0.2	0.7	3.9
Wetland	0.5	2.3	0.0	0.2	0.4	1.0
Control	0.4	2.3	0.0	0.2	0.3	1.4
March						
Wetland	0.0	0.7	0.0	0.2	0.1	0.5
Upland	0.1	0.9	0.0	0.2	0.1	0.7
Control	0.0	1.4	0.0	0.2	0.1	1.1
April						
Wetland	0.0	1.0	0.0	0.2	0.1	0.8
Upland	0.0	1.1	0.0	0.2	0.1	0.9
Control	0.0	1.1	0.0	0.2	0.1	0.8
May						
Upland	0.5	1.4	0.0	0.2	0.1	0.6
Wetland	0.5	1.4	0.0	0.2	0.1	0.6
Control	0.5	1.4	0.0	0.2	0.1	0.6
June	0.0	1.0	0.0	0.2	0.1	0.8
July	0.5	1.4	0.0	0.2	0.1	0.6
August						
Wetland	0.5	1.5	0.0	0.3	0.3	0.8
Upland	0.6	1.7	0.0	0.3	0.4	0.8
Control	0.5	1.3	0.0	0.3	0.3	0.8
September	0.6	1.7	0.0	0.3	0.4	0.8
October						
Wetland	0.5	1.7	0.0	0.5	0.1	0.8
Upland	0.4	1.7	0.0	0.5	0.1	0.8
Control	0.4	1.8	0.0	0.5	0.1	0.8
November						
Wetland	0.5	1.3	0.0	0.5	0.1	0.8
Upland	0.5	1.2	0.0	0.5	0.1	0.8
Control	0.5	1.2	0.0	0.5	0.1	0.8
December						
Wetland	0.8	2.4	0.0	0.5	0.1	0.9
Upland	0.9	2.0	0.0	0.4	0.1	0.7
Control	0.8	2.1	0.0	0.4	0.1	0.8

Summary of influent concentrations (mg/l) for each nutrient measured

	NH_4	TN	NO_2	NO_3	PO_4	ON
Wetland Average	0.39	1.62	.007	0.27	0.22	0.96
Wetland St. Dev	0.24	0.74	.010	0.12	0.19	0.65
Upland Average	0.38	1.90	.005	0.28	0.26	1.29
Upland St. Deviation	0.22	1.40	.009	0.11	0.26	1.30
Control Average	0.37	1.65	.007	0.27	0.21	1.05
Control St. Deviation	0.22	0.77	.01	0.12	0.17	0.70

APPENDIX B

EFFLUENT NUTRIENT DATA

B.1 Notes on Effluent Data

Due to equipment failure, only one storm was administered as prescribed for June, and two July storms were administered after June 10. To account for this, the May influent and effluent data were multiplied by 6 storms instead of 5. No effluent was produced by the July storm volumes, so nutrient removal was 100%.

Effluent Event Mean Concentrations (mg/l) from each treatment for each month, January - July.

	NH4	TN	NO2	NO3	PO4	ON
January						
Wetland	0.12	2.42	0.02	0.19	0.30	2.08
Upland	0.03	2.33	0.07	1.51	0.12	0.71
Control	0.23	5.28	0.04	4.54	0.30	1.82
February						
Wetland	0.03	1.34	0.04	0.25	0.15	0.66
Upland	0.05	3.14	0.10	2.04	0.16	0.95
Control	0.23	7.97	0.05	6.44	0.22	1.25
March						
Wetland	0.03	0.94	0.00	0.29	0.24	0.62
Upland	0.07	1.74	0.02	0.86	0.12	0.79
Control	0.04	3.45	0.09	2.45	0.11	0.87
April						
Wetland	0.03	0.83	0.02	0.20	0.10	0.52
Upland	0.00	1.55	0.01	0.80	0.19	0.74
Control	0.00	1.81	0.03	1.28	0.14	0.50
May						
Wetland	0.05	0.45	0.02	0.13	0.09	0.20
Upland	0.02	1.39	0.01	0.93	0.19	0.39
Control	0.05	2.01	0.06	1.30	0.14	0.61
June						
July						

Effluent Event Mean Concentrations (mg/l) from each treatment for each month, August - December.

August						
Wetland	0.02	0.38	0.00	0.08	0.06	0.22
Upland	0.03	1.28	0.01	0.70	0.10	0.53
Control	0.10	2.10	0.02	1.50	0.08	0.40
September						
Wetland	0.02	1.40	0.00	0.82	0.09	0.56
Upland	0.03	1.29	0.02	0.77	0.09	0.47
Control	0.05	4.40	0.02	3.88	0.13	0.44
October						
Wetland	0.02	0.45	0.00	0.04	0.05	0.39
Upland	0.03	1.04	0.01	0.64	0.08	0.36
Control	0.03	4.40	0.01	3.56	0.07	0.80
November						
Wetland	0.05	0.43	0.00	0.06	0.05	0.33
Upland	0.05	2.33	0.00	1.31	0.09	0.97
Control	0.28	3.45	0.02	2.72	0.07	0.42
December						
Wetland	0.02	0.61	0.00	0.07	0.04	0.52
Upland	0.01	1.74	0.02	1.42	0.05	0.29
Control	0.15	3.21	0.04	2.82	0.06	0.20
Average						
Wetland	0.03	0.77	0.01	0.18	0.10	0.51
Upland	0.03	1.43	0.02	0.87	0.09	0.50
Control	0.10	3.15	0.03	2.50	0.11	0.63

Monthly mass removal (g) achieved by each treatment, January-July.

	NH4	TN	NO2	NO3	PO4	ON
January	# storms	5				
Wetland	3.03	16.68	-0.12	0.23	3.35	13.57
Upland	2.93	8.63	-0.54	-9.90	4.15	16.14
Control	1.60	-11.93	-0.31	-32.30	2.53	9.05
February	# storms	5				
Wetland	3.71	25.77	-0.28	0.05	4.55	24.64
Upland	3.76	0.08	-0.54	-9.81	1.72	2.34
Control	1.78	-34.10	-0.30	-40.02	0.95	2.26
March	# storms	5				
Wetland	0.04	1.76	-0.01	0.14	-0.49	1.60
Upland	0.15	-1.03	-0.12	-2.95	0.21	1.89
Control	0.06	-15.46	-0.71	-18.06	-0.02	3.26
April	# storms	6				
Wetland	0.00	4.24	-0.13	0.65	0.32	4.37
Upland	0.29	-0.81	-0.09	-4.72	-0.61	3.70
Control	0.31	-7.84	-0.28	-12.07	-0.16	4.21
May	# storms	6				
Wetland	5.09	10.26	0.05	1.07	0.15	4.40
Upland	5.40	5.82	0.18	-3.25	0.87	3.78
Control	5.12	-0.97	-0.21	-7.44	-0.13	1.56
June	0.01	0.34	0.00	0.07	0.03	0.26
July	0.13	0.32	0.00	0.05	0.02	0.14

Negative values indicate net leaching

Monthly mass removal (g) achieved by each treatment, Aug -Dec.

	NH4	TN	NO2	NO3	PO4	ON
August	# storms	2				
Wetland	1.56	3.85	0.00	0.75	0.78	1.91
Upland	1.85	2.51	0.05	-0.59	0.92	1.19
Control	1.30	-2.81	-0.07	-4.02	0.63	1.04
September	# storms	3				
Wetland	3.60	1.08	0.11	-3.53	1.62	0.87
Upland	3.55	1.78	-0.03	-3.24	1.64	1.47
Control	3.47	-10.19	0.03	-16.22	1.62	2.50
October	# storms	4				
Wetland	3.37	9.23	0.00	3.39	0.41	2.47
Upland	2.65	4.02	-0.06	-1.68	0.28	3.11
Control	2.81	-18.50	-0.06	-22.00	0.32	0.75
November	# storms	4				
Wetland	3.25	9.81	0.00	3.39	0.42	3.17
Upland	3.00	-7.56	0.00	-7.83	0.02	-2.74
Control	1.22	-16.56	-0.16	-19.96	0.19	2.34
December	# storms	5				
Wetland	7.76	16.40	-0.01	3.86	0.81	4.78
Upland	8.26	2.86	-0.21	-8.79	0.53	3.60
Control	7.10	-7.19	-0.31	-20.03	0.59	6.05
Total						
Wetland	31.53	99.74	-0.38	10.11	11.98	62.19
Upland	31.98	16.94	-1.35	-52.64	9.78	34.87
Control	24.91	-124.89	-2.38	-192.00	6.58	33.42

Negative values indicate net leaching

APPENDIX C

DATA FROM ^{15}N LABELING EXPERIMENT

δN (‰) values for plant material collected from the Upland treatment

date	5-May	15-May	21-May	28-May	5-Jun	17-Aug
# of days after label addition	-2	11	17	24	32	105
A. utahensis	6.56	651.5	503.6	613.7	310.0	200.5
A. utahensis	4.66	559.0	1198.1	X	488.6	321.7
Average	**5.61**	**605.24**	**850.85**	**613.66**	**399.30**	**261.08**
St. Error	**0.67**	**32.70**	**245.54**	**0.00**	**63.14**	**42.86**
Artimesia sp.	7.45	978.6	1321.7	391.8	X	X
B. gracilis	6.55	158.8	X	531.6	X	X
B. gracilis	4.68	129.1	1055.7	200.2	X	X
B. gracilis	5.47	498.0	155.5	X	X	234.6
Average	**5.57**	**261.97**	**605.61**	**365.88**	**NA**	**234.58**
St. Error	**0.44**	**96.63**	**259.87**	**95.67**	**NA**	**0.00**
C. ledifolius	5.16	X	654.9	448.7	299.8	132.9
C. ledifolius	4.17	561.9	X	484.3	330.2	X
C. ledifolius	4.00	469.3	1426.1	826.8	440.0	255.7
Average	**4.44**	**515.61**	**1040.50**	**586.60**	**356.67**	**194.28**
St. Error	**0.30**	**32.75**	**272.66**	**98.42**	**34.77**	**43.40**
S. nutans	6.32	61.8	271.3	645.8	148.3	81.4
S. nutans	7.07	649.7	418.0	247.7	97.2	57.9
Average	**6.70**	**355.77**	**344.65**	**446.71**	**122.75**	**69.67**
St. Error	**0.27**	**207.87**	**51.89**	**140.74**	**18.07**	**8.30**
S. scoparium	7.48	55.5	3809.5	881.5	298.8	210.9
S. scoparium	5.04	238.5	210.7	175.3	209.7	X
S. scoparium	5.83	335.1	279.7	X	X	X
Average	**6.12**	**209.67**	**1433.32**	**528.40**	**254.25**	**210.90**
St. Error	**0.59**	**66.95**	**970.22**	**249.69**	**31.50**	**0.00**

X indicate that no δN value is available for this individual on this date.

δN (‰)values for plant material collected from the Wetland treatment

Species	1-May	10-May	15-May	21-May	28-May	5-Jun	25-Jun
Typha sp.	10.6	91.3	22.4	188.5	479.2	125.6	X
Typha sp.	10.1	23.3	311.2	1186.8	297.0	254.9	X
Typha sp.	10.3	X	316.1	61.0	56.5	52.4	X
Typha sp.	11.2	84.0	129.9	140.6	130.9	134.3	X
Average	10.6	66.2	194.9	394.2	240.9	141.8	X
St. Error	0.2	17.6	62.4	229.9	81.4	36.3	X
D. glomerata	4.7	X	160.7	499.2	543.5	599.1	X
D. glomerata	6.6	X	X	X	327.1	217.1	X
D. glomerata	2.0	X	209.5	2781.6	654.0	342.4	X
D. glomerata	3.6	X	168.1	X	680.3	478.8	X
D. glomerata	6.3	X		942.1	41.5	528.0	X
Average	4.6	X	179.4	1407.7	479.8	487.1	X
St. Error	0.9	X	15.2	698.8	133.4	48.5	X
Phragmities sp.	X	X	X	X	5856.7	6314.9	X
J. effuses	9.3	X	3337.4	470.6	178.3	4085.8	X
J. effuses	9.5	50.7	1993.6	188.6	263.9	3390.0	X
Average	9.4	50.7	2665.5	329.6	221.1	3737.9	X
S. exigu	6.5	X	1767.8	1645.5	1028.3	832.0	X
S. exigu	7.3	X	1413.1	1021.4	920.1	680.6	X
S. exigu	7.7	X	4298.4	3126.6	2088.8	1607.1	X
S. exigu	6.3	X	1012.9	2335.0	466.1	311.9	X
S. exigu	7.2	X	1193.7	912.3	779.7	X	X
Average	7.0	X	1937.2	1808.2	1056.6	857.9	X
St. Error	0.3	X	603.6	416.2	274.8	272.6	X
M. Sativa	X	X	X	X	X	X	8.8
M. Sativa	X	X	X	X	X	X	6.2
Average	X	X	X	X	X	X	7.5

X = No δN data were collected for this date

APPENDIX D

THE NITROGEN CYCLE: A SUMMARY

Nitrogen (N) is a building block of amino acids, and therefore is a critical ingredient in all life forms, from soil microbes to macrovertebrates. Many aquatic systems are N-limited, and one critical concern of improperly managed urbanization is the anthropogenic addition of N to such systems. This anthropogenic addition of N can lead to eutrophication, or excessive algae growth. Harmful effects of eutrophication include accumulation of increased organic matter reducing water depth, reduction of light penetrating the water column that can reduce growth of higher-order plants, and reduction of dissolved oxygen available to other life forms including mollusks, shellfish, and fish. While agricultural runoff may be a larger source of eutrophication on a global scale, stormwater runoff from urban centers can have strong negative effects on local water bodies such as streams, lakes, and bays. The predominant theme of this dissertation is to better understand N treatment in bioretention to reduce N loading and eutrophication of receiving waters as a consequence of urbanization.

As a response to requests by my committee, this appendix describes the N cycle as relevant to urban stormwater and bioretention treatment systems. N exists in many oxidation states, facilitating combinations with many other elements to form molecules and ions that all behave differently. N is especially interesting and challenging to study because biological processes can drive dynamic transformations between oxidation states, cycling N through gaseous, ionic, cationic, and organic forms (Figure D.1). Behavior of the relevant N forms and the relevant transformation processes are described herein.

D.1 Relevant Forms of Nitrogen

D.1.1 Gaseous nitrogen

Nitrogen is the most abundant element in the earth's atmosphere. Nitrogen gas, N_2, is a nonreactive gas that makes up 78% of the atmosphere, and is generally considered not to be bioavailable to most plants. However, N_2 gas is the source from which cyanobacteria, rhizobia, and other diazotrophs fix N to form more bioavailable inorganic forms of N. Nitric Oxide (NO) and Nitrous oxide (N_2O) are other gaseous forms of N that are primarily products of industrial and power generation activities, and may be products of incomplete denitrification within bioretention. NO gas is highly reactive and readily oxidized to NO_2, a toxic brown gas that is a major component of smog. N_2O is a powerful greenhouse gas and is considered important in anthropogenic contributions to global climate change. Both NO_2 and N_2O interact dynamically with ozone (O_3) and play important roles in atmospheric chemistry as pollutants near cities and as ozone regulators in the upper atmosphere.

D.1.2 Gaseous nitrogen

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D.1.3 Inorganic nitrogen

The most biologically available forms of N are ammonia (NH_3), ammonium (NH_4^+), nitrate (NO_3^-), and nitrite (NO_2^-). Inorganic N is often the form of N that is limiting in aquatic and terrestrial ecosystems. Microbes produce inorganic N through fixation from the atmosphere or by mineralizing organic N. Plants and algae can then directly use inorganic N to build nucleic acids, amino acids, and proteins such as chlorophyll. Inorganic N is a primary pollution of concern in unmanaged urban stormwater runoff because adding inorganic N to receiving waters can fuel algal growth that causes eutrophication.

Previous studies have reported that about half of the total N found in stormwater is an inorganic form. Of this fraction, half is NH_4^+ and half is NO_2^- and NO_3^-. Sources of inorganic N in stormwater include animal waste, fertilizers, acid rain, and nitrate formed by lightning strikes. Table D.1 provides a summary of N concentrations and sources of each N species found in stormwater.

D.1.4 Organic nitrogen

Organic N is any molecule where N is bound with carbon and hydrogen. These molecules range in complexity from glycine, NH_2CH_2COOH, to highly complex proteins. While some studies have shown direct utilization of simple forms of organic N by plants, organic N is generally considered unavailable to plants unless first mineralized by decomposing microbes. The predominant form of N in urban stormwater runoff has been shown to be organic N. Organic N likely causes little harm to receiving waters in the short term because these forms are not readily useable by algae. However, over time, N pools can build up and when the proper conditions for mineralization do occur, the release of inorganic N can be rapid and cause abrupt, disruptive algal blooms. Preliminary research demonstrates that bioretention is excellent at removing Organic N from influent. Organic N is typically bound in large molecules, and physical filtration is the most likely mechanism of treatment.

D.2 Nitrogen transformations

Both biological and anthropogenic transformations between N forms are relevant to bioretention treatment of urban stormwater. Anthropogenic fixation is a primary source of N pollution in stormwater, and biological processes drive treatment within bioretention. Transformations between the different N pools are shown in Figure D.1. Oxidation states of the N atoms between each of these forms are shown in Figure D.2.

Changes in oxidation state are representative of changes in available energy and the ability of N to act as an electron donor or receptor. As N is oxidized, electrons and energy are given off; energy inputs are required to reduce N from right to left in Figure D. 2.

D.2.1 Fixation

Fixation is the transformation from atmospheric N_2 to inorganic N. This occurs along four primary pathways: 1) anthropogenically through the Haber-Bosch fertilizer production process where N gas is converted to ammonia fertilizer, 2) through combustion of fossil fuels where NO is released then oxidized to form NO_2 and NO_2^-, which causes acid rain that is high in NO_2^-, 3) biological fixation through Cyanobacteria, Rhizobia bacteria, and Frankia bacteria, and 4) through lightning strikes.

The Haber-Bosch process combines N_2 gas and hydrogen in the presence of an iron catalyst under high heat and pressure to produce NH_3 gas. Once in the inorganic from, NH_3 is readily oxidized to create nitrate and urea. This process of synthesizing fertilizer is the foundation of modern agriculture and is responsible for one third of the world's food production. However, misapplication of these fertilizers in agricultural and suburban settings directly contributes to eutrophication of local receiving waters and anoxic marine dead zones. Runoff from suburban lawns and other ornamental landscaping is a primary source of N contamination associated with unmanaged urban stormwater runoff.

Acid rain is also a predominant source of nitrogen in stormwater runoff. One form of acid rain is created when NO_2 gas reacts with water in the atmosphere and creates nitric acid. This acid is contained in the droplets of rain, snow, or fog, and is referred to as wet deposition. While some natural processes can lead to the formation of acid rain, coal-fired power plants are the primary cause of wet deposition in and around urban areas. Acid rain also includes dry deposition, or the result of NO_2 gas reacting with other particles in the atmosphere and creating N-rich dust. However, research by others suggests that dry deposition of N is immobilized through organic and inorganic processes and not likely to contribute to contamination of receiving waters through stormwater runoff (Burian et al., 2001; Harris et al., 1996; Seitzinger et al., 2001; Taylor et al., 2005).

Microbial N fixation is the primary source of inorganic N in natural ecosystems. Cyanobacteria, Rhizobia bacteria, and Frankia bacteria transform atmospheric N_2 into NH_3 using an enzyme called nitrogenase. These bacteria occur in free form in the soil matrix; however, they are more likely in symbiotic relationships with plants and fungi. The Fabacea family, or bean family, is the most common host of N-fixing bacteria. However, some other species such as alder trees and a few genus in the Rosaceae family are also known to host N-fixing bacteria. In these relationships, the host plant provides a carbon source that supplies the electrons necessary to drive the fixation reactions. In exchange, the bacteria produce enough inorganic N to be used by plants. While some species are known to be able to use NH_4^+ directly, denitrifying bacteria that convert the NH_4^+ to NO_3^- are often also required before plants can use N fixed by microbes. Microbial N fixation is an energy-intensive process, and is likely not a significant source of N if runoff from urban or agricultural settings or N in water leaching through sub-surface flow because microbes will only fix N if it is limiting to their growth. The use of plant species known to associate with N fixing bacteria in bioretention is a topic for further research because many of these plants, such as alfalfa, assimilate greater amounts of N in their tissue than other species, but it is unknown if conditions in bioretention sites would facilitate fixation and therefore N addition to receiving waters or contribute to treatment.

Atmospheric N can also be fixed into nitrate and nitrite in the atmosphere by lightning strikes. A bolt of lightning can provide sufficient energy to break the triple bond of N_2 gas, and in the presence of oxygen and/or water, NO_2^- and NO_3^- can be formed. Wet deposition of N by lightning is likely to contribute small amounts of N to surface waters relative to the anthropogenic processes discussed above.

D.2.2 Assimilation

Assimilation is the biological process of plants and microbes taking up inorganic N forms and incorporating this N into proteins, tissue, genetic material, or other complex carbon-bound molecules. Assimilation is hypothesized to be a primary mechanism of treatment in bioretention. The complex N-containing molecules that are formed in assimilation are not readily transported through ecosystems, nor are they bioavailable to the life forms that drive eutrophication in receiving waters without prior biological processing.

D.2.3 Sorption

Sorption is the physical and chemical attachment of N to media, soil organics, and roots. There are two primary types of sorption relative to N treatment in bioretention: adsorption and ion exchange. Adsorption is the physical bonding of dissolved ions to media. Media used in bioretention and organic material growing in bioretention are typically negatively charged, and so only positively charged ions such as NH_4^+ will bind to them. Ion exchange is the exchange of a weakly bonded ion for an ion that will create a stronger bond at a particular bonding site. For example, if an ion is adsorbed to a soil particle through a single bond, and a different ion that would create a double bond is introduced in solution, the singly bound ion will be released and replaced by the ion that would create a double bond. These kinematics are highly dependent on temperature, pH, salinity, media properties, and relative abundance of different ions in solution, and are very difficult to predict. However, general principles suggest that runoff that is low in pH or high in metal content may create an environment where bound nitrate and nitrite may be remobilized through ion exchange of stronger ions such as metals and salts. These abiotic sorptive reactions can remove ions from solution at very high rates during storm events; however, these mechanisms can easily be undone under the proper physical conditions. Sorption must be followed by assimilation in order to achieve long-term, sustainable N treatment in bioretention.

D.2.4 Immobilization

Immobilization is the removal of N ions from stormwater as it flows through the bioretention cell. Immobilization includes the combination of sorption and assimilation. Sorption is suspected to facilitate attachment of N ions to soil and organic media at high rates, allowing plans and microbes the opportunity to assimilate the sorbed ions at a slower rate. Both mechanisms are necessary for sustainable N removal from stormwater; immobilization refers to the removal of N from solution without differentiating the exact mechanism.

D.2.5 Mineralization

The decomposition of complex organic N to bioavailable, inorganic N is called mineralization. Biological processes that mineralize organic N are often slow processes that limit ecosystem productivity in natural settings. Mineralization of organic N is energy-intensive, and is slowed when inorganic N is readily available. As a result, N additions to bioretention from stormwater runoff may slow decomposition and mineralization, resulting in longer immobilization of assimilated N.

D.2.6 Denitrification

The ultimate goal of N treatment facilities is to obtain denitrification, or the conversion of nitrate to N_2 gas. This process happens through denitrifying bacteria that reduce NO_3^- to NO_2^- then to N_2 gas under anoxic conditions, or in the absence of O_2. Denitrification requires the presence of denitrifying bacteria, anoxic conditions, and a carbon source for the denitrifying bacteria to feed on. If a carbon source is available, these conditions can occur in natural environments on a microscale in time and space, including the underside of convex clay lenses or in micropores in soil media, or these conditions can develop on a larger scale if water is allowed to collect and pond over time without draining. Incomplete reduction can occur, resulting in the production of NO and NO_2 gas, the former a powerful greenhouse gas. Denitrification has been hypothesized as the primary N treatment mechanism by bioretention researchers in mesic climates.

It has been well documented that bioretention effluent is commonly higher in NO_3^- than the influent, resulting in net NO_3^- leaching to receiving waters (Brown & Hunt 2011;Davis et al., 2009; Lucas & Greenway 2011). Insufficient carbon resources to fuel microbial denitrification are likely the cause of this effect, which indicates that the proper conditions for denitrification have not been met for a sufficient enough time to facilitate full denitrification.

D.2.7 Annamox

Annamox is the abbreviation for Anaerobic Ammonium Oxidation, and is the recently discovered process of combining NO_3^- and NH_4 to produce N_2 gas and water. This process is believed to be a primary reaction that limits N availability to primary production in oceans and has been identified as an effective process for treating N-rich wastewater. The role of Annamox in bioretention is unclear at this time. Annamox is not included in Figure D.1.

D.3 Nitrogen in bioretention: A narrative

In an undeveloped landscape, precipitation containing background levels of inorganic N would be infiltrated into vegetated soils and incorporated into the ecosystem before the water percolated to a near-by waterway. In unmanaged urban settings, inorganic N levels in falling precipitation are elevated from industrial emissions, then, instead of falling on vegetated soils, the N-enriched precipitation falls on imperious surfaces. Additional N from previous atmospheric deposition, pet waste, or missapplied fertilizers is entrained as the water flows over urban impervious surfaces and into local surface waters. These surface waters are then artificially enriched with bioavailable inorganic N, which facilitates algal blooms and eutrophication.

Bioretention is an engineered ecosystem designed to capture stormwater at or near the place that it falls to minimize anthropogenic inputs of N to local receiving waters. The N contained in the runoff flows into the garden, where appropriately selected media and plants can immobilize the N first through sorption, then through assimilation or denitrification as would have occurred in an undeveloped natural ecosystem. During a storm event, organic and inorganic N is sorbed to the selected media, plant roots, and microbes, which removes the N from the water as it flows through the bioretention cell. Then, soon after the storm event, plant respiration and transpiration increase, accelerating N assimilation into biomass. Once assimilated, the N is no longer a risk to local waterways because slow natural processes are required to mineralize organic N forms before the N is again bioavailable. Influent N may also fallow a denitrification pathway, reducing inorganic N to N_2 gas. In order to achieve this, a bioretention system must first fix enough carbon from the atmosphere through plant activity to fuel denitrifying bacteria then must be designed to allow inundation for long enough a period that the soils become temporarily anoxic. If either condition is not met, denitrification may be limmited, resulting in buildup of NO_3^-, or partial denitrification may occur, resulting in the undesired production of NO gas and NO_2 gas.

If runoff contains excessive concentrations of heavy metals, salts, or low pH from acid rain, nitrogen treatment potential of bioretention may be compromised. High metals, salts, and acid concentrations may negatively affect the microbial community that drives all nitrogen treatment processes within bioretention. Further, high metals, salts, and acid concentrations may also promote ion exchange, resulting in the release of previously bound N ions. Some plant species can tolerate high metal and salt concentrations in soils, and have likely evolved with complex soil microbial communities to facilitate this tolerance. Thresholds of metal, salt and acid concentrations that bioretention can tolerate before being negatively impacted is likely dynamic, and depends heavily on soil media properties and selected plants. Further study into plant and microbe tolerance of metals and salts as well as the effects of low-pH precipitation are all needed.

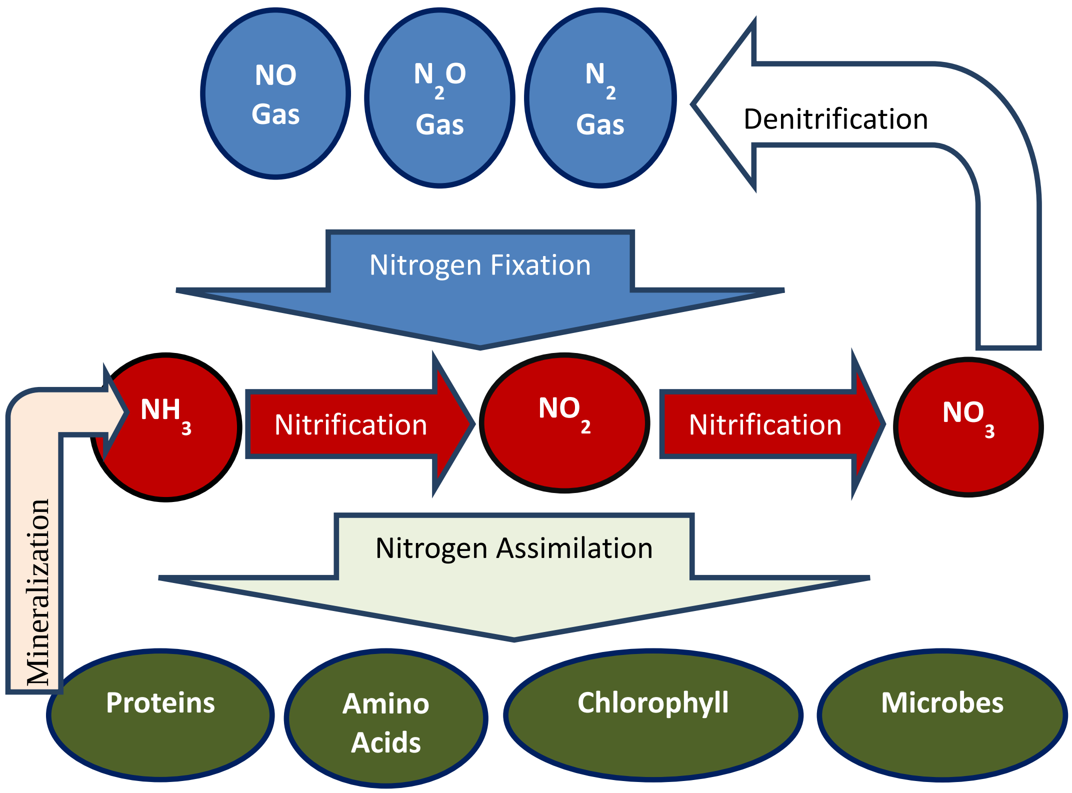

Figure D.1. The nitrogen cycle. Blue represents gaseous N forms, red represents ionic N forms, and green represents organic N forms. Arrows represent the biological and abiotic processes that change the oxidation state of N, facilitating the formation of different molecules.

Table D.1. Summary of nutrient values in stormwater runoff reported in previous studies. The average of these values was used as the target nutrient concentration for the applied synthetic stormwater.

Study	Location	TN	ON	TKN	NH_4	NO_x	DON	TP
Taylor et al. (2005)	Melburn, Aust.	2.13	1.1	1.39	0.29	0.74	0.6	
Hunt et al. (2006)	N. Carolina	1.35	0.57*	0.8	0.23	0.4		0.11
Burian et al. (2001)	s. California				0.37	1.42		
Dietz and Clausen (2006)	Connecticut	1.2	0.5	0.4	0.04	0.3		0.015
Sharkey (2006)	N. Carolina	1.89	1.41*	1.67	0.26	0.23		0.29
	Averages	1.64	0.80	1.07	0.24	0.62	0.60	0.14
Measured average synthetic stormwater concentrations		1.69	1.0		0.43	0.29		0.21

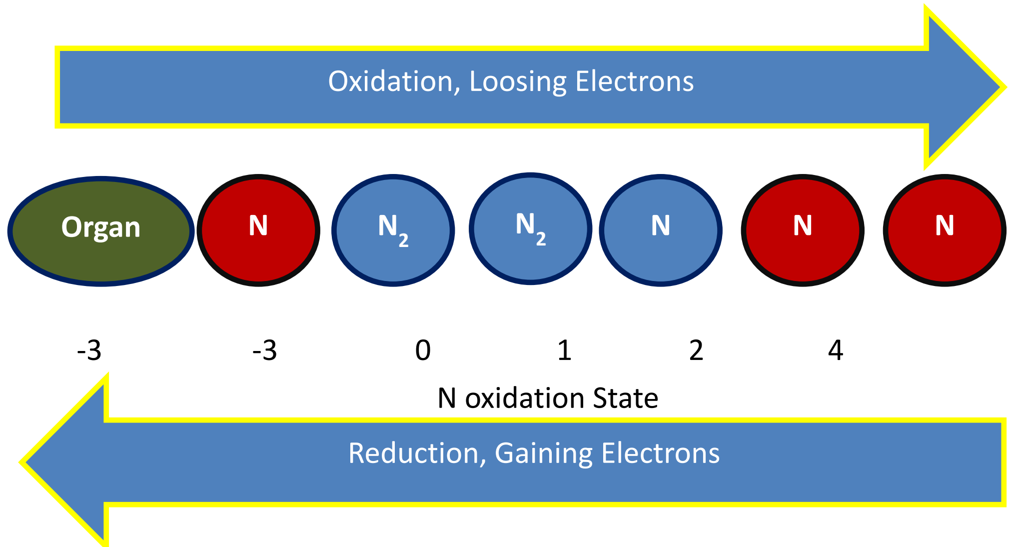

Figure D.2. Oxidation states of N in common forms. Organic N includes amino acids, proteins, and any molecules containing a carbon-nitrogen bond. Red circles indicate inorganic N forms, and blue circles represent gaseous N.

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