

The Performance Impacts of Real-Time Control on Bioretention

A Dissertation Presented for the

Doctor of Philosophy

Degree

The University of Tennessee, Knoxville

Padmini Poojadevi Persaud

May 2021

Dedication

This dissertation was completed for all of the versions of myself. For the past, present, and future.

Acknowledgements

On my graduate journey the legion of incredible people who have influenced and supported me cannot go without mention. First, I would like to thank my family. These are individuals who are pillars of strength, wisdom, and kindness that I hope to also emulate in my life. My parents, Roopa and Lloyd, have always been a source of encouragement and my inspiration to work hard. From the day I was born I have been blessed with two of the most dedicated and tireless people raising me. Every violin lesson, club meeting, girl scout event, and graduation, they have always been there. My sister, Tilka, has become my closest confidant and her words have always been uplifting. She works so hard to be a good sister, friend, and human. I will always be proud of her. I can only hope that I can be as big of an inspiration to her as she is to me. My grandmothers, Barbara and Tilka Sr., are the fiercest women I know and are truly forces to be reckoned with. They are sources of energy and light in all facets of life. My grandfather, Hame, is the first Dr. Persaud and his dedication to his profession, community, and family has set a precedence for people throughout the world; a legacy I hope to also carry forward in my own way.

Over the years I've also made many friends, of which three have been the best. In high school I threw a volleyball and hit someone in the face. Little did I know that would be the start of many adventures, stories, and memories with Zahra Smith. A brilliant artist and creative soul, she has kept me grounded. Margy Orozco has the most drive of anyone I know. Always dedicated to bettering herself and selflessly supporting others, she has always been there for me. Evan Sadlon is a person with the most extraordinary stories. From the 7-11 runs in Sunrise to the hunt for Szechuan sauce in Atlanta, I can always count on our friendship. Knowing him is to know a person who pushes you to be constantly creative and enterprising.

Doing undergraduate research at UF, I was most notably influenced by the graduate students I was working under. Avni Solanki, Kelly Landry, Jerrine Foster, Anna Ness, Yue Hu, and Regina Rodriguez encouraged me to apply for doctoral research and also mentored me in the process as well. They are truly powerful examples of the female engineer. When I finally came to Knoxville, Whitney Lisenbee welcomed me into the Hathaway Research Group (HRG) and became my first real friend. She is a person who I will always admire for her work ethic, dedication, and warmth. My time in the HRG was supported by its members: Jessica Thompson, Andrew Tirpak, Mathew Howard, Aaron Akin, Victoria Rexhausen, Gillian Palino, Katie Lance, Karen Abercrombie, Shengnan Zhou, Ghada Diab, Probal Saha, and Matthew Weathers. The welcoming and encouraging atmosphere truly made my grad school experience memorable. Of course, the leader of the HRG, Dr. Hathaway, is by far the most incredible advisor any grad student could ever hope to have. He has been a teacher, mentor, and friend to me throughout these four years and I am confident that we will continue working together for years to come.

Finally, I would like to thank my fiancé, Darrian Winston. He is my best friend. From the first moment I mentioned wanting to go to grad school, he has been my biggest cheerleader. From helping me get media for my columns, and coming to water them with me, to two years later listening to me practice presentations for my qualifying exams and now again for my defense, he has always been there. He motivates me to do and be the best I can be in every aspect of life. He is a person who leads by example and is someone I will always cherish for the sacrifice, understanding, and love he has so graciously embodied.

Abstract

Bioretention is a green infrastructure practice used to restore natural hydrologic regimes and improve stormwater quality. Extreme weather combined with demands for treatment of a growing number of priority pollutants has put a strain on these systems, making meeting performance goals difficult. The addition of smart stormwater technology has the potential to help address these issues as more control is provided to modify internal conditions and optimize sometimes conflicting outcomes. The use of real-time control is tested in this research to determine how bioretention performance is impacted by this technology. Column studies incorporating forecasts and historic rainfall were used to test two real-time control designs and compare them to traditional free draining and internal water storage bioretention systems. Water quality, hydrologic, and microbial data were all collected to analyze the influence of real-time control on nutrient reduction, volume reduction, and microbial populations. Results of this study show that real-time control can strike a balance between traditional bioretention designs in regard to water quality and hydrologic performance. Real-time control systems also promoted higher growth of nitrifying and denitrifying bacteria by creating a more dynamic wet and dry zone within the soil media. Further research into the role of weather uncertainty is necessary to determine if the predictive elements of the systems are truly necessary. This work can also be expanded to other pollutants of interest and should lead to field studies where laboratory results can be verified. Finally, additional research regarding functional genes in microbial communities will also be necessary to fully understand the activity of nitrifying and denitrifying bacteria in these systems.

Table of Contents

INTRODUCTION.....	1
REFERENCES	4
BACKGROUND.....	7
DEVELOPMENT OF URBAN STORMWATER TREATMENT	7
URBAN STORMWATER COMPOSITION	9
GREEN INFRASTRUCTURE: BIORETENTION.....	11
Bioretention Cells: Physical Attributes	12
Bioretention Cells: Design Configurations.....	15
Bioretention Cells: Internal and External Considerations	20
Bioretention Cells: Performance.....	23
REAL-TIME CONTROL SCHEMES:	27
KNOWLEDGE GAPS AND CONTRIBUTIONS.....	29
REFERENCES	32
CHAPTER 1: REAL TIME CONTROL SCHEMES FOR IMPROVING WATER QUALITY FROM BIORETENTION CELLS	45
ABSTRACT.....	46
INTRODUCTION.....	47
METHODS	50
Bioretention Column Design	50
Bioretention Column Monitoring	53
Weather Data	53
Experimental Procedure.....	54
Water Quality Sampling	57
Statistical Analysis	58
RESULTS AND DISCUSSION.....	58

Soil Moisture and Active Control.....	58
TSS	61
Metals	63
Nutrients	66
Influence of Antecedent Conditions	71
Overall Comparison of Treatments	72
CONCLUSIONS	74
ACKNOWLEDGMENTS.....	75
REFERENCES	76
CHAPTER 2: HYDROLOGIC IMPLICATIONS OF REAL-TIME CONTROL ON BIORETENTION	82
ABSTRACT.....	83
INTRODUCTION.....	84
METHODS	87
Column Design.....	87
Column Monitoring	89
RTC Treatments	89
Experimental Procedure.....	90
Statistical Analysis	94
RESULTS AND DISCUSSION.....	95
Water Balance.....	95
Nitrogen Mass Balance.....	112
CONCLUSIONS AND RECOMMENDATIONS.....	114
ACKNOWLEDGMENTS.....	116
REFERENCES	117
CHAPTER 3: REAL-TIME CONTROL INFLUENCE ON MICROBIAL FUNCTIONALITY IN BIORETENTION	122
ABSTRACT.....	123

INTRODUCTION.....	124
MATERIALS AND METHODS	127
Column Design.....	127
Experimental Procedure.....	128
Water Quality Sampling	129
DNA Preparation and Analysis	131
Statistical Analysis	132
RESULTS.....	133
Water Quality Results.....	134
Antecedent Conditions	141
16S Analysis	142
Analysis conducted on the entire data set.....	143
Nitrifying and Denitrifying Genus.....	146
Comparisons and Correlation Analysis	152
CONCLUSION	153
ACKNOWLEDGEMENT.....	155
REFERENCES	156
SUMMARY & CONCLUSION	162
REFERENCES.....	167
APPENDICES.....	184
VITA	207

List of Tables

Table 0.1 Stormwater pollutant concentrations	10
Table 0.2. Reported nutrient and metal reductions in previous literature (field and laboratory). All values presented as % reduction. Reports of export indicate nitrogen leaching that occurred.	24
Table 0.3. Reported peak flow and volume reductions in previous field scale studies. All values presented as % reduction.	26
Table 1.1 Sediment contributions and stormwater target concentrations.....	57
Table 1.2 Event Mean Concentration (EMC), Median Concentration, Standard Deviation (Std Dev) and Relative Standard Deviation (RSD) for each Treatment Type	62
Table 2.1 Summary of water balance components (%) with standard deviations.....	101
Table 2.2 Volume reduction and TN removal as %. TN mean, median, Standard Deviation (Std Dev), and Relative Standard Deviation (RSD) also listed.	113
Table 3.1 Stormwater target concentrations and contributing chemical additions.....	130
Table 3.2 Percent reductions, median effluent concentrations, mean effluent concentrations, standard deviation (Std. Dev.), and relative standard deviation (RSD) for each column type throughout the 6-week study	138
Table 3.3 List of Nitrifying and denitrifying genus selected for analysis	143
Table 3.4 Average relative abundance (%) of Nitrospira and Pseudomonas over the 6-week study period.....	148

List of Figures

Figure 0.1. Schematic of (a) Free Draining and (b) Internal Water Storage Bioretention	17
Figure 0.2. An Idealized Nitrogen cycle schematic within IWS bioretention.....	19
Figure 1.1 Column design configurations	52
Figure 1.2 Soil moisture at 30-cm depth for storm on 10/31/2018	59
Figure 1.3 Average soil moisture at 30-cm depth for each storm. Storms are defined as the 24-hr period following a storm event.....	59
Figure 1.4 Average soil moisture at 60-cm depth for each storm. Storms are defined as the 24-hr period following a storm event.....	60
Figure 1.5 Zn^{2+} outlet concentrations for all treatment types.....	64
Figure 1.6 Cu^{2+} outlet concentrations for all treatment types.....	64
Figure 1.7 Mn^{2+} outlet concentrations for all treatment types.....	65
Figure 1.8 NO_3^- -N outlet concentrations for all treatment types.....	68
Figure 1.9 NO_2^- -N outlet concentrations for all treatment types.....	68
Figure 1.10 NH_4^+ -N outlet concentrations for all treatment types	70
Figure 2.1 Configuration design with altered volume control column	88
Figure 2.2 (a) Graph of the size of each rain event in cm. All other figures (b)-(e) are the water balance for each storm event for each configuration over the 6-week study period	96
Figure 2.3 Water tracking for each column type based on water balance of the 6-week study period	99
Figure 2.4 Spearman correlations comparing Drainage, Seepage and Storage/ET to Influent volumes.....	107

Figure 2.5 Spearman correlations comparing Drainage, Seepage, and Storage to antecedent dry periods	110
Figure 3.1 Bar plot of storm size (cm) for the 6-week study period	134
Figure 3.2 Nitrogen species concentrations for the influent and effluent of each column type..	135
Figure 3.3 Alpha a) Box plots of Observed and Shannon Diversity b) Plot of Alpha Diversity per storm event	144
Figure 3.4 Weighted and unweighted unifrac PCoA plots for all taxa	146
Figure 3.5 Bar plots for relative abundance of nitrifying and denitrifying genus	147

Introduction

In developed areas, impermeable surfaces impede infiltration and prevent nutrient processing from occurring after rain events (Brabec et al. 2002; Bronstert et al. 2002). Growing populations have led to the expansion of impermeable areas to facilitate housing, services, and traffic volume, creating the potential for flooding and public safety issues (Konrad 2003). Stormwater management infrastructure has been developed to quickly move runoff away from cities to prevent flooding; however, this efficient drainage has resulted in downstream environmental consequences (Bi et al. 2015). Urban stormwater is a major contributor of environmental pollutants found in rivers and streams and causes surges in flow in urban water systems (USEPA 2016). Downstream eutrophication caused by nitrogen and phosphorous inputs is common in urbanized watersheds as stormwater carries these pollutants into river and stream systems. The Chesapeake Bay is a popular example of this, as algal blooms and fish kills are commonly experienced in this system (Bosch et al. 2018).

Green infrastructure practices were developed to combat these environmental and hydrologic effects by providing site-scale treatment that mimics natural processing (Fletcher et al. 2015). Bioretention cells are of particular importance in this research as they are designed to lower peak flows while also improving water quality (TSM 2015; NCDEQ 2018). The hydrologic benefits of bioretention include reductions in volume and peak flow as stormwater is infiltrated rather than channeled to surface water. As a consequence, receiving water bodies are not overburdened. Targeted practice designs have aimed to balance hydrologic and water quality challenges; however, innovative system optimize can meet more specific water quality and hydrologic goals can be met when a system is designed specifically to target a particular treatment.

Bioretention cells typically infiltrate stormwater through layers of plants, mulch, media, sand, and gravel to reduce runoff volumes while also allowing pollutant reductions to occur through biologic and physical removal pathways (Laurenson et al. 2013). Despite this, nitrate leaching from these practices have been observed in previous literature and is an ongoing issue for bioretention practices that freely drain via an underdrain (FD). Such configurations do not provide adequate environments for nitrification and denitrification (Hatt et al. 2009; Gilchrist et al. 2014; Manka et al. 2016). FD configurations allow for aerobic processing within bioretention cells as gravity drives drainage and water quickly passes through. While useful for performing nitrification, denitrification in this system occurs only in anaerobic microsites in the media. The lack of anaerobic areas in FD practices allows nitrification to dominate, converting ammonium into excess nitrate that is exported. Improved bioretention designs form an anaerobic zone by creating internal water storage (IWS) (Hunt et al. 2008; Glaister et al. 2017; Persaud et al. 2019). The use of IWS allows nitrification to occur in the upper aerobic area of a practice, and denitrification to occur in the lower anaerobic zone (Li et al. 2014). While this configuration is better for water quality purposes, in some cases it can limit the volume of water a practice can infiltrate during a storm event. Further, studies such as (Persaud et al. 2019) have shown that nitrification may sometimes be hampered in these systems. Bioretention designs which balance between these system types may allow proper water quality and hydrologic processing.

Innovations in real-time control (RTC) technology show promise in managing stormwater flows and improving water quality in green infrastructure practices (Mullapudi et al. 2018; Middleton and Barrett 2008). RTC uses a system of automated valves to maintain prescribed water levels within a practice and accommodate environmental changes, like incoming weather, while monitoring system conditions (Kerkez et al. 2016; Parolari et al. 2018). In theory, using RTC to

control the storage and release of water from a bioretention cell has the potential to improve water quality and change hydrologic processing, (Roman et al. 2017; Middleton and Barrett 2008) however, limited research has been performed to determine the actual influence RTC has on these practices. In specific regard to nitrification and denitrification, the implementation of RTC would also affect soil moisture and subsequently, the microbial populations within the bioretention cell. As microbes are responsible for executing nitrogen conversion pathways, they are essential when considering changes in water quality. By targeting the physical characteristics that influence microbial populations, RTC has the potential to maintain ideal conditions for nutrient processing.

The effects of RTC use for bioretention in regard to water quality, hydrology, and microbial abundance are currently unknown. The chapters of this proposal will ask questions regarding 1) the effect of real-time control systems on water quality, 2) the effect real-time control systems have on hydrology with targets on volume reduction and infiltration effects, and 3) the effect of real-time control on microbial populations. This research aims to further the potential of real-time control systems in bioretention and better understand the limits and potential of this technology. This work will conclude with suggestions for control schemes and directions for future research that could guide the use of RTC in other green infrastructure practices.

References

- Bi, E. G., F. Monette, and J. Gasperi. 2015. 'Analysis of the influence of rainfall variables on urban effluents concentrations and fluxes in wet weather', *Journal of Hydrology*, 523: 320-32.
- Bosch, D. J., M. B. Wagena, A. C. Ross, A. S. Collick, and Z. M. Easton. 2018. 'Meeting Water Quality Goals under Climate Change in Chesapeake Bay Watershed, USA', *Journal of the American Water Resources Association*, 54: 1239-57.
- Brabec, E., S. Schulte, and P. L. Richards. 2002. 'Impervious surfaces and water quality: A review of current literature and its implications for watershed planning', *Journal of Planning Literature*, 16: 499-514.
- Bronstert, A., D. Niehoff, and G. Burger. 2002. 'Effects of climate and land-use change on storm runoff generation: present knowledge and modelling capabilities', *Hydrological Processes*, 16: 509-29.
- Fletcher, T. D., W. Shuster, W. F. Hunt, R. Ashley, D. Butler, S. Arthur, S. Trowsdale, S. Barraud, A. Semadeni-Davies, J. L. Bertrand-Krajewski, P. S. Mikkelsen, G. Rivard, M. Uhl, D. Dagenais, and M. Viklander. 2015. 'SUDS, LID, BMPs, WSUD and more - The evolution and application of terminology surrounding urban drainage', *Urban Water Journal*, 12: 525-42.
- Gilchrist, S., M. Borst, and E. K. Stander. 2014. 'Factorial Study of Rain Garden Design for Nitrogen Removal', *Journal of Irrigation and Drainage Engineering*, 140: 11.
- Glaister, B. J., T. D. Fletcher, P. L. M. Cook, and B. E. Hatt. 2017. 'Interactions between design, plant growth and the treatment performance of stormwater biofilters', *Ecological Engineering*, 105: 21-31.
- Hatt, B. E., T. D. Fletcher, and A. Deletic. 2009. 'Pollutant removal performance of field-scale stormwater biofiltration systems', *Water Science and Technology*, 59: 1567-76.
- Hunt, W. F., J. T. Smith, S. J. Jadlocki, J. M. Hathaway, and P. R. Eubanks. 2008. 'Pollutant removal and peak flow mitigation by a bioretention cell in urban Charlotte, NC', *Journal of Environmental Engineering-Asce*, 134: 403-08.

- Kerkez, B., C. Gruden, M. Lewis, L. Montestruque, M. Quigley, B. Wong, A. Bedig, R. Kertesz, T. Braun, O. Cadwalader, A. Poresky, and C. Pak. 2016. 'Smarter Stormwater Systems', *Environmental Science & Technology*, 50: 7267-73.
- Konrad, Christopher P. 2003. "Effects of Urban Development on Floods." In.: USGS.
- Laurenson, G., S. Laurenson, N. Bolan, S. Beecham, and I. Clark. 2013. 'The Role of Bioretention Systems in the Treatment of Stormwater.' in D. L. Sparks (ed.), *Advances in Agronomy, Vol 120* (Elsevier Academic Press Inc: San Diego).
- Li, M. H., M. Swapp, M. H. Kim, K. H. Chu, and C. Y. Sung. 2014. 'Comparing Bioretention Designs With and Without an Internal Water Storage Layer for Treating Highway Runoff', *Water Environment Research*, 86: 387-97.
- Manka, B. N., J. M. Hathaway, R. A. Tirpak, Q. He, and W. F. Hunt. 2016. 'Driving forces of effluent nutrient variability in field scale bioretention', *Ecological Engineering*, 94: 622-28.
- Middleton, J. R., and M. E. Barrett. 2008. 'Water quality performance of a batch-type stormwater detention basin', *Water Environment Research*, 80: 172-78.
- Mullapudi, A., M. Bartos, B. Wong, and B. Kerkez. 2018. 'Shaping Streamflow Using a Real-Time Stormwater Control Network', *Sensors*, 18: 11.
- NCDEQ, North Carolina Department of Environmental Quality. 2018. "Stormwater Design Manual." In *North Carolina Environmental Quality*.
- Parolari, A. J., S. Pelrine, and M. S. Bartlett. 2018. 'Stochastic water balance dynamics of passive and controlled stormwater basins', *Advances in Water Resources*, 122: 328-39.
- Persaud, P. P., A. A. Akin, B. Kerkez, D. T. McCarthy, and J. M. Hathaway. 2019. 'Real time control schemes for improving water quality from bioretention cells', *Blue-Green Systems*, 1: 55-71.
- Roman, D., A. Braga, N. Shetty, and P. Culligan. 2017. 'Design and Modeling of an Adaptively Controlled Rainwater Harvesting System', *Water*, 9: 14.

TSM, Tennessee Stormwater Management. 2015. "Management and Design Guidance Manual." In *Tennessee Stormwater Managment*.

USEPA, United States Environmental Protection Agency. 2016. "National Summary of State Information." In *USEPA*.

Background

The first portion of this dissertation is a literature review of stormwater management and bioretention practices. The first sections detail the historical need for stormwater management as well as associated regulations and infrastructure changes. Next, the development of bioretention and the current state of the practice is described, and finally, real-time control systems and their current usage are examined. While real-time control may constitute the next phase in the evolution of the bioretention practice, offering more connected stormwater management networks, there are still implications for hydrology and water quality that have yet to be explored. Furthermore, public perceptions and trust in system level water management will likely become a bigger factor in the implementation of real-time control as the technology is developed and implemented.

Development of Urban Stormwater Treatment

The advancement of society began with the growth of cities near bodies of water. Rivers and streams provided water for the growth of crops while also allowing transportation. Globally, this proximity to bodies of water determined which major cities would flourish; however, this urbanization has led to an increase in flooding in these local waterways (Konrad 2003). The initial construction of infrastructure was not designed to take environmental health and safety into consideration, it focused on moving water away from cities as fast as possible. Essentially, urban stormwater was diverted directly to the nearby waterways. With this diversion, unintended consequences, like poor water quality and increased peak flow brewed as society continued to grow and change (Willems 2013).

Since this time, urban centers have continued to expand to accommodate population increases. For this reason, old infrastructure in established large cities struggles to keep up with

water and sewage demands (Tauhid and Zawani 2018). The age of the pipes and structures originally implemented also results in issues with leaking, contamination, and environmental dumping (ASCE 2017). Furthermore, this expansion brings with it a larger metropolitan area, which includes not only businesses and stores, but residential areas, schools, and recreational amenities. This land cover conversion is marked by the increased use of asphalt, concrete, and other nonporous materials used for building (Li et al. 2019). Thus, the amount of impervious area in most cities has increased over time, which subsequently means a rise in the amount of urban stormwater that is being produced and degradation of stream and river health (Brabec et al. 2002).

Compounding these concerns, the increasing frequency of extreme weather events brings high intensity rainfall and flooding in some areas and major drought in others (O’Gorman 2015). For instance, the east coast of the United States has been experiencing record-breaking hurricanes while the west coast is subjected to extended drought conditions. Areas experiencing drought face a host of issues regarding water rights and usage, which limits residential water availability and puts a strain on occupants of the area (Cook et al. 2004; Mishra and Singh 2010). Areas experiencing a higher frequency of rain events may not have issues with water availability; however, stormwater carries pollutants built up on impervious areas to streams and rivers following rain events. Large watershed scale impacts resulting from this contamination include eutrophication, fish kills, and economic losses in water centric tourism-based economies (Bullerjahn et al. 2016; Davis et al. 2019).

Stormwater management has historically targeted hydrologic threats, however, environmental impacts and pollution concerns were not addressed. As public interest began developing about water related issues, legislation was passed (Clean Water Act of 1972) to address pollution concerns and set water quality standards for surface water contaminants (USEPA 2002;

Collins et al. 2010). Part of the clean water act included legislation like the National Pollutant Discharge Elimination System (NPDES) which set forth guidance for municipalities to help regulate stormwater discharges from municipal stormwater sewer systems (MS4s), construction sites, and industrial areas (USEPA 2015). While this legislation limits allowable pollutant loads in streams and rivers, strategies for reduction have specifically focused on limiting point source pollutant generation. Mitigation strategies for non-point source pollution are also important as water quality dynamics evolve with changing land-use.

Urban Stormwater Composition

Stormwater is a major contributor of pollutants in rivers and streams (USEPA 2016). When impaired, these systems are classified as (303d) streams or rivers and the sources of contamination are identified (USEPA 2010). A total maximum daily load (TMDL) is established for each contaminated water body which requires a plan for reaching target pollutant loads to improve water quality to meet the intended use of the stream or river. States monitor the waters to track improvements to water quality and subsequent issues that may arise based on local land use (USEPA 2009). While restoration and remediation practices are implemented to better water quality, an integral component to pollutant classification is the understanding of pollutant dynamics.

Stormwater pollutants are generated from a variety of known sources like industrial waste, construction debris, pet waste, fertilizer, and vehicle emissions (Chen and Chang 2014; Hogan et al. 2014). In particular, heavy metals and nutrients are carried into stream and river systems which subsequently pollute bays and river deltas (Table 0.1) (Bratieres et al. 2008). Of particular concern are nitrogen containing nutrients (NH_4^+ , NO_3^- , and NO_2^-) which have been known to cause

Table 0.1 Stormwater pollutant concentrations

Pollutant	Concentration (mg/L)
NO_x - N	0.75
NH₄⁺-N	0.27
TDP	0.04
Cu²⁺	0.05
Zn²⁺	0.14
Pb²⁺	0.25
Cr⁶⁺	0.25
Mn²⁺	0.0045
Fe³⁺	0.03
Ni²⁺	1
Cd²⁺	0.025

eutrophication in large watersheds (Bosch et al. 2018). The resulting algal blooms consume dissolved oxygen in water, which then leads to fish kills. Recreational use of these waters becomes a public health concern, and tourism and fishing industries that may be dependent on local aquatic life are also affected (Bullerjahn et al. 2016; Steffen et al. 2017). Heavy metals in the water can also contribute to toxicity levels, which can influence safe human exposure limits (Delpla et al. 2009; Kobielska et al. 2018). Biomagnification can also become an issue as toxic metals are accumulated in fish species that are harvested for human consumption.

Green Infrastructure: Bioretention

In the latest iteration of stormwater management devices, a variety of green infrastructure (GI) practices were developed to address hydrologic and water quality concerns in developing areas. These practices can be implemented through retrofit as well as predevelopment planning. GI practices were developed to manage some of the water quality and hydrologic issues brought on by urban stormwater (Fletcher et al. 2015). GI can take the form of practices like swales, wetlands, and green roofs depending on the affected area. They are meant to manage stormwater and provide ecosystem services in specific areas; thus, each practice is distinct and adaptable (Levin and Mehring 2015; BenDor et al. 2018). These practices are implemented in areas that generate runoff (parking lots, roads, buildings, residential communities, etc.), and are meant to mitigate stormwater issues at a smaller point-source scale (Cadenasso et al. 2008; Bi et al. 2015; Yang and Chui 2018). In this way, larger downstream effects can be managed, and larger waterways can take on more natural flow patterns. At the same time, they provide multiple ecosystem services to communities like the support of nutrient cycling as well as broader societal benefits prompted by green spaces (Meerow and Newell 2017). As this research focuses on the

design of bioretention areas, the physical qualities of each component in these systems is important to understand in the context of the overall function of a cell. Bioretention is adaptable as cells can be shaped and sized to fit in parking lots, residential communities, or more commercial areas. They are intended to be natural, low maintenance systems, that when properly functioning, promote biodiversity and serve a community.

Bioretention Cells: Physical Attributes

Similar to rain gardens, bioretention cells are depressions in the ground that reduce peak flows and improve water quality. In contrast, however, they typically have a deeper profile than rain gardens and consist of layers of gravel, media, and plants (Figure 0.1) which allow infiltration and nutrient processing (Davis et al. 2003). These practices are implemented in a variety of areas like parking lots, neighborhoods, and even commercial areas. Initial bioretention design focused on restoring pre-development base flow and groundwater recharge. Later variations to bioretention design have then targeted water quality treatment more specifically through physical changes to a practice (Fletcher et al. 2015). Prior research into the different components of this practice show the various influences that media type, plant selection, and practice configuration that result in hydrologic and nutrient processing differences (Laurenson et al. 2013; Payne et al. 2014). The following sections will describe physical attributes of a bioretention practice that affect water quality and hydrologic dynamics of a practice. The layers of a practice are detailed from upper to lower portions starting with plants, then media, and finally drainage layers.

Plant Specifications

Bioretention practices involve the use of plants for a multitude of purposes. They contribute to nutrient uptake, maintain and improve soil porosity over time, and contribute to the aesthetic value of this practice in public spaces (Passeport et al. 2009; Le Coustumer et al. 2012; Glaister et al. 2017; Dagenais et al. 2018). Typical plants used in the practice are native to the area and have a significant influence on nutrient, hydrologic, and microbial processing (Chandrasena et al. 2017). Selected plants are usually very hearty, being drought and water-resistant. Depending on the local climate and weather patterns, the chosen plants will need to be tolerant to local weather to have the greatest chance of survival. While planning a bioretention cell, it is common to have some plant die off and certain plants will have to be replaced (CSN 2013). Planting location within the practice is also an area of concern, as factors like sun exposure, watering needs, and typical root growth should be taken into consideration (CSN 2013; TDEC 2014). For example, trees will require more water than flowers or shrubs and have more lateral root systems. They grow more into surface soils, which indicates that proximity to roads, or buildings will have to be a long-term planning consideration. Flowers like the Purple Coneflower (*Echinacea purpurea*), and Black-eyed Susan (*Rudbeckia fulgida*) provide a colorful aesthetic value in their blooming seasons but should be planted in deeper areas of a bioretention practice to meet their watering requirements.

If a practice is intending to target pollutant removal, specific plant usage can also contribute (Chandrasena et al. 2017; Glaister et al. 2017). Variations in nitrogen sequestration rates based on plant type were exhibited in research performed by Payne et al. (2014). That study also found that plants have a limit to the amount of nutrient uptake they can perform. More specifically, the plants are not able to sequester any more nutrients in a pollutant stream once their quota for nutrients is satisfied. A study performed in Australia by Bratieres et al. (2008) found *Carex* spp. was able to

remove total nitrogen, while Read et al. (2008) found the same for *Juncus* spp. However, Read et al. (2008) also noted that *Juncus* spp. did not remove lead. In this way, other processing in a bioretention cell would need to occur to better serve the system and limit poor water quality effluent.

Bioretention Media

Bioretention media mixtures differ based on location depending on local soil needs and regulations. In the state of Tennessee, bioretention media is comprised of sand (85-88%), fines (8-12%), and organic matter (3-5%) (TSM 2015). The high sand content allows runoff to infiltrate the soil while the organic matter acts as a carbon source to allow plant growth and feed microbial activity (Thompson et al. 2008). Proper soil media can heavily influence bioretention performance (Hatt et al. 2009a). The media as a whole should generally be nutrient poor to allow pollutant absorption onto soil particles as one avenue of water quality improvement (Wang, Chua, et al. 2017; Yan et al. 2017). When soils are not nutrient or metal deficient, there is risk of leaching. Nutrients exported from a bioretention cell can cause eutrophication issues further downstream, thus negating their intended benefits. Saturated conditions also allow the potential for metal ions to mobilize and transport through effluent (Lovley et al. 2004). There is also a risk for clogging as TSS is trapped in the upper portions of the system, typically in the mulch layer. This can also impede infiltration (Liu et al. 2014)

Organic matter

The use of organic matter in bioretention practices has various benefits and drawbacks. Typically, in the form of compost, it is nutrient rich and can promote plant growth and

establishment in the early stages of building a practice. Mixed into bioretention media it can also keep soil moist for longer periods, which can be useful in maintaining plant health in dry climates. However, it can also contribute to the nutrient export from a system (Bratieres et al. 2008; Gilchrist et al. 2014; Jay et al. 2019). This can also be seen when nutrient rich mulch is used in topsoils. Generally, the small amount of organic matter mixed into bioretention media give plants and microbes a carbon source to fuel their growth and function (Houdeshel et al. 2012). This should be done with caution, as plant establishment should ideally occur without fertilization to limit the nutrient inputs of the system. As a practice establishes, carbon inputs from the plants can migrate further into the depths of a bioretention practice and further enrich microbial processing as well (Passeport et al. 2009). In this way, the system should be resupplying itself with carbon inputs, thus additional organic matter and fertilizers are unnecessary.

Drainage Layers

Under layers of bioretention media are drainage layers of gravel and pea gravel, which are common in bioretention design as they allow water to be stored before being infiltrated into surrounding soils. Underdrains built into these layers allow for additional drainage in cases of over-inundation. The gravel and pea gravel layers also provide separation between bioretention media and surrounding soils. This allows continuous drainage of media and increases the total volume of available storage in the system. This is important during continuous rain events where volume reduction is important.

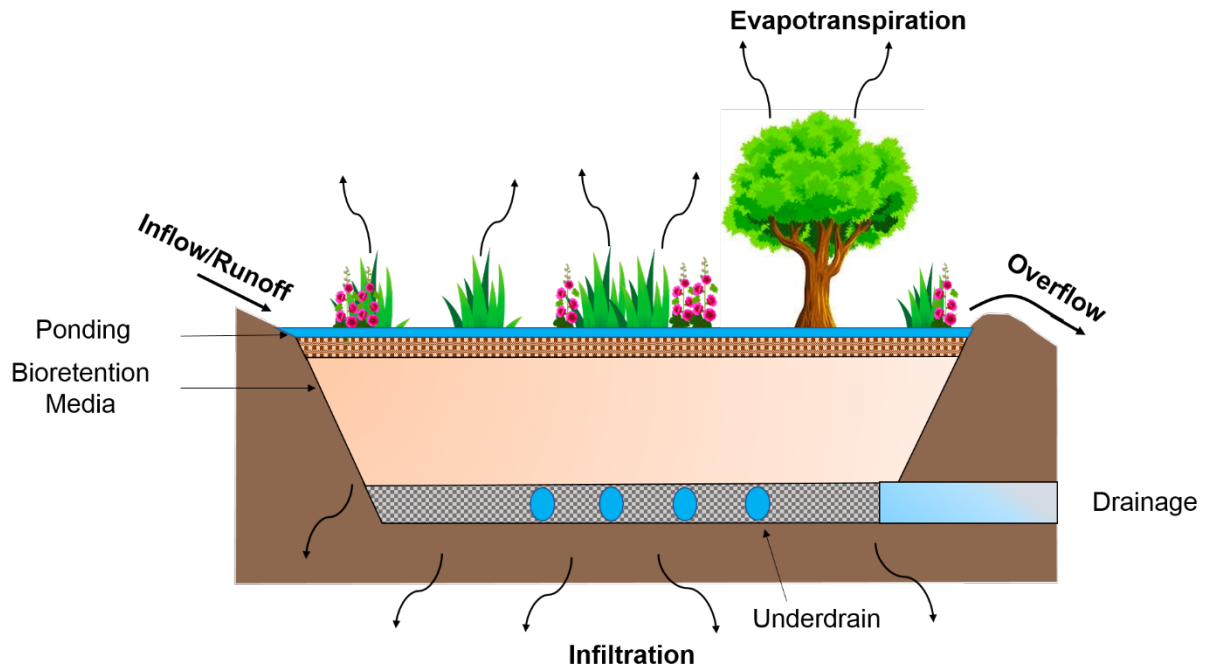
Bioretention Cells: Design Configurations

Even while the materials used inside of a bioretention practice can remain the same, differences in outlet configuration mark more functional differences in bioretention practices.

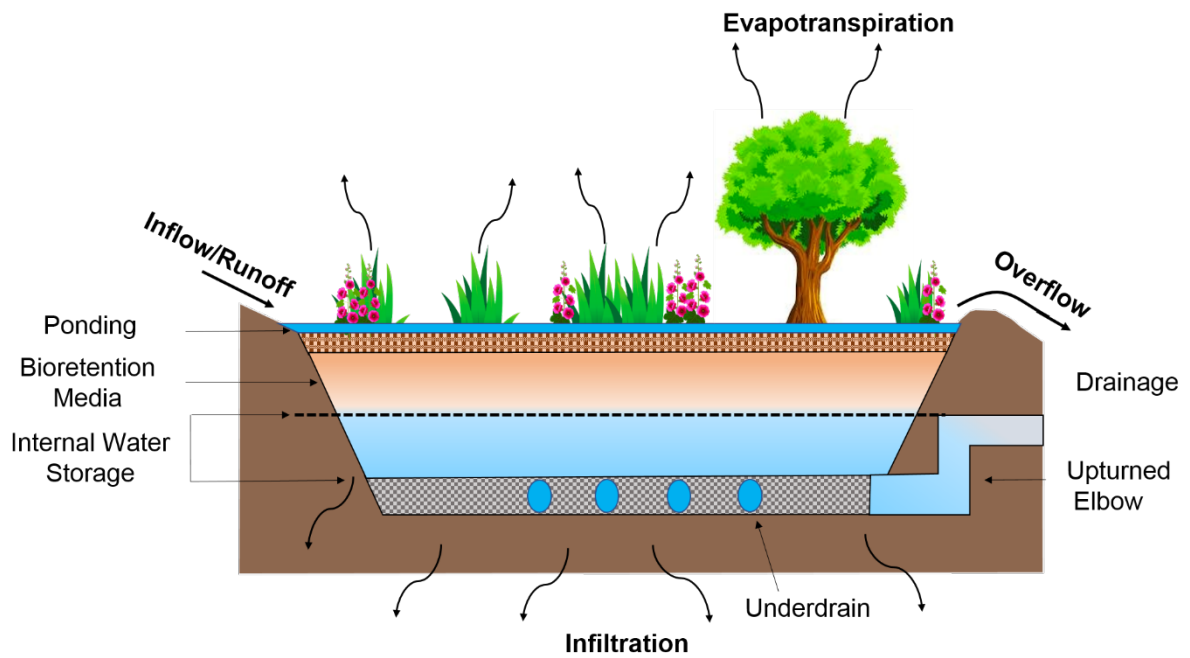
Overall design configuration is an important factor in bioretention function as they force particular processing conditions to form. Every bioretention cell is site specific in construction and focuses on specific goals. Local needs influence the design of bioretention cells to serve a community or watershed. For instance, in some cases, a cell may manage greater stormwater volumes, while in another case the design may target nutrient processing. Free draining (FD) and Internal Water Storage (IWS) designs are the two design strategies currently employed in the United States (Figure 0.1).

Free Draining (FD) Bioretention

FD bioretention systems are the first iteration of bioretention design in the United States. They infiltrate water as much as possible to aid in runoff reduction and groundwater recharge. In the United States, these systems are unlined, which allows stormwater to infiltrate quickly through media and into surrounding soils (Dietz and Clausen 2005). In some cases, they are also equipped with underdrains, which channels flows into existing stormwater networks when the system is overwhelmed. Because of their infiltration-based design, these systems have greater storage potential in their soils compared to other design configurations; however, they commonly leach nutrients such as nitrate and may not infiltrate as much because water is allowed to pond in their system (Hatt et al. 2009b; Blecken et al. 2010; Gilchrist et al. 2014; Li et al. 2014). When FD systems operate with an underdrain, their design is aerobic in nature, allowing limited anaerobic processing to convert nitrate into nitrogen gas (Davis et al. 2006; Hsieh et al. 2007; Hunt et al. 2008). Nitrate reductions that do occur typically happen within anaerobic microsites in the soil matrix.



(a)



(b)

Figure 0.1. Schematic of (a) Free Draining and (b) Internal Water Storage Bioretention

Internal Water Storage (IWS) Bioretention

IWS systems target nutrient processing through their design. While these systems are unlined in the United States, their outlets are either elevated or modified from those of FD, preventing discharge directly into stormwater systems and creating an anaerobic zone at the bottom of the practice (assuming soils have a low enough in situ infiltration rate) (Figure 0.2). The upper portion of an IWS system is unsaturated to create an aerobic zone. This also maintains plant species as the root zone is oxygenated. By allowing both aerobic and anaerobic zones to form in the system, better processing of nutrients, particularly nitrogen can occur. As more complete nitrification and denitrification are allowed to occur within these systems, there is a greater reduction in nitrate (Hunt et al. 2008; Brown and Hunt 2011; Glaister et al. 2017). The issue with using this practice is the limited storage space (Mangangka et al. 2015). The creation of the saturated zone limits the amount of incoming stormwater managed. Hydrologically the practice may not be able to reduce as much volume but allowing water to collect within the system pushes infiltration and groundwater recharge further (Brown and Hunt 2011; Hunt et al. 2012). Formation of the anaerobic zone is further promoted in areas where surrounding subsoils have lower infiltration rates. Water is held within the bioretention system for longer periods which aids in creating and maintaining anoxic conditions. Areas with sand or silt subsoils, that are unlined, cannot maintain an IWS as easily because of the greater infiltration rates of those soils.

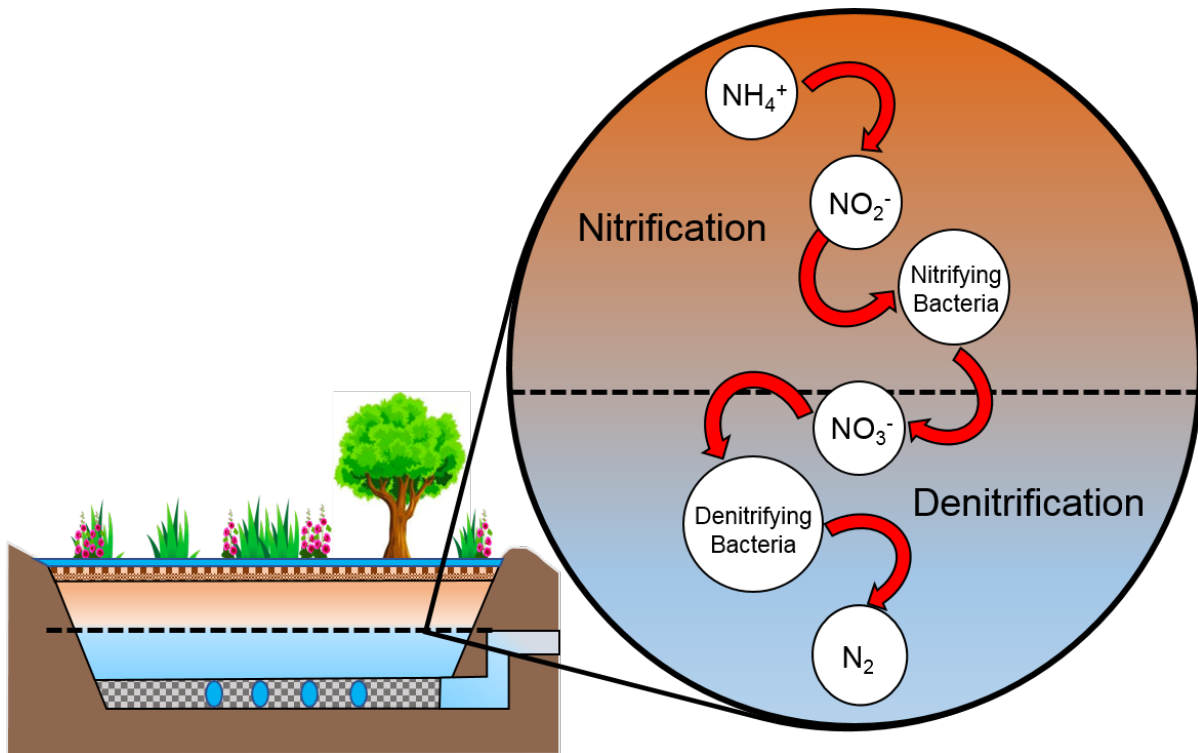


Figure 0.2. An Idealized Nitrogen cycle schematic within IWS bioretention

Bioretention Cells: Internal and External Considerations

Drying and Wetting Cycles

Wet and dry cycles fluctuate on short (daily) and long (seasonally) time scales, but for the purposes of this research, weekly time frames are considered. These wet periods are followed by dry periods that, similarly, can last a varying amount of time. Bioretention practices are subjected to these variations in weather, which can impact nutrient processing by altering conditions for plant and microbe function (Fierer and Schimel 2002; Blecken et al. 2009b). Drying and wetting cycles take on different characteristics and timescales based on local climate, but in general can add unintended stressors to the system (Liu et al. 2013; Bi et al. 2015). The following will be a closer look at how design characteristics influence microbial activity and nitrogen processing specifically.

Microbial activity

Periods of inundation or drought lead to microbial inactivity as microbes are sensitive to environmental stressors such as soil water content (Fierer and Schimel 2002). Indeed, plant type, soil moisture, oxygen levels, temperature, and carbon availability are among the many influences that affect microbial activity or inactivity (Vangestel et al. 1993; Peralta et al. 2013; Waller et al. 2018). The biggest contributor to all of these factors is stormwater inflow patterns. When there is no inflow, microbes enter into dormancy and essentially stop functioning (Schimel 2018). Until the restoration of soil moisture levels, microbes will not become active, or process pollutants, which elucidates the concerns that arise with bioretention practices in dry conditions. An indicator of resuscitation is a spike in carbon dioxide production before returning to normal functional conditions (Placella et al. 2012).

The sensitivity of microbes forces them to react to changing environmental conditions rather than fight them. Microbes facing these extended stressors have limited functionality without external intervention; however, they have shown signs of adaptation such as increases in ribosome under water stress to yield higher protein synthesis under improved conditions (Sukenik et al. 2012; Salazar et al. 2018). In other instances, environmental primers are suspected of sustaining microbial populations (aerobic in flood conditions and anaerobic in drought) to allow continued performance (Peralta et al. 2013). With promising signs of evolutionary adaptation, microbial communities seem to be able to adapt to new environmental extremes. This does not negate the need to design for their optimal performance in nutrient processing systems and considerations for promoting microbial health in bioretention practices are important. Research on specific microbial conditions in bioretention have focused on design configurations and plant selection that can trigger nitrification and denitrification (Morse et al. 2018; Chen et al. 2019; Igielski et al. 2019). These studies focus on the functional genes present in a practice and how presence of these genes correlates to design specifications. IWS systems are correlated to more denitrifying genes than FD systems, and so recommendations are to focus on IWS design to promote these bacteria and complete nutrient processing (Chen et al. 2013), however, optimizing this design has not been done and alternative bioretention design has not yet been considered. Chapter 3 in the following research will identify nitrifying and denitrifying bacteria associated with new bioretention design as real-time control systems are used in alternate design configurations.

Nitrogen processing

Of particular importance in this research is the nitrogen processing capability of bioretention practices. As noted above, nitrogen compounds cause eutrophication in downstream

systems (Zhang et al. 2019) and bioretention practices have the potential to process these compounds through nitrogen cycling (Davis et al. 2006). Complete nitrogen cycle processing involves nitrification and denitrification, which, respectively, are oxic and anoxic conditions. Aerobic environments are oxygen rich and are typically in the upper portions of a bioretention cell (Hsieh et al. 2007; Mangangka et al. 2015). This area includes plant root zones and nitrifying bacteria like *Nitrobacter* and *Nitrosomonas*. Through nitrification, ammonia converts to nitrate which is one of the first steps in the nitrogen cycle (Zumft 1997).

In FD systems, the main anaerobic sections for performing denitrification are microsites found within the soil. Large anaerobic environments are created when IWS systems are built and a saturated zone forms in the lower portion of a practice (if in situ soils are poorly infiltrating and allow saturation) (Li et al. 2014). This oxygen poor environment promotes denitrifying bacteria like *Pseudomonas*, and *Achromobacter* species, which convert nitrate to nitrogen gas, completing the nitrogen cycle within the system (Madigan 2019). Analysis in later chapters will focus on nitrifying and denitrifying gene identification, however, many species are capable of performing these functions (Chen et al. 2013). An idealized schematic for nitrogen processing in bioretention is displayed in Figure 0.2. The formation of N_2O in denitrification is also possible but should be avoided as this is a greenhouse gas.

Facilitation of this process in bioretention has primarily focused on overall design alterations that promote aerobic or anaerobic environmental conditions, however, plant type, and media properties are influential as well (Bratieres et al. 2008; Read et al. 2010). These factors can stimulate microbial populations, or act as nutrient sinks themselves and contribute to nitrogen removal within bioretention. Previous research has used these factors to examine microbial

contributions to nitrogen processing and found that microbe productivity is also influenced more by design than by environmental factors (Morse et al. 2018; Waller et al. 2018).

Bioretention Cells: Performance

Bioretention cell performance is evaluated based on water quality (nutrients and metals) and hydrology (peak flow and volume reduction) goals. A sampling of the current literature shows variations in performance which further inform the subsequent chapters of this research.

Water Quality

In this research, particular emphasis has been placed on nitrogen reduction as facilitation of nitrogen processing has proven to be variable. Table 0.2 highlights these findings as 15 studies focusing on water quality objectives pertinent to this research. The results showed variable performance for nutrient reductions. These studies represent a variety of lab scale and field research that was conducted to test various removal enhancement techniques. The differences in nutrient reductions had to do with a host of variables in testing, such as the use of compost in media mixes. There were also differences in outlet configuration (some using free draining and some using internal water storage systems) and environmental factors, like wet and dry periods, would have also influenced these results. Of note are the studies that reported nitrogen exports whether that be NO_3^- -N or TN. *These studies show issues facilitating denitrification within bioretention and suggest environmental or design modifications could improve removal* (Bratieres et al. 2008; Blecken et al. 2010).

Table 0.2. Reported nutrient and metal reductions in previous literature (field and laboratory). All values presented as % reduction. Reports of export indicate nitrogen leaching that occurred.

Source	NO ₃ ⁻ -N	NO ₂ ⁻ -N	NH ₄ ⁺ -N	TN	TSS	Cu	Pb	Zn
Barrett et al. (2013)				59-79	88-97			
Blecken et al. (2009a)						67-99	96-99	96-99
Blecken et al. (2010)	export		74	export	98			
Bratieres et al. (2008)	export			export	99			
Davis et al. (2003)						99	99	98
Davis et al. (2006)	70			67-72				
Goh et al. (2019)				30-66	75-97			
Hatt, Deletic, et al. (2007)					>80	>90	>90	>90
Henderson et al. (2007)			72-94	63-77				
Hsieh et al. (2007)	export		68					
Hunt et al. (2008)			73	32	60	54	31	77
Luo et al. (2020)	88-97		85-98	77-95				
Mangangka et al. (2015)	39-42	23-65	49-82	39-48	62-81			
Passeport et al. (2009)								
(North Site)		43	78	56				
Passeport et al. (2009)								
(South Site)		1	88	47				
Wang, Zhao, et al. (2017)						>97	>88	>75

Collectively observing TSS (total suspended solids) and metals performance shows largely positive results for removal in bioretention systems. As this is more of a physical process, solids would largely be removed in the upper portions of a bioretention system through filtering (Hunt et al. 2012; Laurenson et al. 2013). Metals could also adhere onto solids and soil microsites which would allow greater removal in the upper portions and significantly lower effluent concentrations. While TSS and metals removal results are positive, the potential for clogging in older bioretention systems is higher. Maintenance of the systems would be needed to ensure consistent filtration throughout the lifespan of a bioretention practice (Davis et al. 2003). In conducting this research, it is expected that TSS and metal removal will follow trends seen in literature and be largely positive.

Hydrology

A review of literature that focused on hydrologic goals of a bioretention practices focused on peak flow and volume reduction. Table 0.3 shows the reported results from a variety of studies that take outlet configuration, soil properties, and environmental conditions into account when testing bioretention systems. Furthermore, the underlying hydrology, meaning the native subsoil infiltration capacity, can also affect the performance of bioretention systems. A variety of field and column studies were analyzed and a main feature affecting hydrology in these systems was rainfall volume and intensity. Larger flows would ultimately render a bioretention system ineffective while smaller flows were captured entirely. The ideal storm size for a system to manage would fall into the category of a 2-year storm (Wang et al. 2019). These results would apply to peak flow reduction as well as peak flow was reduced considerably for smaller storm sizes. The use of IWS was also shown to delay effluent flow.

Table 0.3. Reported peak flow and volume reductions in previous field scale studies. All values presented as % reduction.

Source	Peak Flow Reduction	Volume Reduction
Brown and Hunt (2011) (Deeper)	-	87
Brown and Hunt (2011) (Shallow)	-	75
Davis et al. (2012) (Maryland)	-	77
Davis et al. (2012) (Pennsylvania)	-	52
Davis et al. (2012) (North Carolina)	-	86
Hopkins et al. (2019)	59	-
Hunt et al. (2008)	99	-
Li et al. (2009)	-	20-50
Passeport et al. (2009) (North Site)	18	-
Passeport et al. (2009) (South Site)	14	-
Sun et al. (2019)	-	53
Wang et al. (2019)	-	20-34
Winston et al. (2016)	24-96	36-59

Overall, while peak flow is reduced, effluent flow leaving these systems has a longer processing time (Wright et al. 2018). Drying and wetting periods were also a factor in these studies as drying periods allowed more storage space to build up within a bioretention system and facilitate more volume reduction (Brown and Hunt 2011; Hatt, Fletcher, et al. 2007).

In lab studies a point of concern seemed to be the intensity of application for each rain event. Natural rainfall patterns would have allowed a slower flow into each system and allow more natural flows to occur (Sun et al. 2019). Quickly inundating systems in lab studies limits the volume they are able to manage. Many lab studies do not account for effluent losses due to infiltration and are slightly skewed. Future research should mimic the infiltration process in lab studies to better replicate native hydrologic conditions.

Real-Time Control Schemes:

To advance bioretention design, incorporating smart stormwater systems may be the next innovation in an increasingly connected world (Kerkez, Gruden, et al. 2016). The aim of this research is to examine consequences of using real-time control in bioretention, thus, this section will provide a brief overview of the current use and application of real-time control systems. As the use of this technology is essentially another design alteration for the practice, there are performance implications to examine. Public acceptance and trust in this technology is growing, providing the potential for stormwater systems to be more adaptable to changing environmental conditions.

As previously stated, microbial (e.g., nitrogen) processing requires specific conditions for pollutant removal to occur. Real-time control technologies use sensor readings and/or other data inputs to inform control schemes and facilitate specific conditions or outcomes for a practice

(Middleton and Barrett 2008). For instance, common installations thus far have consisted of wet ponds where control valves hold or release water based on sensor readings and/or incoming weather patterns (Kerkez, Gruden, et al. 2016; Bartos et al. 2018). In the same way, bioretention conditions may be monitored and adjusted to optimize aerobic and anaerobic conditions (allowing sustained microbial function), while also balancing hydrologic goals. In long-term scenarios like drought, these systems have the potential to maintain biological processing by retaining water (Kerkez, Daniels, et al. 2016). With plant and microbial populations supported, this prevents flushing of nutrients when a rain event does finally occur (as has been shown to occur in both lab and field conditions).

Research into real-time control has included using monitoring and sensor driven flow control technologies to study changes in a watershed (Bartos et al. 2018; Mullapudi et al. 2018; Parolari et al. 2018), however long-term consequences on residential areas, water treatment systems, and environmental services are unknown. While few studies have looked into small scale applications of using the technology, improvements in water conservation and distribution were seen in rainwater harvesting systems, and suspended solid loads were reduced in a detention basin (Middleton and Barrett 2008; Roman et al. 2017). Further research into small scale use of real-time control, particularly in green infrastructure, would aid in defining the specific in situ conditions necessary for a performance goal to be realized. This could limit a blanket application of this technology but also highlights the adaptability of these systems to better suit an area.

This research will examine the consequences of using active control in bioretention systems through an examination of water quality, hydrology, and microbial implications. Chapter 1 in this research will broadly compare water quality differences caused by two real-time control bioretention designs and compare them to FD and IWS bioretention practices. Chapter 2 will

investigate the hydrologic differences between real-time control and traditional bioretention. Chapter 3 investigates differences in microbes specifically associated with nitrification and denitrification in real-time control and traditional bioretention designs. These areas of research, while broadly understood that differences will arise, have not yet been studied for bioretention. This will further this practice and inform the use of active control technology in other green infrastructure.

Knowledge Gaps and Contributions

Having reviewed bioretention performance literature and the potential of real-time control, this section will now point out knowledge gaps in literature. These topics necessitate more research and will frame the questions being posed in this proposal. Topics covered will be limited to those pertaining to the use of real-time control in bioretention design as it is the focus of this study, and there is limited research on the implications of using these systems in green infrastructure practices. Over the course of this research, two real-time control systems are compared to traditional bioretention practices to determine the efficacy of implementing the technology. This work is comprised of column and modeling studies that investigate performance through water quality, hydrology, and microbial parameters.

Contribution 1: GI design centers on stormwater management in two main facets; volume reduction, peak flow mitigation, and water quality improvement. Historically, volume reduction and peak flow mitigation have been the main goals, but nutrient reduction is of growing importance. Bioretention has proven to be an adaptable practice that has the potential to improve flow conditions, as well as impact nutrient dynamics. Altering bioretention design promotes water quality improvements (nitrate reductions) through the addition of IWS; however, nitrogen

processing outcomes are not consistent (Table 0.2). FD and IWS are two functional extremes but using sensor technology to control saturation could strike a balance between the two systems and allow aerobic and anaerobic processing to occur. When designing with water quality goals in mind, the hydrologic ramifications are unknown. For example, can the satisfaction of flood mitigation standards still be achieved if retention time is increased? Chapters 1 and 2 explore the use of two real-time control schemes and compare them with FD and IWS practices to understand differences in performance for these systems for both water quality improvement and hydrologic performance. These chapters aim to explore and define real-time control's ability to balance the sometimes-conflicting conditions leading to both water quality and hydrologic improvements. Further, Chapter 1 aims to explore if real-time control can reduce the nitrogen export from bioretention that has been shown in literature.

Contribution 2: Research on the processes and exchanges that occur within a bioretention cell has historically focused on “black box” approaches. Specifically, nutrient processing is measured by differences in influent and effluent water concentrations. There have been few studies that examine denitrifying bacteria as well as factors of influence which include soil moisture, nutrient availability, and vertical stratification. These conditions all play an important role in triggering nitrifying and denitrifying bacteria, and the addition of real-time control technology further complicates such assessments as soil moisture properties in upper and lower portions of a bioretention practice are shifted from natural patterns. Studies exploring the microbial characteristics of real-time control and traditional bioretention practices are conducted in chapters 3. Details regarding population characteristics and functional genes will allow greater insight and improvement in targeting microbial operating conditions, design constraints, and control schemes.

While studies on real-time control performance have focused on hydrologic impacts, nutrient reduction has not been a predominant performance goal. Real-time control systems will have more variable retention times and could trigger more sudden shifts within bioretention cells that could impact nitrogen processing. Understanding the performance of these systems will provide more reassurances when implementing this new bioretention design. With this new information, public understanding of the practice is bolstered and more informed decisions on implementation and design can be made.

References

- ASCE. 2017. "2017 Infrastructure Report Card: A Comprehensive Assessment of America's Infrastructure." In. Reston, VA.
- Barrett, M. E., M. Limouzin, and D. F. Lawler. 2013. 'Effects of Media and Plant Selection on Biofiltration Performance', *Journal of Environmental Engineering-Asce*, 139: 462-70.
- Bartos, M., B. Wong, and B. Kerkez. 2018. 'Open storm: a complete framework for sensing and control of urban watersheds', *Environmental Science-Water Research & Technology*, 4: 346-58.
- BenDor, T. K., V. Shandas, B. Miles, K. Belt, and L. Olander. 2018. 'Ecosystem services and US stormwater planning: An approach for improving urban stormwater decisions', *Environmental Science & Policy*, 88: 92-103.
- Bi, E. G., F. Monette, and J. Gasperi. 2015. 'Analysis of the influence of rainfall variables on urban effluents concentrations and fluxes in wet weather', *Journal of Hydrology*, 523: 320-32.
- Blecken, G. T., Y. Zinger, A. Deletic, T. D. Fletcher, A. Hedstrom, and M. Viklander. 2010. 'Laboratory study on stormwater biofiltration Nutrient and sediment removal in cold temperatures', *Journal of Hydrology*, 394: 507-14.
- Blecken, G. T., Y. Zinger, A. Deletic, T. D. Fletcher, and M. Viklander. 2009a. 'Impact of a submerged zone and a carbon source on heavy metal removal in stormwater biofilters', *Ecological Engineering*, 35: 769-78.
- . 2009b. 'Influence of intermittent wetting and drying conditions on heavy metal removal by stormwater biofilters', *Water Research*, 43: 4590-98.
- Bosch, D. J., M. B. Wagena, A. C. Ross, A. S. Collick, and Z. M. Easton. 2018. 'Meeting Water Quality Goals under Climate Change in Chesapeake Bay Watershed, USA', *Journal of the American Water Resources Association*, 54: 1239-57.

- Brabec, E., S. Schulte, and P. L. Richards. 2002. 'Impervious surfaces and water quality: A review of current literature and its implications for watershed planning', *Journal of Planning Literature*, 16: 499-514.
- Bratieres, K., T. D. Fletcher, A. Deletic, and Y. Zinger. 2008. 'Nutrient and sediment removal by stormwater biofilters: A large-scale design optimisation study', *Water Research*, 42: 3930-40.
- Brown, R. A., and W. F. Hunt. 2011. 'Underdrain Configuration to Enhance Bioretention Exfiltration to Reduce Pollutant Loads', *Journal of Environmental Engineering-Asce*, 137: 1082-91.
- Bullerjahn, G. S., R. M. McKay, T. W. Davis, D. B. Baker, G. L. Boyer, L. V. D'Anglada, G. J. Doucette, J. C. Ho, E. G. Irwin, C. L. Kling, R. M. Kudela, R. Kurmayer, A. M. Michalak, J. D. Ortiz, T. G. Otten, H. W. Paerl, B. Q. Qin, B. L. Sohngen, R. P. Stumpf, P. M. Visser, and S. W. Wilhelm. 2016. 'Global solutions to regional problems: Collecting global expertise to address the problem of harmful cyanobacterial blooms. A Lake Erie case study', *Harmful Algae*, 54: 223-38.
- Cadenasso, M. L., S. T. A. Pickett, P. M. Groffman, L. E. Band, G. S. Brush, M. E. Galvin, J. M. Grove, G. Hagar, V. Marshall, B. P. McGrath, J. P. M. Oneil-Dunne, W. P. Stack, and A. R. Troy. 2008. 'Exchanges across land-water-scape boundaries in urban systems - Strategies for reducing nitrate pollution', *Year in Ecology and Conservation Biology 2008*, 1134: 213-32.
- Chandrasena, G. I., M. Shirdashtzadeh, Y. L. Li, A. Deletic, J. M. Hathaway, and D. T. McCarthy. 2017. 'Retention and survival of E. coli in stormwater biofilters: Role of vegetation, rhizosphere microorganisms and antimicrobial filter media', *Ecological Engineering*, 102: 166-77.
- Chen, H. J., and H. Chang. 2014. 'Response of discharge, TSS, and E. coli to rainfall events in urban, suburban, and rural watersheds', *Environmental Science-Processes & Impacts*, 16: 2313-24.
- Chen, T., Y. P. Liu, B. Zhang, and L. H. Sun. 2019. 'Plant rhizosphere, soil microenvironment, and functional genes in the nitrogen removal process of bioretention', *Environmental Science-Processes & Impacts*, 21: 2070-79.

- Chen, X. L., E. Peltier, B. S. M. Sturm, and C. B. Young. 2013. 'Nitrogen removal and nitrifying and denitrifying bacteria quantification in a stormwater bioretention system', *Water Research*, 47: 1691-700.
- Collins, K. A., T. J. Lawrence, E. K. Stander, R. J. Jontos, S. S. Kaushal, T. A. Newcomer, N. B. Grimm, and M. L. C. Ekberg. 2010. 'Opportunities and challenges for managing nitrogen in urban stormwater: A review and synthesis', *Ecological Engineering*, 36: 1507-19.
- Cook, E. R., C. A. Woodhouse, C. M. Eakin, D. M. Meko, and D. W. Stahle. 2004. 'Long-term aridity changes in the western United States', *Science*, 306: 1015-18.
- CSN. 2013. "Bioretention Illustrated: A Visual Guide for Constructing, Inspecting, Maintaining and Verifying the Bioretention Practice." In. Prince William County, VA: Chesapeake Stormwater Network.
- Dagenais, D., J. Brisson, and T. D. Fletcher. 2018. 'The role of plants in bioretention systems; does the science underpin current guidance?', *Ecological Engineering*, 120: 532-45.
- Davis, A. P., M. Shokouhian, H. Sharma, and C. Minami. 2006. 'Water quality improvement through bioretention media: Nitrogen and phosphorus removal', *Water Environment Research*, 78: 284-93.
- Davis, A. P., M. Shokouhian, H. Sharma, C. Minami, and D. Winogradoff. 2003. 'Water quality improvement through bioretention: Lead, copper, and zinc removal', *Water Environment Research*, 75: 73-82.
- Davis, A. P., R. G. Traver, W. F. Hunt, R. Lee, R. A. Brown, and J. M. Olszewski. 2012. 'Hydrologic Performance of Bioretention Storm-Water Control Measures', *Journal of Hydrologic Engineering*, 17: 604-14.
- Davis, T. W., R. Stumpf, G. S. Bullerjahn, R. M. L. McKay, J. D. Chaffin, T. B. Bridgeman, and C. Winslow. 2019. 'Science meets policy: A framework for determining impairment designation

- criteria for large waterbodies affected by cyanobacterial harmful algal blooms', *Harmful Algae*, 81: 59-64.
- Delpla, I., A. V. Jung, E. Baures, M. Clement, and O. Thomas. 2009. 'Impacts of climate change on surface water quality in relation to drinking water production', *Environment International*, 35: 1225-33.
- Dietz, M. E., and J. C. Clausen. 2005. 'A field evaluation of rain garden flow and pollutant treatment', *Water Air and Soil Pollution*, 167: 123-38.
- Fierer, N., and J. P. Schimel. 2002. 'Effects of drying-rewetting frequency on soil carbon and nitrogen transformations', *Soil Biology & Biochemistry*, 34: 777-87.
- Fletcher, T. D., W. Shuster, W. F. Hunt, R. Ashley, D. Butler, S. Arthur, S. Trowsdale, S. Barraud, A. Semadeni-Davies, J. L. Bertrand-Krajewski, P. S. Mikkelsen, G. Rivard, M. Uhl, D. Dagenais, and M. Viklander. 2015. 'SUDS, LID, BMPs, WSUD and more - The evolution and application of terminology surrounding urban drainage', *Urban Water Journal*, 12: 525-42.
- Gilchrist, S., M. Borst, and E. K. Stander. 2014. 'Factorial Study of Rain Garden Design for Nitrogen Removal', *Journal of Irrigation and Drainage Engineering*, 140: 11.
- Glaister, B. J., T. D. Fletcher, P. L. M. Cook, and B. E. Hatt. 2017. 'Interactions between design, plant growth and the treatment performance of stormwater biofilters', *Ecological Engineering*, 105: 21-31.
- Goh, H. W., K. S. Lem, N. A. Azizan, C. K. Chang, A. Talei, C. S. Leow, and N. A. Zakaria. 2019. 'A review of bioretention components and nutrient removal under different climatesfuture directions for tropics', *Environmental Science and Pollution Research*, 26: 14904-19.
- Hatt, B. E., A. Deletic, and T. D. Fletcher. 2007. 'Stormwater reuse: designing biofiltration systems for reliable treatment', *Water Science and Technology*, 55: 201-09.

- Hatt, B. E., T. D. Fletcher, and A. Deletic. 2007. 'Treatment performance of gravel filter media: Implications for design and application of stormwater infiltration systems', *Water Research*, 41: 2513-24.
- . 2009a. 'Hydrologic and pollutant removal performance of stormwater biofiltration systems at the field scale', *Journal of Hydrology*, 365: 310-21.
- . 2009b. 'Pollutant removal performance of field-scale stormwater biofiltration systems', *Water Science and Technology*, 59: 1567-76.
- Henderson, C., M. Greenway, and I. Phillips. 2007. 'Removal of dissolved nitrogen, phosphorus and carbon from stormwater by biofiltration mesocosms', *Water Science and Technology*, 55: 183-91.
- Hogan, D. M., S. T. Jarnagin, J. V. Loperfido, and K. Van Ness. 2014. 'Mitigating the Effects of Landscape Development on Streams in Urbanizing Watersheds', *Journal of the American Water Resources Association*, 50: 163-78.
- Hopkins, K. G., A. S. Bhaskar, S. A. Woznicki, and R. M. Fanelli. 2019. 'Changes in event-based streamflow magnitude and timing after suburban development with infiltration-based stormwater management', *Hydrological Processes*: 17.
- Houdeshel, C. D., C. A. Pomeroy, and K. R. Hultine. 2012. 'Bioretention Design for Xeric Climates Based on Ecological Principles', *Journal of the American Water Resources Association*, 48: 1178-90.
- Hsieh, C. H., A. P. Davis, and B. A. Needelman. 2007. 'Nitrogen removal from urban stormwater runoff through layered bioretention columns', *Water Environment Research*, 79: 2404-11.
- Hunt, W. F., A. P. Davis, and R. G. Traver. 2012. 'Meeting Hydrologic and Water Quality Goals through Targeted Bioretention Design', *Journal of Environmental Engineering-Asce*, 138: 698-707.

- Hunt, W. F., J. T. Smith, S. J. Jadlocki, J. M. Hathaway, and P. R. Eubanks. 2008. 'Pollutant removal and peak flow mitigation by a bioretention cell in urban Charlotte, NC', *Journal of Environmental Engineering-Asce*, 134: 403-08.
- Igielski, S., B. V. Kjellerup, and A. P. Davis. 2019. 'Understanding urban stormwater denitrification in bioretention internal water storage zones', *Water Environment Research*, 91: 32-44.
- Jay, J. G., M. Tyler-Plog, S. L. Brown, and F. Grothkopp. 2019. 'Nutrient, Metal, and Organics Removal from Stormwater Using a Range of Bioretention Soil Mixtures', *Journal of Environmental Quality*, 48: 493-501.
- Kerkez, B., M. Daniels, S. Graves, V. Chandrasekar, K. Keiser, C. Martin, M. Dye, M. Maskey, and F. Vernon. 2016. 'Cloud Hosted Real-time Data Services for the Geosciences (CHORDS)', *Geoscience Data Journal*, 3: 4-8.
- Kerkez, B., C. Gruden, M. Lewis, L. Montestruque, M. Quigley, B. Wong, A. Bedig, R. Kertesz, T. Braun, O. Cadwalader, A. Poresky, and C. Pak. 2016. 'Smarter Stormwater Systems', *Environmental Science & Technology*, 50: 7267-73.
- Kobielska, P. A., A. J. Howarth, O. K. Farha, and S. Nayak. 2018. 'Metal-organic frameworks for heavy metal removal from water', *Coordination Chemistry Reviews*, 358: 92-107.
- Konrad, Christopher P. 2003. "Effects of Urban Development on Floods." In.: USGS.
- Laurenson, G., S. Laurenson, N. Bolan, S. Beecham, and I. Clark. 2013. 'The Role of Bioretention Systems in the Treatment of Stormwater.' in D. L. Sparks (ed.), *Advances in Agronomy, Vol 120* (Elsevier Academic Press Inc: San Diego).
- Le Coustumer, S., T. D. Fletcher, A. Deletic, S. Barraud, and P. Poelsma. 2012. 'The influence of design parameters on clogging of stormwater biofilters: A large-scale column study', *Water Research*, 46: 6743-52.

- Levin, L. A., and A. S. Mehring. 2015. 'Optimization of bioretention systems through application of ecological theory', *Wiley Interdisciplinary Reviews-Water*, 2: 259-70.
- Li, C. L., M. Liu, Y. M. Hu, M. Zong, M. H. Zhao, and M. T. Walter. 2019. 'Characteristics of impervious surface and its effect on direct runoff: a case study in a rapidly urbanized area', *Water Science and Technology-Water Supply*, 19: 1885-91.
- Li, H., L. J. Sharkey, W. F. Hunt, and A. P. Davis. 2009. 'Mitigation of Impervious Surface Hydrology Using Bioretention in North Carolina and Maryland', *Journal of Hydrologic Engineering*, 14: 407-15.
- Li, Y. L., A. Deletic, and D. T. McCarthy. 2014. 'Removal of E.coli from urban stormwater using antimicrobial-modified filter media', *Journal of Hazardous Materials*, 271: 73-81.
- Liu, A., P. Egodawatta, Y. T. Guan, and A. Goonetilleke. 2013. 'Influence of rainfall and catchment characteristics on urban stormwater quality', *Science of the Total Environment*, 444: 255-62.
- Liu, J., D. J. Sample, C. Bell, and Y. T. Guan. 2014. 'Review and Research Needs of Bioretention Used for the Treatment of Urban Stormwater', *Water*, 6: 1069-99.
- Lovley, D. R., D. E. Holmes, and K. P. Nevin. 2004. 'Dissimilatory Fe(III) and Mn(IV) reduction.' in R. K. Poole (ed.), *Advances in Microbial Physiology, Vol. 49* (Academic Press Ltd-Elsevier Science Ltd: London).
- Luo, Y. H., X. P. Yue, Y. Q. Duan, A. J. Zhou, Y. J. Gao, and X. Zhang. 2020. 'A bilayer media bioretention system for enhanced nitrogen removal from road runoff', *Science of the Total Environment*, 705: 8.
- Madigan, M., Bender, K., Buckley, D., Sattley, W., Stahl, D. . 2019. *Brock Biology of Microorganisms (15th Global Edition ed.)* (Pearson).

- Mangangka, I. R., A. Liu, P. Egodawatta, and A. Goonetilleke. 2015. 'Performance characterisation of a stormwater treatment bioretention basin', *Journal of Environmental Management*, 150: 173-78.
- Meerow, S., and J. P. Newell. 2017. 'Spatial planning for multifunctional green infrastructure: Growing resilience in Detroit', *Landscape and Urban Planning*, 159: 62-75.
- Middleton, J. R., and M. E. Barrett. 2008. 'Water quality performance of a batch-type stormwater detention basin', *Water Environment Research*, 80: 172-78.
- Mishra, A. K., and V. P. Singh. 2010. 'A review of drought concepts', *Journal of Hydrology*, 391: 204-16.
- Morse, N., E. Payne, R. Henry, B. Hatt, G. Chandrasena, J. Shapleigh, P. Cook, S. Coutts, J. Hathaway, M. T. Walter, and D. McCarthy. 2018. 'Plant-Microbe Interactions Drive Denitrification Rates, Dissolved Nitrogen Removal, and the Abundance of Denitrification Genes in Stormwater Control Measures', *Environmental Science & Technology*, 52: 9320-29.
- Mullapudi, A., M. Bartos, B. Wong, and B. Kerkez. 2018. 'Shaping Streamflow Using a Real-Time Stormwater Control Network', *Sensors*, 18: 11.
- O’Gorman, Paul A. 2015. 'Precipitation Extremes Under Climate Change', *Current Climate Change Reports*, 1: 49-59.
- Parolari, A. J., S. Pelrine, and M. S. Bartlett. 2018. 'Stochastic water balance dynamics of passive and controlled stormwater basins', *Advances in Water Resources*, 122: 328-39.
- Passeport, E., W. F. Hunt, D. E. Line, R. A. Smith, and R. A. Brown. 2009. 'Field Study of the Ability of Two Grassed Bioretention Cells to Reduce Storm-Water Runoff Pollution', *Journal of Irrigation and Drainage Engineering*, 135: 505-10.

- Payne, E. G. I., T. D. Fletcher, P. L. M. Cook, A. Deletic, and B. E. Hatt. 2014. 'Processes and Drivers of Nitrogen Removal in Stormwater Biofiltration', *Critical Reviews in Environmental Science and Technology*, 44: 796-846.
- Peralta, A. L., S. Ludmer, and A. D. Kent. 2013. 'Hydrologic history influences microbial community composition and nitrogen cycling under experimental drying/wetting treatments', *Soil Biology & Biochemistry*, 66: 29-37.
- Placella, S. A., E. L. Brodie, and M. K. Firestone. 2012. 'Rainfall-induced carbon dioxide pulses result from sequential resuscitation of phylogenetically clustered microbial groups', *Proceedings of the National Academy of Sciences of the United States of America*, 109: 10931-36.
- Read, J., T. D. Fletcher, T. Wevill, and A. Deletic. 2010. 'Plant Traits that Enhance Pollutant Removal from Stormwater in Biofiltration Systems', *International Journal of Phytoremediation*, 12: 34-53.
- Read, J., T. Wevill, T. Fletcher, and A. Deletic. 2008. 'Variation among plant species on pollutant removal from stormwater non biofiltration systems', *Water Research*, 42: 893-902.
- Roman, D., A. Braga, N. Shetty, and P. Culligan. 2017. 'Design and Modeling of an Adaptively Controlled Rainwater Harvesting System', *Water*, 9: 14.
- Salazar, A., B. N. Sulman, and J. S. Dukes. 2018. 'Microbial dormancy promotes microbial biomass and respiration across pulses of drying-wetting stress', *Soil Biology & Biochemistry*, 116: 237-44.
- Schimel, J. P. 2018. 'Life in Dry Soils: Effects of Drought on Soil Microbial Communities and Processes.' in D. J. Futuyma (ed.), *Annual Review of Ecology, Evolution, and Systematics*, Vol 49 (Annual Reviews: Palo Alto).
- Steffen, M. M., T. W. Davis, R. M. L. McKay, G. S. Bullerjahn, L. E. Krausfeldt, J. M. A. Stough, M. L. Neitzey, N. E. Gilbert, G. L. Boyer, T. H. Johengen, D. C. Gossiaux, A. M. Burtner, D. Palladino, M. D. Rowe, G. J. Dick, K. A. Meyer, S. Levy, B. E. Boone, R. P. Stumpf, T. T. Wynne, P. V.

- Zimba, D. Gutierrez, and S. W. Wilhelm. 2017. 'Ecophysiological Examination of the Lake Erie Microcystis Bloom in 2014: Linkages between Biology and the Water Supply Shutdown of Toledo, OH', *Environmental Science & Technology*, 51: 6745-55.
- Sukenik, A., O. Hadas, A. Kaplan, and A. Quesada. 2012. 'Invasion of Nostocales (cyanobacteria) to subtropical and temperate freshwater lakes - physiological, regional, and global driving forces', *Frontiers in Microbiology*, 3: 9.
- Sun, Y. W., C. Pomeroy, Q. Y. Li, and C. D. Xu. 2019. 'Impacts of rainfall and catchment characteristics on bioretention cell performance', *Water Science and Engineering*, 12: 98-107.
- Tauhid, F. A., and H. Zawani. 2018. 'Mitigating Climate Change Related Floods in Urban Poor Areas: Green Infrastructure Approach', *Journal of Regional and City Planning*, 29: 98-112.
- TDEC. 2014. "Tennessee Permanent Stormwater Management and Design Guidance Manual." In, 141-76.
- Thompson, A. M., A. C. Paul, and N. J. Balster. 2008. 'Physical and hydraulic properties of engineered soil media for bioretention basins', *Transactions of the Asabe*, 51: 499-514.
- TSM, Tennessee Stormwater Management. 2015. "Management and Design Guidance Manual." In *Tennessee Stormwater Management*.
- USEPA. 2002. "Federal Water Pollution Control Act." In, edited by USEPA. Washington, D.C.
- . 2009. 'TMDL Program Results Analysis Fact Sheet', USEPA Office of Water.
- . 2010. "TMDL Program Results Analysis Fact Sheet." In. Washington, DC: EPA Office of Water.
- . 2015. "Central Tenets of the National Pollutant Discharge Elimination System (NPDES) Permitting Program." In, edited by USEPA. Washington, D.C.

- USEPA, United States Environmental Protection Agency. 2016. "National Summary of State Information."
In *USEPA*.
- Vangestel, M., R. Merckx, and K. Vlassak. 1993. 'MICROBIAL BIOMASS RESPONSES TO SOIL DRYING AND REWETTING - THE FATE OF FAST-GROWING AND SLOW-GROWING MICROORGANISMS IN SOILS FROM DIFFERENT CLIMATES', *Soil Biology & Biochemistry*, 25: 109-23.
- Waller, L. J., G. K. Evanylo, L. A. H. Krometis, M. S. Strickland, T. Wynn-Thompson, and B. D. Badgley. 2018. 'Engineered and Environmental Controls of Microbial Denitrification in Established Bioretention Cells', *Environmental Science & Technology*, 52: 5358-66.
- Wang, J., L. H. C. Chua, and P. Shanahan. 2017. 'Evaluation of pollutant removal efficiency of a bioretention basin and implications for stormwater management in tropical cities', *Environmental Science-Water Research & Technology*, 3: 78-91.
- Wang, J. L., Y. L. Zhao, L. Q. Yang, N. N. Tu, G. P. Xi, and X. Fang. 2017. 'Removal of Heavy Metals from Urban Stormwater Runoff Using Bioretention Media Mix', *Water*, 9: 22.
- Wang, M., D. Q. Zhang, S. W. Lou, Q. H. Hou, Y. J. Liu, Y. N. Cheng, J. D. Qi, and S. K. Tan. 2019. 'Assessing Hydrological Effects of Bioretention Cells for Urban Stormwater Runoff in Response to Climatic Changes', *Water*, 11: 20.
- Willems, P. 2013. 'Revision of urban drainage design rules after assessment of climate change impacts on precipitation extremes at Uccle, Belgium', *Journal of Hydrology*, 496: 166-77.
- Winston, R. J., J. D. Dorsey, and W. F. Hunt. 2016. 'Quantifying volume reduction and peak flow mitigation for three bioretention cells in clay soils in northeast Ohio', *Science of the Total Environment*, 553: 83-95.

- Wright, O. M., E. Istanbuluoglu, R. R. Horner, C. L. DeGasperi, and J. Simmonds. 2018. 'Is there a limit to bioretention effectiveness? Evaluation of stormwater bioretention treatment using a lumped urban ecohydrologic model and ecologically based design criteria', *Hydrological Processes*, 32: 2318-34.
- Yan, Q., B. R. James, and A. P. Davis. 2017. 'Lab-Scale Column Studies for Enhanced Phosphorus Sorption from Synthetic Urban Stormwater Using Modified Bioretention Media', *Journal of Environmental Engineering*, 143: 13.
- Yang, Y., and T. F. M. Chui. 2018. 'Optimizing surface and contributing areas of bioretention cells for stormwater runoff quality and quantity management', *Journal of Environmental Management*, 206: 1090-103.
- Zhang, J. Y., R. P. Singh, Y. Z. Liu, and D. F. Fu. 2019. 'Design and operation of submerged layer in bioretention for enhanced nitrate removal', *Journal of Water Supply Research and Technology-Aqua*, 68: 744-56.
- Zumft, W. G. 1997. 'Cell biology and molecular basis of denitrification', *Microbiology and Molecular Biology Reviews*, 61: 533-+.

**NOTE: The following manuscript has been published in Blue-Green Systems in August
2019.**

Chapter 1 : Real Time Control Schemes for Improving Water Quality

From Bioretention Cells

P.P. Persaud¹, A. A. Akin¹, B. Kerkez², D.T. McCarthy³, and J.M. Hathaway¹

¹ Dept. of Civil and Environmental Engineering, University of Tennessee, 325 John D. Tickle Building, 851 Neyland Dr., Knoxville, TN, 37996 USA

² Dept. of Civil and Environmental Engineering, University of Michigan, Ann Arbor, MI, USA

³ Environmental and Public Health Microbiology Lab (EPHM Lab), Dept. of Civil Engineering, Monash University, Melbourne, VIC, AUS

*Corresponding author: hathaway@utk.edu

Abstract

Extreme weather and the proliferation of impervious areas in urban watersheds increases the frequency of flood events and deepens water quality concerns. Bioretention is a type of green infrastructure practice developed to mitigate these impacts by reducing peak flows, runoff volume, and nutrient loads in stormwater. However, studies have shown inconsistency in the ability of bioretention to manage some pollutants, particularly some forms of nitrogen. Innovative sensor and control technologies are being tested to actively manage urban stormwater, primarily in open water stormwater systems such as wet ponds. Through these cyber-physical controls, it may be possible to optimize storage time and/or soil moisture dynamics within bioretention cells to create more favorable conditions for water quality improvements. A column study testing the influence of active control on bioretention system performance was conducted over a nine-week period. Active control columns were regulated based on either maintaining a specific water level or soil moisture content and were compared to free draining and internal water storage standards. Actively controlled bioretention columns performed similarly, with the soil moisture-based control showing the best performance with over 86% removal of metals and TSS while also exhibiting the highest

ammonium removal (43%) and second highest nitrate removal (74%). While all column types showed mostly similar TSS and metal removal trends (median 94 and 98%, respectively), traditionally free draining and internal water storage configurations promoted aerobic and anaerobic processes, respectively, which suggests that actively controlled systems have greater potential for targeting both processes. The results suggest that active controls can improve upon standard bioretention designs, but further optimization is required to balance the water quality benefits gained by retention time against storage needs for impending storms.

Keywords: stormwater, bioretention, biofilter, real time control, water quality

Introduction

Degradation of urban waterways has caused poor water quality and a decline in ecosystem services worldwide. Stormwater is one major source of impairment for urban systems, leading watershed managers to seek mitigation strategies (USEPA, 2016). As such, the use of green infrastructure (a principal component of Water Sensitive Urban Design) has become more prevalent for treating stormwater runoff before its release into larger stream systems due to its holistic social, ecological, and hydrological benefits (Fletcher et al., 2015; Larsen et al., 2016). In particular, bioretention cells (also known as bioretention areas or biofilters) have shown promise for reducing the effects of stormwater pollution on urban waterways. Bioretention cells are designed to replicate natural environmental processing. Using permeable soil media and native plant species, they incorporate infiltration and various pollution removal mechanisms to reduce both the volume and pollutant concentrations in stormwater runoff. Bioretention practices have shown the ability to significantly reduce nutrient, metal, and pathogenic bacteria concentrations in

urban runoff. (Henderson et al., 2007; Hunt et al., 2008; Hatt et al., 2009; Hathaway & Hunt, 2012).

The interactions between media, plants, and microbes are a primary source of research in literature when seeking to understand bioretention function (Hathaway & Hunt, 2012; Glaister et al., 2017; Yan et al., 2017). Variations in bioretention design have been used to optimize functionality of these systems by changing these interactions. Free draining (FD) bioretention systems, a common first-generation design, have shown good success, but often are dominated by aerobic conditions and lack the ability to consistently perform both nitrification and denitrification (other than in internal microsites)(Hathaway & Hunt, 2012; Laurenson et al., 2013; Tang & Tian, 2016; McPhillips et al., 2018). Implementation of internal water storage (IWS) zones have been used in an attempt to allow both aerobic and anaerobic environments to promote nitrification and denitrification processes, however, there is the potential for lost storage capacity with these systems depending on the underlying soil infiltration rate (Dietz & Clausen, 2005; Li et al., 2014; Waller et al., 2018). Both designs, while generally successful, are static. That is, they cannot adapt to changing conditions, or switch between storing and releasing water to optimize runoff reduction and water quality performance.

The optimal operation of bioretention systems is largely site-specific because no two areas are under the same environmental stressors. Further, extreme weather events may necessitate occasional deviations in operation to accommodate large volumes of water. Active control systems have been recently studied to manage stormwater systems using networks of valves and sensors (Mullapudi et al., 2017; Mullapudi et al., 2018). They direct stormwater flows in and out of a watershed network to mitigate flood effects in cities (Parolari et al., 2018), however, research has typically focused on ponds and other storage systems. The effects of active controls on bioretention

cells have only recently been considered for the purposes of water harvesting with a focus on indicator bacteria reduction (Shen et al., submitted). Using active control systems to optimize bioretention function has the potential to allow consideration of sometimes conflicting objectives, but also creates a dynamic environment for soils, plants, and microbes that is unexplored in this cyber-physical context and may have unintended consequences.

Adding active control to the already dynamic bioretention environment poses some challenges, but also provides opportunities. Drying and wetting cycles are an unavoidable and highly influential component of bioretention cell function and can lead to inefficiencies in performance (Manka et al., 2016). Drying periods affect soil structure and biological processes which can lead to metals export, microbial dormancy, and lowered water holding capacity of a soil (Blecken et al., 2009; Laurenson et al., 2013). When dry periods end and a storm event channels stormwater into a treatment area, a flushing effect is observed. Drying periods cause mineralization and exposure of previously unavailable organic matter in soils which cannot be sufficiently processed by microbes due to their inactivity in dry periods; although some microbial communities have developed resistances to dry climate conditions (Zhou et al., 2016; Salazar et al., 2018). The result is a nutrient export once wet conditions arise (Vangestel et al., 1993; Pulleman & Tietema, 1999). Drying and wetting cycles also alter soil respiration rates which can increase or decrease depending on soil type (Fierer & Schimel, 2002). Lowered soil respiration occurs once a bioretention media is constantly inundated with moisture which prohibits plant root access to oxygen or facilitation of nutrient uptake, and leads to die off (Colmer, 2003; Payne et al., 2014). This further reduces the efficiency of bioretention areas. Active controls can both exacerbate these effects, for instance if water is released based on forecasted rainfall that doesn't occur or can

improve these conditions if the outlet is managed to maintain a more consistent soil moisture regime.

This research aims to improve the understanding of how the performance of bioretention systems can be improved using active control to regulate operating conditions. This work highlights the benefits and tradeoffs of active controls in comparison to traditional bioretention designs. By comparing two standard passive bioretention designs to two active control strategies, the objectives are to (1) quantify and examine metal and nutrient removal from the four treatments, and (2) investigate and compare the performance of the two active control schemes.

Methods

Bioretention Column Design

The experiment was designed to mimic traditional operational conditions of bioretention in the United States whereby impermeable liners are uncommon, thus captured runoff is allowed to exfiltrate the system (i.e. seepage) at a rate consistent with the in-situ soil infiltration capacity. The columns were constructed using 30 cm diameter gray PVC with both a small valve to mimic seepage (seepage outlet) and an underdrain on the bottom of each column to allow drainage. Seepage outlets were adjusted to mimic an infiltration rate of 0.20 cm/hr (in the range of a clay soil type) and were frequently maintained to avoid biological fouling. The interior of each column was sanded to minimize the effects of preferential flow. Columns were filled using layers of gravel (washed #57 stone), washed pea gravel, sand, bioretention media and mulch with a 10-centimeter ponding zone (Figure 1.1). The composition of the bioretention media was 85-88% sand, 10% clay/fines, and 2-3% organic matter, consistent with design suggestions in the United States for

Tennessee and North Carolina (TSM, 2015; NCDEQ, 2018). Each column contained one *Echinacea purpurea* (purple coneflower) and one *Juncus effuses* (common rush). Bioretention columns were kept in a climate-controlled greenhouse where temperatures were maintained at seasonal averages (15-27°C).

Four outlet configurations were tested with five replicates being used for each configuration, a total of 20 columns (Figure 1.1). Configuration one was traditional free drainage (FD), where the underdrain provided unobstructed drainage from the column (i.e. drained via gravity). The second configuration was internal water storage (IWS, also known as a Submerged Zone), where a submerged zone of 45 centimeters was present in the bottom of the column and regulated by an upturned elbow in the piping. The remaining two configurations were actively managed using automated, remotely controlled, ball valves based on two experimental active control schemes. Both configurations relied on historic rainfall data and historic rainfall predictions as described in the Experimental Procedure.

For configuration three (SM, Soil Moisture), the system valve was opened and closed as needed to maintain, to the degree possible, field capacity in the column soils based on real time monitoring data and rainfall predictions (See Monitoring Description below). Field capacity was used as a target in this study because it is the optimal moisture level to facilitate microbial activity (Barros et al., 1995). Finally, configuration four (VC, Volume Control) involved use of a level controller to maintain water storage levels at 30 centimeters based on continuous monitoring and rainfall predictions as further described in the Experimental Procedure. A lower water storage level allowed the opportunity to test the ability of the active control to achieve similar water quality performance to IWS despite the smaller internal storage depth.

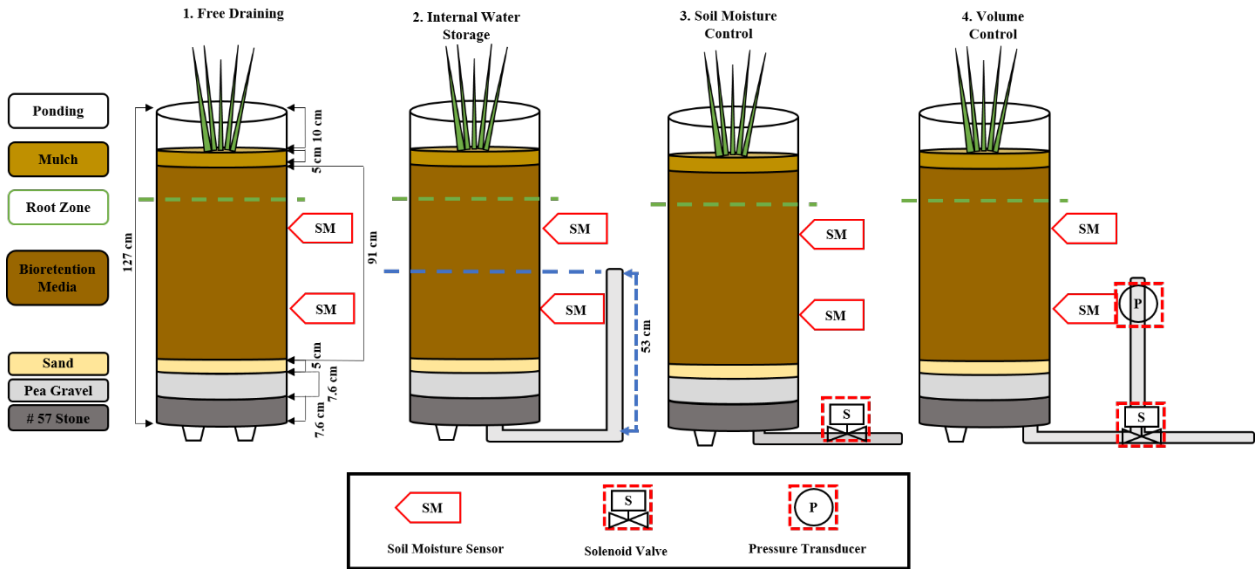


Figure 1.1 Column design configurations

Bioretention Column Monitoring

Decagon GS1 soil moisture sensors were buried in each column at depths of 30.5 and 61 cm from the top of the media. The sensors were calibrated in the Department of Civil and Environmental Engineering hydraulics laboratory by incrementally saturating a known volume of bioretention media and recording raw sensor readings (a method consistent with manufacturer suggestions for calibration). Water storage levels were measured in the fourth configuration (VC) using a Stevens pressure transducer. Error estimates for the soil moisture sensors and pressure transducers were $\pm 0.03 \text{ m}^3/\text{m}^3$ and $\pm 0.02\%$ respectively. Continuous monitoring for each column were stored on an InfluxDB database and visualized using Grafana, including soil moisture readings, pressure transducer depths (configurations 3 and 4), and when active control valves opened or closed. Active control was achieved using photon microcontrollers to trigger valves to drain or retain water consistent with the corresponding management scheme.

Weather Data

Nine weeks of rainfall data recorded by the National Oceanic and Atmospheric Administration (NOAA) from June to July 2017, in Knoxville TN, were mimicked in this study, that is, were used to inform the number and size of applications to the columns. Although this study was carried out in the autumn of 2018, data for the months of June and July 2017 were used due to high density and variety of precipitation events observed over that period. A total of 18 events occurred during this period ranging from 0.18 to 3.81 cm, with a median size of 0.56 cm. In addition, the precipitation forecast preceding each storm event (at 12 hours before a given event) was obtained to inform the active control treatments (configurations SM and VC). These historic quantitative precipitation forecasts and events for 2017 were obtained from the National Oceanic

and Atmospheric Administration (NOAA). The weather station at the McGhee-Tyson airport in Alcoa, Tennessee, was used as a reference location when obtaining forecast data.

Experimental Procedure

Pre-Event

During each day of the study, weather predictions for the next day of the rainfall time series were observed to determine if a rain event was projected to occur. If so, the predicted rainfall depth was sent out that night via wireless communication to signal the release, if necessary, of stored water from actively controlled columns in accordance with their respective schemes. For treatment three, the runoff produced as a result of the predicted rainfall was quantified and considered along with the current soil moisture conditions at the 30 cm sensor. The amount of predicted runoff that could be captured given the existing soil moisture, without exceeding field capacity, was calculated and any amount in excess of this value was preemptively released from the valve to provide the necessary additional storage. Drainage from the system was still possible despite the system being at field capacity as (1) our measurement of field capacity was likely an overestimate as it was calculated in a laboratory setting (Kirkham, 2005), (2) water was released from deeper in the profile where water was stored in places such as the gravel layer, and (3) opening of the drainage port created a new equilibrium in the system. For treatment four, the amount of predicted runoff that could be captured without exceeding the targeted internal water storage depth was determined, and any excess amount was preemptively released to make room for the predicted event. The influence of weather uncertainty in the control scheme meant that a predicted storm event did not always occur even though the valve opened and released water in preparation. In the same respect,

storm events sometimes occurred when there was no forecasted event. Although this type of forecast error added complexity to the experimental method, it was necessary to realistically reflect the function of actively controlled bioretention which are subject to weather uncertainty.

During Event

During the event, columns three and four were actively managed to maintain targeted conditions. For instance, during an event for treatment three, once soil moisture readings exceeded field capacity the active control valve drained until field capacity was reached. Likewise, for treatment four, the column was triggered to open as needed to maintain the 30-cm depth. These schemes thus provided both a preemptive and adaptive control to manage internal conditions.

Stormwater Application

Storm events smaller than 1mm were excluded from this study as runoff would not be produced from a typical urban catchment for these storms (Guo & Adams, 1998; Le Coustumer et al., 2012). Previous work researching bioretention systems have used local climate data to determine dosing volumes. Chandrasena et al. (2017) reports using a storm size of 5.75 mm per event while Glaister et al. (2017) and Morse et al. (2018) used a local yearly average of 540 mm. Each storm event was applied based on 20:1 sizing ratio for each column (TSM, 2015). Columns were dosed with synthetic stormwater following procedures outlined by Bratieres et al. (2008). In short, tap water in the greenhouse was used to make the stormwater mixture, which was supplemented with various chemicals to meet, to the extent possible, target concentrations shown in Table 1.1. Sediment added to the stormwater mixture was collected from a local concrete-lined detention pond and sieved to 300 μm to remove larger particles and meet a target total suspended solids (TSS) concentration of 150 ppm. The nutrient contributions from the sediment were

analyzed prior to mixing and were considered when preparing the final mixture. The stormwater mixture was continuously and vigorously mixed as columns were dosed with the prescribed amount of stormwater for a given event. Each dose was applied in three passes to ensure consistency in the stormwater concentrations received by each column. In the event that a column reached capacity, as evidenced by the column filling to the top and no longer receding, the application was ceased.

Water Quality Sampling

Water samples of column discharge were collected 24 hours after each event, allowing completion of free drainage. An initial water quality sample of the inflow was also taken when semi-artificial stormwater was applied. Because rainfall predictions signaled opening of active control valves in preparation for anticipated rainfall events, samples were also occasionally collected of column discharge due to predicted precipitation that did not occur. That is, the columns were actively controlled and discharged for an impending event that did not happen. Samples were analyzed for TSS using standard methods (SM 2540 D), for nutrients ($\text{NO}_2\text{-N}$, $\text{NO}_3\text{-N}$, and $\text{NH}_4^+\text{-N}$) using ion chromatography, and for dissolved metals (Cu^{2+} , Zn^{2+} , and Mn^{2+}) using inductively coupled plasma mass spectrometry (ICP-MS). Prior to sampling for nutrients and metals, samples were filtered through 0.45 μm Whatman disposable filters. They were also acidified using a 1% dilution with concentrated nitric acid prior to ICP-MS analysis. Samples were stored in refrigeration after filtration awaiting analysis.

Table 1.1 Sediment contributions and stormwater target concentrations

Constituent	Sediment Contribution (mg/L)	Target Concentration (mg/L)
NO_x - N	0.01879	0.75
NH₄⁺-N	0.00335	0.27
TDP	0.002	0.04
Cu²⁺	0.0055	0.05
Zn²⁺	0.0043	0.25
Pb²⁺	0.0045	0.14
Cr⁶⁺	0.0026	0.025
Mn²⁺	0.0012	0.25
Fe³⁺	0.0151	1
Ni²⁺	0.0003	0.03
Cd²⁺	0.0006	0.0045

Statistical Analysis

Statistical analysis for this research was conducted using MATLAB R2018a. Percent reduction for each pollutant was calculated by subtracting the outflow from inflow Event Mean Concentration (EMC) then dividing by the inflow EMC. First, a Kolmogorov-Smirnov test was used to confirm the presence or absence of normality on raw data. Data was found to be non-normally distributed, necessitating non-parametric statistical analysis. The Wilcoxon signed-rank test was used to determine statistical differences among the treatments and antecedent rainfall effects on water quality were measured using a Spearman's Rank correlation coefficient. A 0.05 significance level was used to indicate statistical significance.

Results and Discussion

Soil Moisture and Active Control

To explain the water quality results from this study, an understanding of how each treatment affects system hydrology is necessary. In particular, soil moisture dynamics are critical to biogeochemical processes in these systems. As noted above, soil moisture readings collected throughout this study were taken at 30 and 60 centimeters below the surface of the bioretention media. Field capacity of the bioretention media was measured to be 28% (v/v) and was used as the marker for active control in the SM treatment. The readings for one storm event are shown in Figure 1.2 while the average readings for each storm event are shown in Figure 1.3 and Figure 1.4, which highlight trends in treatment types. The period for each storm (for the sake of soil moisture summary statistics) was defined as the 24 hours following the start of each storm event. As expected, the IWS treatment has a higher soil moisture content than the other treatments, in particular for the deeper sensor, while the FD was the driest system at the shallow (30 cm) reading.

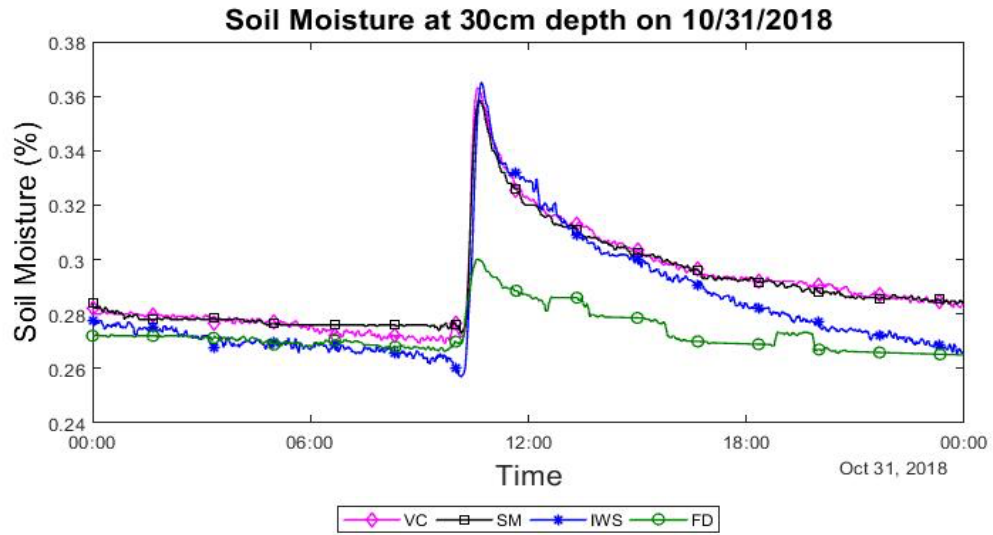


Figure 1.2 Soil moisture at 30-cm depth for storm on 10/31/2018

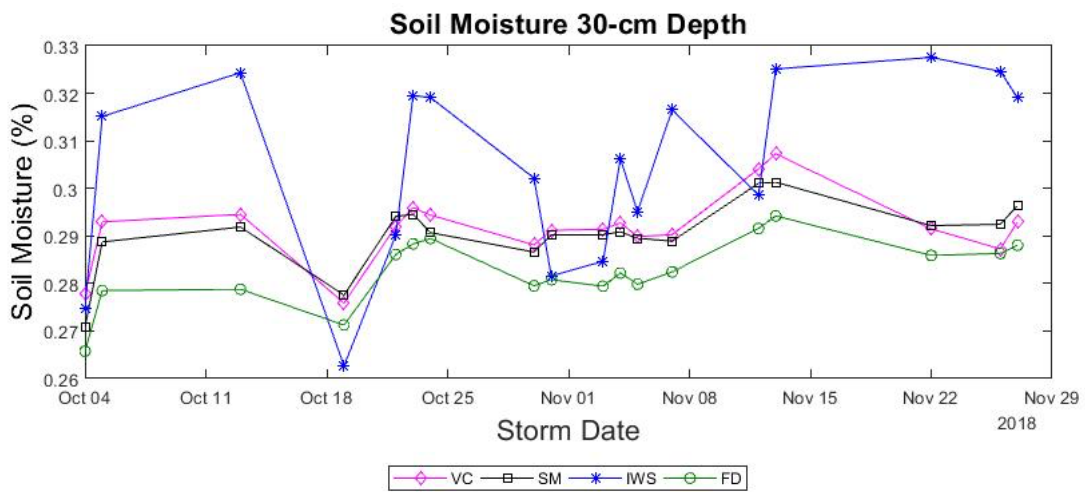


Figure 1.3 Average soil moisture at 30-cm depth for each storm. Storms are defined as the 24-hr period following a storm event

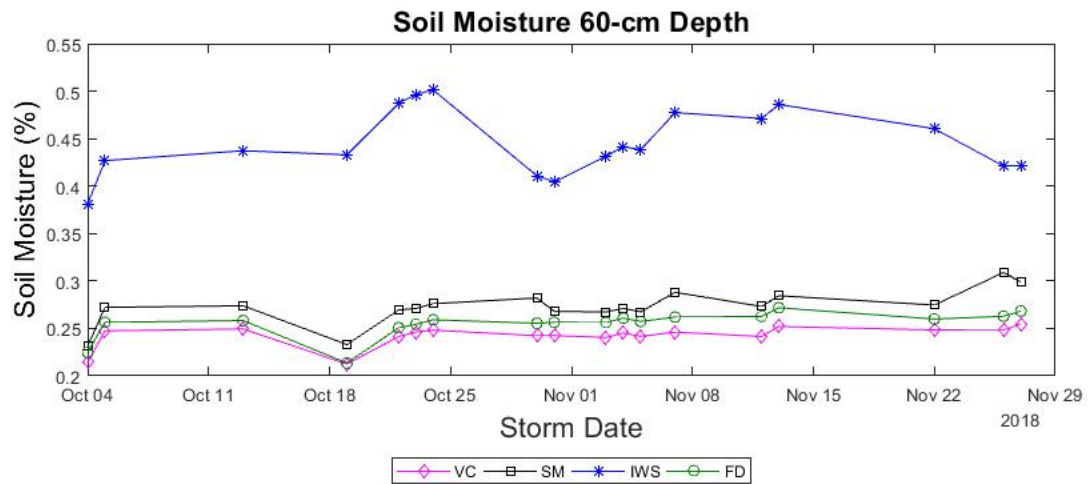


Figure 1.4 Average soil moisture at 60-cm depth for each storm. Storms are defined as the 24-hr period following a storm event

This is a result IWS creating internal storage and promoting wetter conditions, while FD being freely drained and retaining less moisture in the upper soil profile. Comparable soil moisture patterns were observed for the active control treatments. At the 30 cm sensor, both active control treatments operated between IWS and FD, while at the 60 cm depth, VC, SM, and FD all showed similar soil moisture readings and patterns. SM was slightly more wet than FD, while VC was slightly drier than FD at the deeper depth.

The difference in control scheme between VC and SM treatments was in the operation of the solenoid valve to store or release water. More sporadic open and close cycles were seen for the VC treatments while SM treatments exhibited a more stable open and close cycle for each storm event (over the course of the study opening an average of 693 and 50 times respectively). Because VC treatments were based on a target storage depth within the column, collection and reaction times between pressure transducer readings and solenoid valves to maintain a 30-centimeter storage depth caused more frequent opening and closing of the solenoid valve. This could be corrected in future studies by allowing depths ranging from 28 to 32 cm, for instance. On the other hand, the SM treatments required the maintenance of a specific soil moisture reading. Soil moisture sensors would trigger release only when field capacity was exceeded. The collection and reaction timing were slower, and solenoids were open and closed for longer periods of time while soil moisture changes occurred at the 30-cm sensor. Essentially, once the solenoid was open, there was a delay in soil moisture changes as water percolated out of the system.

TSS

TSS removal for all treatments was above 97% (Table 1.2), which is unsurprising given that this parameter is typically removed by upper soil layers which are generally not influenced by

Table 1.2 Event Mean Concentration (EMC), Median Concentration, Standard Deviation (Std Dev) and Relative Standard Deviation (RSD) for each Treatment Type

Pollutant	Configuration	EMC (mg/L)	Median (mg/L)	Reduction %	Std Dev	RSD %
Cu²⁺	VC ^a	0.012	0.009	64.9	0.007	60.0
	SM ^b	0.004	0.004	86.6	0.001	20.7
	IWS ^c	0.003	0.003	90.4	0.001	23.6
	FD ^b	0.004	0.004	87.2	0.001	18.3
Mn²⁺	VC ^a	0.008	0.005	95.2	0.008	96.3
	SM ^a	0.009	0.007	94.8	0.008	85.5
	IWS ^b	1.961	1.984	-995.6	0.445	22.7
	FD ^a	0.011	0.006	93.5	0.013	113.7
Zn²⁺	VC ^a	0.010	0.006	95.3	0.007	69.1
	SM ^b	0.015	0.012	93.1	0.009	60.1
	IWS ^a	0.013	0.004	94.5	0.025	183.9
	FD ^b	0.014	0.011	93.5	0.007	50.9
NH₄⁺-N	VC ^a	0.008	0.007	41.2	0.004	46.6
	SM ^b	0.013	0.009	43	0.013	107.4
	IWS ^a	0.026	0.028	26.3	0.017	65.0
	FD ^b	0.010	0.011	39.1	0.004	42.5
NO₂⁻-N	VC ^a	0.087	0.041	-19.9	0.167	192.6
	SM	0.100	0.046	-18.6	0.196	196.0
	IWS ^b	0.069	0.049	-87.9	0.102	146.9
	FD ^a	0.062	0.046	-14.7	0.116	185.9
NO₃⁻-N	VC ^a	0.824	0.743	73.6	0.545	66.2
	SM ^b	0.759	0.525	74.3	0.621	81.8
	IWS ^c	0.138	0.096	95.6	0.206	149.6
	FD ^d	1.007	0.977	67.2	0.524	52.1
TSS	VC	2.681	1.6	97.4	2.930	109.3
	SM	2.913	1.1	97	3.929	134.9
	IWS	1.940	1.2	98.2	2.353	121.3
	FD	1.749	1.7	98.1	2.985	170.7

* letters indicate significant difference ($\alpha = 0.05$) per Wilcoxon Sign-Rank test within each pollutant, if no letter is present, there is no significant difference for that configuration

ttreatment type (Hathaway & Hunt, 2012). This is consistent with previous studies which report TSS reduction between 80% and 98% (Hatt et al., 2009; Blecken et al., 2010). Median effluent TSS concentrations were between 1.1 and 1.7 mg/L among the treatments with FD having the highest (1.7 mg/L) and SM (1.1 mg/L) having the lowest values. Similar laboratory studies of bioretention by Blecken et al. (2010), and Bratieres et al. (2008) reported comparable TSS concentrations of 2 mg/L, and 0.9-7.2 mg/L respectively.

Metals

Overall, effective removal of metals was observed across all treatment types, in particular for Zn^{2+} (Figure 1.5). This is consistent with observations seen in previous studies such as Laurenson et al. (2013) in which over 90% removal was reported for Zn^{2+} . When comparing Zn^{2+} to Cu^{2+} and Mn^{2+} , however, Zn^{2+} has over 93% removal for all treatment types, while more variability is noted for the other constituents. The more variable results for Cu^{2+} and Mn^{2+} (Figure 1.6 and Figure 1.7) are likely due to differences in treatment type and the variable conditions they provide. The overall magnitude of removal observed for Cu^{2+} is in line with previous studies from Blecken et al. (2009) and Laurenson et al. (2013) who showed 70% and >90% removal of Cu^{2+} , respectively, between treatments.

Removal of Cu^{2+} between SM, IWS, and FD treatments were all similar, ranging from approximately 87 to 90%. However, there was an observable difference in Cu^{2+} removal by the VC treatment, which could be a result of the more frequent, rapid, small water releases associated with this treatment type (as compared to SM). The more frequent storage and release by the VC scheme may alter the redox potentials within VC treatments by allowing oxygen into the system when active control valves open and close which limits Cu^{2+} sequestration through adsorption.

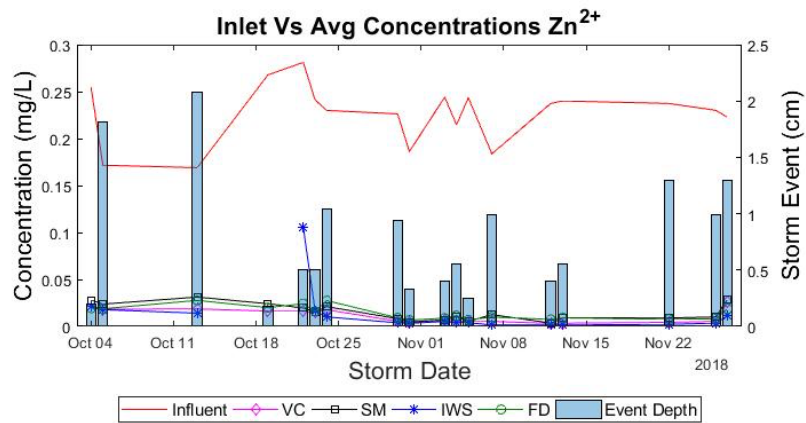


Figure 1.5 Zn^{2+} outlet concentrations for all treatment types

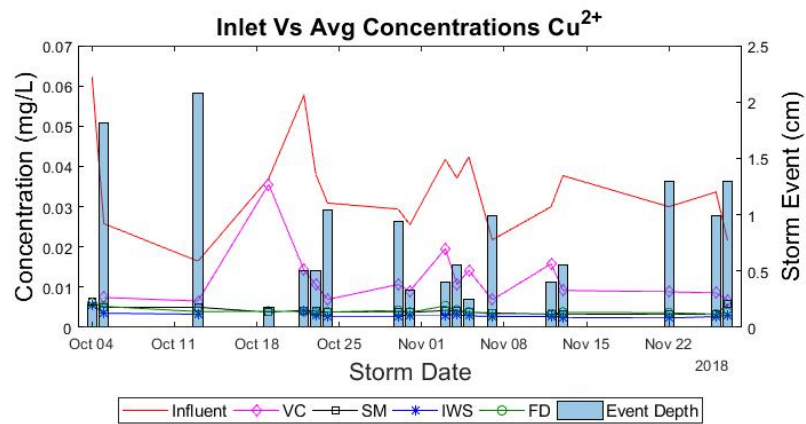


Figure 1.6 Cu^{2+} outlet concentrations for all treatment types

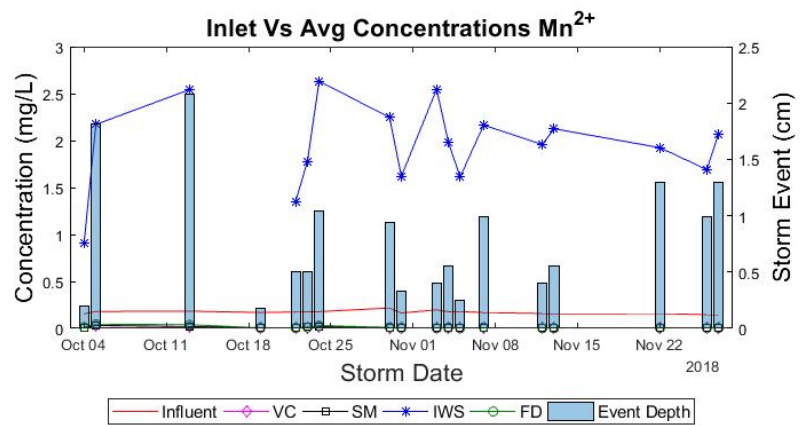


Figure 1.7 Mn²⁺ outlet concentrations for all treatment types

Similar changes in redox potential have led to Cu^{2+} dissolution because of oxic and anoxic variability within a given system (HamiltonTaylor et al., 1996; Chaudry & Zwolsman, 2008). The other treatment types had more stability in transporting water through the columns and were able to remove Cu^{2+} from stormwater influent more effectively. As noted above, these frequent releases may have been mitigated to some degree by utilizing a scheme that allowed an acceptable range of storage depths as opposed to one singular objective (30 cm). This would allow active control systems to better maintain a consistently anaerobic zone by minimizing level fluctuations.

In Figure 1.7, Mn^{2+} is exported from IWS treatments while all other columns showed similar removal trends to other metals. Media descriptions from the bioretention media manufacturer showed high levels of manganese in the media mix (Manganese Index =175). Furthermore, anaerobic heterotrophs within the IWS systems use manganese compounds within soils as electron donors and the reduced metal ions then leach out of the system (Nealson & Saffarini, 1994; Lee et al., 2001; Lovley et al., 2004). Because Mn^{2+} is soluble and mobile it can be transported readily, effectively flushing from the system. The IWS treatment appeared to facilitate anaerobic processing more than the other treatments (which is logical based on the soil moisture data) which likely explains the Mn^{2+} leaching. It should be noted that manganese concentrations are not typically a criterion in design manuals for bioretention media mixtures.

Nutrients

Nitrogen processing within bioretention systems was a focal point of this study, because export of nitrogen (NO_3^- -N) has been observed in previous studies after long periods of dry conditions and due to a presumed lack of the necessary anaerobic conditions in some bioretention designs, a required condition for denitrification (Hatt et al., 2007; Hsieh et al., 2007; Blecken et

al., 2010; Manka et al., 2016). As noted above, the SM active control treatment was designed to target field capacity to bolster microbial activity, specifically aerobic and anaerobic microbial processes, in an attempt to meet multiple nitrogen processing objectives (Barros et al., 1995; Schimel, 2018). Nutrient dynamics are described through the lense of microbial activity, which should be considered as influencing nutrient processing.

NO_3^- -N showed high variability in performance between treatment types with mean effluent concentrations ranging from 0.14 to 1.01 mg/L for the IWS and FD treatments, respectively (Figure 1.8). The FD treatment showed the least NO_3^- -N removal (67.2%), which is not surprising as it is the treatment considered to primarily foster an aerobic environment, resulting in the most limited conditions to facilitate denitrification (Collins et al., 2010). FD has no designated anaerobic zones to allow conversion, so any denitrification would have to be facilitated within the micropores of the bioretention media. The lack of denitrification as a result of the aerobic environment promoted by FD systems has been noted in studies such as Davis et al. (2006) and Li et al. (2014). IWS shows the greatest removal of all treatments (95.6% removal). This is attributed to a constant anaerobic zone in the IWS columns which facilitates denitrification. The VC and SM treatments showed similar performance with removal percentages between that of FD and IWS (73% and 74% removal, respectively). The VC treatment has a more shallow anaerobic zone than IWS and more frequent release which allows more aerobic processing than the IWS systems. At the same time, SM treatments allow a more stable release and are not dictated by a particular storage depth, but still retain more water than the FD treatment, resulting in aerobic and anaerobic processing.

Although unintentional, this spike in NO_2^- -N (Figure 1.9) does act as a sort of chemical tracer for the system, allowing an understanding of the differences in recovery times for each

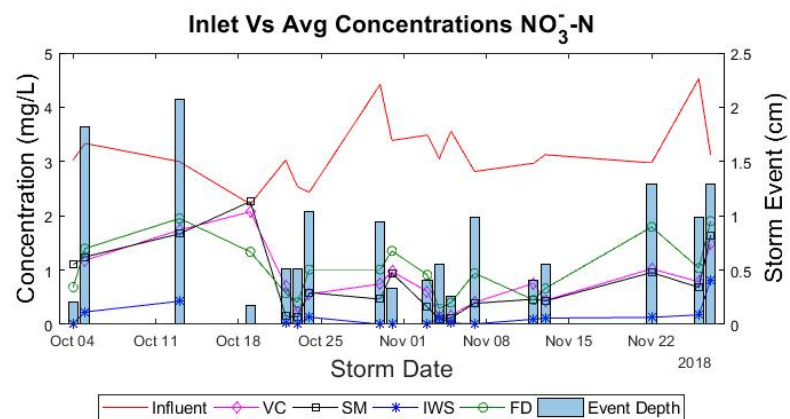


Figure 1.8 NO_3^- -N outlet concentrations for all treatment types

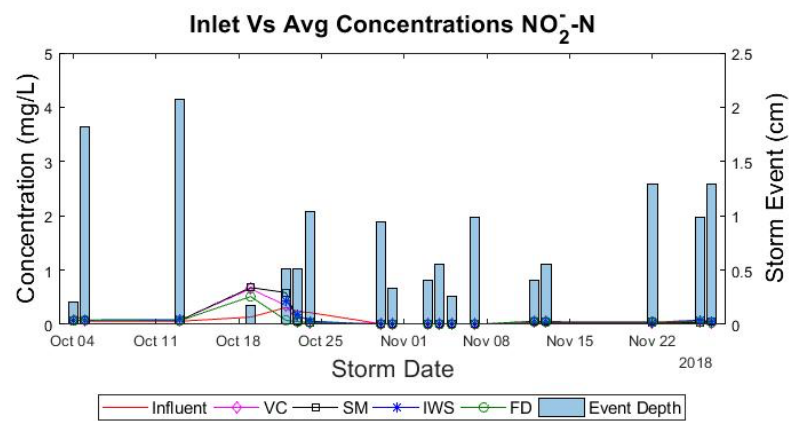


Figure 1.9 NO_2^- -N outlet concentrations for all treatment types

treatment type, that is, the amount of time required to bring the system back to producing typical NO_2^- -N effluent concentrations. This was generally linked to the amount of flushing provided by each treatment type. The FD treatment recovers after the next applied storm while other treatments required additional storm events before effluent NO_2^- -N concentrations return to a baseline in the system. This observation is likely due to the speed with which water moves through each treatment type. The FD treatment has the fastest flow through the system because it freely drains, with the VC, SM and IWS following in decreasing flow speed and increased water storage. The rate of flushing also infers differences in detention times between the systems, which likely also influences performance.

Nitrification is being promoted within all treatment types but most notably the FD treatment. As discussed above, this is expected based on the primarily aerobic environment provided by FD designs. Conversely, the IWS NH_4^+ -N effluent concentrations (Figure 1.10) are indicative of more limited aerobic processing, which is similar to results shown in Tang and Tian (2016) where IWS had less NH_4^+ -N reduction than the traditionally free draining column (63% and 71% respectively). The NH_4^+ -N remaining in the IWS system and being exported indicates the issue of incomplete aerobic processing. As noted above, VC and SM treatments both allow for more of an anaerobic zone than the FD treatments and less than that of the IWS treatment. They perform similarly to the FD treatment in regard to NH_4^+ -N reduction because of their presumed greater depth of aerobic zone but perform better than FD in regard to NO_3^- removal. This shows that there is more anaerobic processing facilitated in the actively controlled treatments, and that these systems may allow a balance between the conditions observed in FD and IWS designs.

Although NO_2^- -N is a less frequently reported and discussed parameter, it is often lumped with NO_3^- -N and reported as NO_x -N, it provides some insight into the denitrification process in

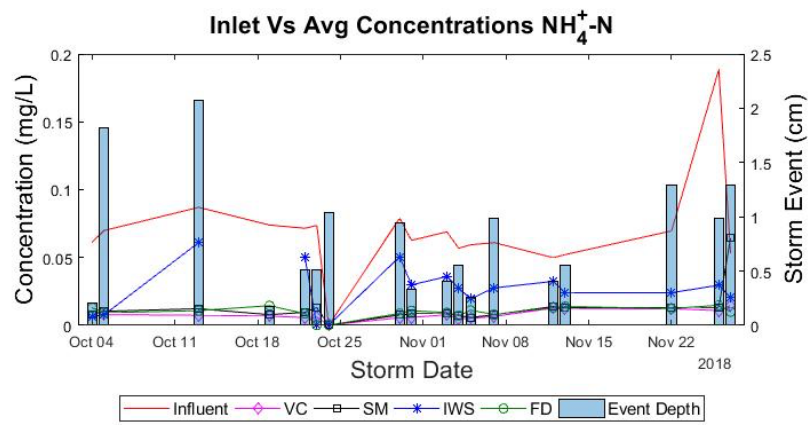


Figure 1.10 $\text{NH}_4^+\text{-N}$ outlet concentrations for all treatment types

the treatments. Overall, there is a consistent export of NO_2^- -N from all treatments (Figure 1.9). When coupled with data observed for NO_3^- -N and NH_4^+ -N, NO_2^- -N trends suggest the possibility of incomplete denitrification in the columns. That is, NO_3^- -N is converted to NO_2^- -N and produces N_2O gas (a greenhouse gas of major concern), but full conversion to N_2 gas is not occurring. This is potentially due to an inadequately deep saturated zone (lack of substantial anaerobic conditions) within these systems. This is worthy of further study, as completing the denitrification cycle is of critical importance for nitrogen management in biofilters.

A period of particular interest is the storm events and subsequent treatment that occurred in mid-October. Export of NO_2^- -N was noted, and to a smaller degree an increase in NO_3^- -N for some treatments, which follows the largest event during the study period and occurred during the smallest stormwater application of the study. It should also be noted that the upper soil layers for all columns were relatively dry during this event compared to the rest of the study. While the exact cause of this export is unknown, it is likely the result of large shifts in soil moisture between the two events and the subsequent impacts to biogeochemical processes (i.e. Manka et al. 2016). It should be noted that the IWS treatment was able to completely capture this event due to available storage.

Influence of Antecedent Conditions

Correlations between both 5-day antecedent rainfall and pollutant removal, and the antecedent number of dry days and pollutant removal (for all pollutants) were analyzed to determine the influence that wet and dry conditions have on water quality (which has been shown in studies such as Hatt et al. (2007) and Tang and Tian (2016)). Nutrient processing through physical and microbial interactions have the potential to be overloaded when a system is tasked

with managing frequent storm events. Likewise, periods of drought can affect biogeochemical processes, causing leaching from bioretention cells during subsequent storms (as was proposed for the mid-October event).

Contrary to results found by Manka et al. (2016) and Hatt et. al (2007), there was typically no significant correlation between removal and either measure of antecedent conditions. The one exception was a slightly negative correlation between NO_3^- -N removal by IWS treatment and the 5-day antecedent rainfall (Spearman Rank Correlation Coefficient = -0.54). Thus, there is minimal influence of wetting and drying periods on water quality. However, this study exhibited shorter dry periods (longest dry period of 9 days) and the lack of correlation is consistent with work done by Blecken et al. (2009) in which no effects were seen for dry periods shorter than 3 weeks. Work done by Hatt et al. (2007) also utilized longer dry and wet periods in examining removal performance. Further study on long term hydrologic implications of active control systems should be conducted to determine further correlations between treatment and removal. It is possible that active controls could be used to manage soil moisture more effectively during dry conditions, but this is an untested hypothesis.

Overall Comparison of Treatments

Traditional FD and IWS treatments can be considered controls to compare the efficacy of VC and SM treatments. FD and IWS represent the extremes of bioretention function, promoting aerobic and anaerobic conditions, respectively, and differential detention times. VC and SM treatments were actively controlled, leading to more variable patterns of water release compared to FD and IWS, and subsequent differences in storage times and soil moisture patterns. These

trends were found to influence water quality, being an explanatory factor for dissimilarities in metal and nutrient effluent concentrations from the treatments.

Overall, deeper water storage zones lead to better anaerobic nutrient processing of nitrate (denitrification), while shallower water storage zones allow for greater aerobic treatment and conversion of $\text{NH}_4^+\text{-N}$ to $\text{NO}_3^-\text{-N}$. In this study, this understanding played out by the FD treatments more effective at performing nitrification (i.e. $\text{NH}_4^+\text{-N}$ concentration reductions were accomplished), while the IWS treatment showed the most reduction in $\text{NO}_3^-\text{-N}$ concentrations, indicating more denitrification when compared to other treatments. The VC and SM treatments were found to be better at performing nitrification than the FD treatment but not better at performing denitrification than the IWS treatment, that is, they were able to provide both nitrification and denitrification in moderation (compared to other treatments). This provides some hope that continued scheme development for actively controlled bioretention may lead to systems that can balance the conflicting aerobic and anaerobic environments needed for fully processing nitrogen.

In terms of metals, the treatments largely performed similarly other than the IWS treatment exported Mn^{2+} , and the VC treatment removing Cu^{2+} with less efficiency. This resulted in a few notable observations as to how active controls could influence metal concentrations (e.g. by effecting redox potential). Similar results are evident with TSS removal being over 97%.

Although not a focus of this study, it should be noted that the IWS treatment was able to store runoff from smaller rain events which would result in total runoff reduction. For fewer storms, SM and VC treatments were also able to do the same as the threshold for active release was not reached. The hydrologic implications of the various treatment types should be further studied to understand how active controls can be used to balance volume reduction and water quality improvement. We

hypothesize that active controls will be able to meet these multiple objectives more effectively than static systems.

Conclusions

This column study tested the use of active control systems, as compared to static designs, over a 9-week period by observing water quality improvements provided by each treatment. Historic weather predictions were coupled with observed precipitation events to replicate weather conditions from June and July 2017, which amounted to a total of 18 storm events. Most notable was the influence of the treatments on nutrients. For nutrients in the static systems, the largely aerobic free draining performed best for $\text{NH}_4^+\text{-N}$, while the more anaerobic environment provided by the internal water storage led to the best performance for $\text{NO}_3^-\text{-N}$. Deeper media depths could remedy this issue in future implementation of IWS treatment, that is, a larger aerobic zone above the IWS could be provided. As the optimum IWS depth for water quality has not been explored in literature, these data suggest that balancing nitrification and denitrification is critical and more scientifically informed IWS design is possible. The soil moisture and volume control treatments were able to balance these two environments, removing $\text{NH}_4^+\text{-N}$ by more than 40% and $\text{NO}_3^-\text{-N}$ by more than 73%. Differences between soil moisture and volume control were minimal for nutrients. This suggests that active controlled systems may strike a balance between traditional free draining and internal water storage systems.

Numerous factors influenced the results of this research and should be considered in future research. First, using one scheme per configuration targets only one control objective but having multiple control objectives could further improve the effectiveness of active control. Second, having a seepage port allows the columns to better mimic field conditions in a laboratory setting, but also results in biofouling and should be carefully monitored. Finally, Eastern Tennessee is

subject to frequent, hard to predict thunderstorms in summer months which could have affected the rainfall forecast data used herein. Using rainfall data from easier to predict seasons may affect the results of the study.

Future research into actively controlled bioretention systems should include more hydrologic quantification of bioretention systems outfitted with this technology in both laboratory and field-scale studies. This should be coupled with further development of active control schemes to balance water quality and hydrologic objectives using soil moisture readings at variable media depths and by incorporating depth sensor measurements. The use of weather predictions in designing schemes for active control systems is critical, and additional study should be performed to compare preemptive control based on weather predictions to more adaptive control during storm events able. That is, if retention time is a critical treatment consideration, can it be further optimized by considering uncertainties in weather prediction? Despite these questions, active control systems show promise for the future of designing more efficient bioretention systems that are adaptive to external and internal environmental processes.

Acknowledgments

This work was funded by the National Science Foundation (NSF Grant number 1737432). The authors would like to thank University of Tennessee Civil Engineering workshop, Abhiram Mullapudi of the University of Michigan Real-Time Water Systems Lab, and East Tennessee AgResearch and Education Center who were crucial in building and monitoring of this study.

References

- Barros, N., Gomezorellana, I., Feijoo, S., & Balsa, R. (1995). THE EFFECT OF SOIL-MOISTURE ON SOIL MICROBIAL ACTIVITY STUDIED BY MICROCALORIMETRY. *Thermochimica Acta*, 249, 161-168. doi:10.1016/0040-6031(95)90686-x
- Blecken, G. T., Zinger, Y., Deletic, A., Fletcher, T. D., Hedstrom, A., & Viklander, M. (2010). Laboratory study on stormwater biofiltration Nutrient and sediment removal in cold temperatures. *Journal of Hydrology*, 394(3-4), 507-514. doi:10.1016/j.jhydrol.2010.10.010
- Blecken, G. T., Zinger, Y., Deletic, A., Fletcher, T. D., & Viklander, M. (2009). Influence of intermittent wetting and drying conditions on heavy metal removal by stormwater biofilters. *Water Research*, 43(18), 4590-4598. doi:10.1016/j.watres.2009.07.008
- Bratieres, K., Fletcher, T. D., Deletic, A., & Zinger, Y. (2008). Nutrient and sediment removal by stormwater biofilters: A large-scale design optimisation study. *Water Research*, 42(14), 3930-3940. doi:10.1016/j.watres.2008.06.009
- Chandrasena, G. I., Shirdashtzadeh, M., Li, Y. L., Deletic, A., Hathaway, J. M., & McCarthy, D. T. (2017). Retention and survival of E. coli in stormwater biofilters: Role of vegetation, rhizosphere microorganisms and antimicrobial filter media. *Ecological Engineering*, 102, 166-177. doi:10.1016/j.ecoleng.2017.02.009
- Chaudry, M. A., & Zwolsman, J. J. G. (2008). Seasonal dynamics of dissolved trace metals in the scheldt estuary: Relationship with redox conditions and phytoplankton activity. *Estuaries and Coasts*, 31(2), 430-443. doi:10.1007/s12237-007-9030-7
- Collins, K. A., Lawrence, T. J., Stander, E. K., Jontos, R. J., Kaushal, S. S., Newcomer, T. A., . . . Ekberg, M. L. C. (2010). Opportunities and challenges for managing nitrogen in urban stormwater: A review and synthesis. *Ecological Engineering*, 36(11), 1507-1519. doi:10.1016/j.ecoleng.2010.03.015

- Colmer, T. D. (2003). Long-distance transport of gases in plants: a perspective on internal aeration and radial oxygen loss from roots. *Plant Cell and Environment*, 26(1), 17-36. doi:10.1046/j.1365-3040.2003.00846.x
- Davis, A. P., Shokouhian, M., Sharma, H., & Minami, C. (2006). Water quality improvement through bioretention media: Nitrogen and phosphorus removal. *Water Environment Research*, 78(3), 284-293. doi:10.2175/106143005x94376
- Dietz, M. E., & Clausen, J. C. (2005). A field evaluation of rain garden flow and pollutant treatment. *Water Air and Soil Pollution*, 167(1-4), 123-138. doi:10.1007/s11270-005-8266-8
- Fierer, N., & Schimel, J. P. (2002). Effects of drying-rewetting frequency on soil carbon and nitrogen transformations. *Soil Biology & Biochemistry*, 34(6), 777-787. doi:10.1016/s0038-0717(02)00007-x
- Fletcher, T. D., Shuster, W., Hunt, W. F., Ashley, R., Butler, D., Arthur, S., . . . Viklander, M. (2015). SUDS, LID, BMPs, WSUD and more - The evolution and application of terminology surrounding urban drainage. *Urban Water Journal*, 12(7), 525-542. doi:10.1080/1573062x.2014.916314
- Glaister, B. J., Fletcher, T. D., Cook, P. L. M., & Hatt, B. E. (2017). Interactions between design, plant growth and the treatment performance of stormwater biofilters. *Ecological Engineering*, 105, 21-31. doi:10.1016/j.ecoleng.2017.04.030
- Guo, Y. P., & Adams, B. J. (1998). Hydrologic analysis of urban catchments with event-based probabilistic models - 1. Runoff volume. *Water Resources Research*, 34(12), 3421-3431. doi:10.1029/98wr02449
- HamiltonTaylor, J., Davison, W., & Morfett, K. (1996). A laboratory study of the biogeochemical cycling of Fe, Mn, Zn and Cu across the sediment-water interface of a productive lake. *Aquatic Sciences*, 58(3), 191-209. doi:10.1007/bf00877508

- Hathaway, J. M., & Hunt, W. F. (2012). Indicator Bacteria Performance of Storm Water Control Measures in Wilmington, North Carolina. *Journal of Irrigation and Drainage Engineering*, 138(2), 185-197. doi:10.1061/(asce)ir.1943-4774.0000378
- Hatt, B. E., Fletcher, T. D., & Deletic, A. (2009). Pollutant removal performance of field-scale stormwater biofiltration systems. *Water Science and Technology*, 59(8), 1567-1576. doi:10.2166/wst.2009.173
- Hatt, E., Fletcher, D., & Deletic, A. (2007). Hydraulic and pollutant removal performance of stormwater filters under variable wetting and drying regimes. *Water Science and Technology*, 56(12), 11-19. doi:10.2166/wst.2007.751
- Henderson, C., Greenway, M., & Phillips, I. (2007). Removal of dissolved nitrogen, phosphorus and carbon from stormwater by biofiltration mesocosms. *Water Science and Technology*, 55(4), 183-191. doi:10.2166/wst.2007.108
- Hsieh, C. H., Davis, A. P., & Needelman, B. A. (2007). Nitrogen removal from urban stormwater runoff through layered bioretention columns. *Water Environment Research*, 79(12), 2404-2411. doi:10.2175/106143007x183844
- Hunt, W. F., Smith, J. T., Jadlocki, S. J., Hathaway, J. M., & Eubanks, P. R. (2008). Pollutant removal and peak flow mitigation by a bioretention cell in urban Charlotte, NC. *Journal of Environmental Engineering-Asce*, 134(5), 403-408. doi:10.1061/(asce)0733-9372(2008)134:5(403)
- Kirkham, M. B. (2005). *Field Capacity, Wilting Point, Available Water, and the Non-Limiting Water Range*. Amsterdam: Elsevier Science Bv.
- Larsen, T. A., Hoffmann, S., Luthi, C., Truffer, B., & Maurer, M. (2016). Emerging solutions to the water challenges of an urbanizing world. *Science*, 352(6288), 928-933. doi:10.1126/science.aad8641
- Laurenson, G., Laurenson, S., Bolan, N., Beecham, S., & Clark, I. (2013). The Role of Bioretention Systems in the Treatment of Stormwater. In D. L. Sparks (Ed.), *Advances in Agronomy, Vol 120* (Vol. 120, pp. 223-274). San Diego: Elsevier Academic Press Inc.

- Le Coustumer, S., Fletcher, T. D., Deletic, A., Barraud, S., & Poelsma, P. (2012). The influence of design parameters on clogging of stormwater biofilters: A large-scale column study. *Water Research*, 46(20), 6743-6752. doi:10.1016/j.watres.2012.01.026
- Lee, E. Y., Noh, S. R., Cho, K. S., & Ryu, H. W. (2001). Leaching of Mn, Co, and Ni from manganese nodules using an anaerobic bioleaching method. *Journal of Bioscience and Bioengineering*, 92(4), 354-359. doi:10.1263/jbb.92.354
- Li, M. H., Swapp, M., Kim, M. H., Chu, K. H., & Sung, C. Y. (2014). Comparing Bioretention Designs With and Without an Internal Water Storage Layer for Treating Highway Runoff. *Water Environment Research*, 86(5), 387-397. doi:10.2175/106143013x13789303501920
- Lovley, D. R., Holmes, D. E., & Nevin, K. P. (2004). Dissimilatory Fe(III) and Mn(IV) reduction. In R. K. Poole (Ed.), *Advances in Microbial Physiology*, Vol. 49 (Vol. 49, pp. 219-286). London: Academic Press Ltd-Elsevier Science Ltd.
- Manka, B. N., Hathaway, J. M., Tirpak, R. A., He, Q., & Hunt, W. F. (2016). Driving forces of effluent nutrient variability in field scale bioretention. *Ecological Engineering*, 94, 622-628. doi:10.1016/j.ecoleng.2016.06.024
- McPhillips, L., Goodale, C., & Walter, M. T. (2018). Nutrient Leaching and Greenhouse Gas Emissions in Grassed Detention and Bioretention Stormwater Basins. *Journal of Sustainable Water in the Built Environment*, 4(1), 10. doi:10.1061/jswbay.0000837
- Morse, N., Payne, E., Henry, R., Hatt, B., Chandrasena, G., Shapleigh, J., . . . McCarthy, D. (2018). Plant-Microbe Interactions Drive Denitrification Rates, Dissolved Nitrogen Removal, and the Abundance of Denitrification Genes in Stormwater Control Measures. *Environmental Science & Technology*, 52(16), 9320-9329. doi:10.1021/acs.est.8b02133
- Mullapudi, A., Bartos, M., Wong, B., & Kerkez, B. (2018). Shaping Streamflow Using a Real-Time Stormwater Control Network. *Sensors*, 18(7), 11. doi:10.3390/s18072259

- Mullapudi, A., Wong, B. P., & Kerkez, B. (2017). Emerging investigators series: building a theory for smart stormwater systems. *Environmental Science-Water Research & Technology*, 3(1), 66-77. doi:10.1039/c6ew00211k
- NCDEQ, N. C. D. o. E. Q. (2018). Stormwater Design Manual. *North Carolina Environmental Quality*. Retrieved from <https://files.nc.gov/ncdeq/Energy+Mineral+and+Land+Resources/Stormwater/BMP+Manual/C-2%20%20Bioretention%201-19-2018%20FINAL.pdf>
- Nealson, K. H., & Saffarini, D. (1994). IRON AND MANGANESE IN ANAEROBIC RESPIRATION - ENVIRONMENTAL SIGNIFICANCE, PHYSIOLOGY, AND REGULATION. *Annual Review of Microbiology*, 48, 311-343. doi:10.1146/annurev.mi.48.100194.001523
- Parolari, A. J., Pelrine, S., & Bartlett, M. S. (2018). Stochastic water balance dynamics of passive and controlled stormwater basins. *Advances in Water Resources*, 122, 328-339. doi:10.1016/j.advwatres.2018.10.016
- Payne, E. G. I., Fletcher, T. D., Cook, P. L. M., Deletic, A., & Hatt, B. E. (2014). Processes and Drivers of Nitrogen Removal in Stormwater Biofiltration. *Critical Reviews in Environmental Science and Technology*, 44(7), 796-846. doi:10.1080/10643389.2012.741310
- Pulleman, M., & Tietema, A. (1999). Microbial C and N transformations during drying and rewetting of coniferous forest floor material. *Soil Biology & Biochemistry*, 31(2), 275-285. doi:10.1016/s0038-0717(98)00116-3
- Salazar, A., Sulman, B. N., & Dukes, J. S. (2018). Microbial dormancy promotes microbial biomass and respiration across pulses of drying-wetting stress. *Soil Biology & Biochemistry*, 116, 237-244. doi:10.1016/j.soilbio.2017.10.017
- Schimel, J. P. (2018). Life in Dry Soils: Effects of Drought on Soil Microbial Communities and Processes. In D. J. Futuyma (Ed.), *Annual Review of Ecology, Evolution, and Systematics*, Vol 49 (Vol. 49, pp. 409-432). Palo Alto: Annual Reviews.

- Tang, N. Y., & Tian, L. (2016). Nitrogen removal by three types of bioretention columns under wetting and drying regimes. *Journal of Central South University*, 23(2), 324-332. doi:10.1007/s11771-016-3077-1
- TSM, T. S. M. (2015). Management and Design Guidance Manual. *Tennessee Stormwater Managment*. Retrieved from <https://tnpermanentstormwater.org/manual.asp#>
- USEPA, U. S. E. P. A. (Producer). (2016). National Summary of State Information. *USEPA*. Retrieved from https://ofmpub.epa.gov/waters10/attains_index.control#causes
- Vangestel, M., Merckx, R., & Vlassak, K. (1993). MICROBIAL BIOMASS RESPONSES TO SOIL DRYING AND REWETTING - THE FATE OF FAST-GROWING AND SLOW-GROWING MICROORGANISMS IN SOILS FROM DIFFERENT CLIMATES. *Soil Biology & Biochemistry*, 25(1), 109-123. doi:10.1016/0038-0717(93)90249-b
- Waller, L. J., Evanylo, G. K., Krometis, L. A. H., Strickland, M. S., Wynn-Thompson, T., & Badgley, B. D. (2018). Engineered and Environmental Controls of Microbial Denitrification in Established Bioretention Cells. *Environmental Science & Technology*, 52(9), 5358-5366. doi:10.1021/acs.est.7b06704
- Yan, Q., James, B. R., & Davis, A. P. (2017). Lab-Scale Column Studies for Enhanced Phosphorus Sorption from Synthetic Urban Stormwater Using Modified Bioretention Media. *Journal of Environmental Engineering*, 143(1), 13. doi:10.1061/(asce)ee.1943-7870.0001159
- Zhou, X., Fornara, D., Ikenaga, M., Akagi, I., Zhang, R. F., & Jia, Z. J. (2016). The Resilience of Microbial Community under Drying and Rewetting Cycles of Three Forest Soils. *Frontiers in Microbiology*, 7, 12. doi:10.3389/fmicb.2016.01101

Chapter 2 : Hydrologic Implications of Real-time Control on Bioretention

¹Dept. of Civil and Environmental Engineering, University of Tennessee, 325 John D. Tickle Building, 851 Neyland Dr., Knoxville, TN, 37996 USA

*Corresponding author: hathaway@utk.edu

Primary Research question:

What effects do real-time control schemes have on bioretention hydrologic functioning?

Abstract

Bioretention practices have been developed to restore natural hydrologic regimes by reducing runoff volume and mitigating the peak flows of urban runoff. This is critical as cities encounter more extreme weather and their aging infrastructure is left ill equipped to manage new stormwater management challenges. However, functional improvements to bioretention through the use of real-time control (RTC) systems can allow more responsive system adjustments to manage incoming runoff volumes. A six-week column study was conducted in which static bioretention designs (i.e., free draining (FD) and internal water storage (IWS)) were tested against two RTC designs that focused on either regulating soil moisture (SM) or maximizing internal storage volumes (VC). Of the two RTC designs, the SM configuration showed the most storage capability (18%) and the lowest rate of Bypass (7%) compared to the VC configuration which showed storage and Bypass rates of 11% each. The FD And IWS configurations exhibited storage at 9% and 16% and Bypass at 2% and 11% respectively. This shows the potential for RTC to meet multiple, sometimes conflicting objectives, in this case volume reduction and bypass minimization. Analysis of Total Nitrogen (TN) removal revealed the greatest removal to occur in

the VC configuration (82% removal) while the SM configuration performed similarly to IWS and FD. Differences between TN removal and volume reduction results indicate removal was aided by biogeochemical processing and not just the result of volume reductions. Correlation analysis revealed strong correlations between storm size and effluent volumes, but no correlation with seepage. Infiltration rates appeared to plateau regardless of the length of antecedent dry periods, suggesting that infiltration is limited by water availability. Future research into the use of RTC systems for bioretention should include variable rainfall intensity in lab scale studies to better understand flow dynamics over time. Studies should also be conducted in field-scale experiments to understand the larger practical implications of their design and implementation.

Introduction

Expansion of cities worldwide has led to the proliferation of impervious surfaces and a subsequent increase in surface runoff. Routing this runoff to nearby water bodies causes a surge in flow which natural systems have difficulty managing (Brabec et al. 2002; Bronstert et al. 2002). As a result, flooding, erosion, and changes in stream geomorphology threaten the surrounding landscape and inhabited areas (Konrad 2003). With the goal of volume reduction and peak flow mitigation, green infrastructure practices, like bioretention, have been developed to combat hydrologic regime shifts while still allowing urban development (Dietz and Clausen 2005; Hunt et al. 2008; Liu et al. 2014; Fletcher et al. 2015; CSN 2013). They employ natural environmental processing to infiltrate stormwater into surrounding soils and reduce the amount of stormwater entering stream and river systems (TSM 2015; TDEC 2014; Hopkins et al. 2019; Hamel and Fletcher 2014). Being a more at-source reduction practice, bioretention areas are typically small and target areas like parking lots or roadways where water is concentrated and otherwise would be

channeled into storm drains (Burns et al. 2012). These practices store stormwater and gradually release it back into stormwater systems while also allowing some groundwater recharge. This results in a reduction of peak flows in receiving water bodies.

Bioretention practices are traditionally free draining systems (i.e. drain via gravity) that allow water to filter through a highly porous media and (often) utilize underdrains to carry excess infiltrate to existing stormwater networks (CSN 2013). The use of internal water storage (IWS) systems has become a common adaptation to promote more conducive conditions for water quality treatment processes by creating an aerobic and anaerobic zone within the cell (Manka et al. 2016; Waller et al. 2018). However, because these practices intentionally reduce their storage capacity, they can be more limited in the amount of incoming stormwater they can manage (depending on underlying soil permeability) (Davis et al. 2012). Research in high intensity rainfall studies show that LID practices, like bioretention, provide flood reduction benefits for smaller (2-year storms) rain events but are ineffective for larger (10-year storms) events (Juan et al. 2017; Bell et al. 2020; Wang et al. 2019). The development of real-time or active control systems may allow improvements to bioretention design through controlled storage and release of stormwater. In previous studies exploring nutrient concentration reductions, this technology has been shown to create a balance between free draining and IWS systems, allowing co-creation of aerobic and anaerobic conditions through manipulation of internal water storage and soil moisture dynamics (Persaud et al. 2019). In the same way, these technologies may be able to maximize runoff storage while still allowing available storage for incoming rain events. Further, tracking of all hydrologic inputs and outputs allows a nutrient mass balance to better understanding the relationships between water quality and hydrologic performance in GI systems (building on previous research focused on pollutant concentrations).

Real-time control (RTC) systems utilize weather predictions, sensor data, and/or other data streams to adjust control valves for storage and release of stormwater (Kerkez, Daniels, et al. 2016; Kerkez, Gruden, et al. 2016). This allows substantially more versatility and dynamic responses in comparison to traditionally free draining and IWS systems which rely on gravity and physical design characteristics to meet treatment goals. Development of active control technology focuses on control schemes, which are used to maintain or reach specific objectives for system conditions (Kerkez, Gruden, et al. 2016; Parolari et al. 2018; Roman et al. 2017). These can be tailored for water quality or hydrologic objectives depending on the needs of the practice. Research performed by Mullanpudi et al. (2018) and Bartos et al. (2018) showed the capability RTC to shape stream flow and be applied in large-scale urban watershed systems as the technology includes sensing, control, and cloud computing. The potential for water quality improvement has also been studied and shows promise in the use of actively controlled systems (Persaud et al. 2019; Middleton and Barrett 2008); however, the majority of these studies have focused on open water systems such as wet ponds. The hydrologic implications of using active control on bioretention practices has not been determined.

In an effort to address this gap in knowledge, a six-week column study was conducted to explore the hydrologic implications of using RTC in bioretention practices. Water balances of traditional (FD), IWS, and RTC systems were conducted and compared to examine differences between treatments. This research will aim to 1) assess the overall volume management potential of RTC systems, 2) relate these hydrologic outcomes to holistic water quality performance in the form of total nitrogen mass balance, and 3) provide recommendations for optimization whereby both water quality and hydrologic goals are satisfied. This performance evaluation for bioretention can also provide insight into implications for other filtration-based green infrastructure practices.

Methods

Column Design

Four treatments with five replicates each were used in this study. Columns that mimic bioretention function were employed to replicate the following designs: free draining (FD), internal water storage (IWS), and two real-time control schemes (VC and SM as described below). All columns were constructed using 30 cm diameter gray PVC which was sanded prior to filling to decrease the chances of preferential flow patterns forming. They were each fitted with ports on the bottom to replicate seepage at a rate of 0.20 cm/hr (to mimic a clay soil type). Columns were filled with layers of gravel (washed #57 stone), washed pea gravel, sand, bioretention media, and mulch with a 10 cm ponding zone. Each column was also planted with *Echinacea purpurea* (purple coneflower) and one *Juncus effuses* (common rush)(Figure 2.1).

Column Monitoring

Each column was monitored for soil moisture with Decagon GS1 sensors buried at 30 and 60 cm depths measured down from the top of the media. Soil moisture sensor calibration was performed in the hydraulics laboratory of the Department of Civil and Environmental Engineering at the University of Tennessee according to manufacturer recommendations. A Stevens pressure transducer was used to monitor water levels within the columns. Pressure transducer calibration was performed by incremental submergence of the sensor in known water depths and recording raw values to form regression equations. Monitoring data was stored on an InfluxDB database and viewed using Grafana.

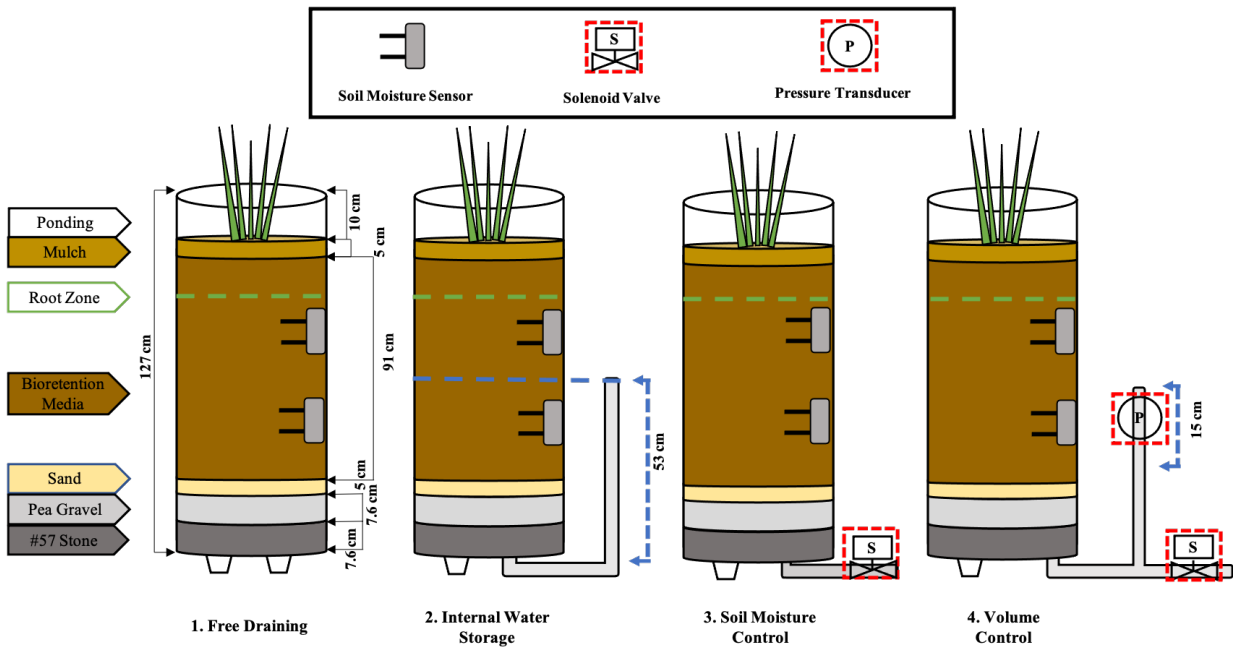


Figure 2.1 Configuration design with altered volume control column

RTC Treatments

RTC columns in this study operate under one of two control schemes. VC configurations maintain an internal storage zone using a Stevens pressure transducer to monitor water levels within the column. Weather predictions are sent to these columns to determine the amount of water release needed to maintain targeted water levels after the event refills the column. During an application event, if water levels are exceeded, the system will also allow water release to further maintain water levels. For SM configurations, the control scheme utilized GS1 soil moisture sensors at the 30 cm depth to maintain field capacity of the bioretention media. Weather predictions were also sent to these columns which allowed water release depending on the volume of stormwater expected to enter the system. Further details on the function of the RTC treatments are provided below.

In a previous study by Persaud et al. (2019), oxygen appeared to enter the columns through the outlet whenever the system would drain because the fast release of water did not allow sustained anaerobic conditions. Thus, a new control scheme was developed to allow a 15 cm range of allowable storage depth (between 22 and 37 cm), thereby reducing the rapid opening and closing of the valve that was observed when a specific storage depth was chosen as the objective. Building off the observations of (Persaud et al. 2019) that rapid drainage was occurring at the outlets of VC columns, a reducer was put on the column outlets to allow a slower release of water. This would keep the lower half of the VC column constantly saturated while allowing the control system to adjust for incoming flows.

Experimental Procedure

This six-week column study was conducted in the Summer of 2019 and involved 12 storm events which ranged in size from 0.22 cm to 2.54 cm. The experiment emulated 12 weeks of historical data for Alcoa, TN, recorded from August 8th to September 19th, 2018. Experiments were conducted in a climate-controlled greenhouse with the average temperature being 28°C during the time of the study.

Pre-Event Experimental Method

Weather prediction data for the 12 weeks being mimicked were obtained from the National Oceanic and Atmospheric Administration (NOAA) prior to the start of this study. The McGhee Tyson airport in Alcoa, TN, was used as a reference location. To properly represent RTC function, a weather prediction was sent to the actively controlled bioretention columns 12-hours prior to the event happening. Consistent with real-world conditions, in some cases a prediction was made and no event followed, in other cases no prediction existed yet a rain event occurred regardless. In cases where a prediction was made per NOAA, RTC columns would respond according to their control schemes by releasing water based on either soil moisture level (SM) or water storage level (VC) objectives (as described in the RTC Treatments section).

Specifically, the amount of runoff generated from the predicted rainfall depth was first calculated. SM and VC columns would then determine how much drainage should be released from their controllable valves such that this inflow volume would result in the objective soil moisture or water level being met post storm. That is, how much water currently in the column should be discharged such that the inflow could replace it. For SM columns pre-event, water at the bottom of the practice was allowed to drain out of the system which would lower soil moisture

levels in the upper portions of the media and trigger a subsequent closing of the control valve when the targeted value was met. For VC treatments pre-event, the columns would drain to a new water level until the time of the incoming precipitation. In this way the system would be prepared for incoming weather. The SM system-based decisions on soil moisture readings at the 30 cm depth.

Application

Columns were dosed with a semi-synthetic stormwater mixture which was created using dechlorinated tap water and additional chemicals to match typical pollutant concentrations (Bratieres et al. 2008). The volume of the stormwater mix applied to each column was based on a 20:1 sizing ratio, that is, general guidance stating that bioretention areas are approximately 5% of their contributing watershed (TSM 2015). Applications took place in three passes to ensure even distributions of sediment and contaminants. For larger applications, some column types were unable to take on the intended application volume. Once a column was fully ponded, any further application was done after a 30-minute wait period. If columns were still fully ponded, the application process would be considered complete and any remaining synthetic stormwater meant to be applied was counted as Bypass.

Hydrologic Measurements

All inputs and outputs of stormwater to the columns were recorded allowing a water balance of each column to determine hydrologic partitioning. The volume of water dosed for each storm event, Bypass volumes, outlet volumes, and seepage volumes were measured to determine differences in water management among treatments. The Prediction variable was used in calculations for RTC columns to denote the amount of drainage that occurred after a prediction

was sent to columns. That is, the amount of volume released from the outlet of RTC columns when control valves were signaled to open when expecting incoming weather. As traditional bioretention columns were not fitted with control schemes, only RTC columns include this variable. To measure volumes, two collection bins were used: one for drainage at the outlet for each column and one for seepage measurements under each seepage port. When storm events were more than 3 days apart, seepage volumes were taken more frequently to ensure constant seepage and volume tracking took place. For the purposes of the water balance, Equation 1 was used to determine Storage/ET (storage and evapotranspiration) over the course of the entire study as well as for each storm. The drainage volume for one storm event is described as the volume of drainage exhibited by a column for a 24-hour period after the time of storm event dosing.

Water Balance Equation:

$$\text{Inlet} - \text{Bypass} - \text{Drainage} - \text{Seepage} - \text{Prediction} = \text{Storage/ET} \quad (\text{Eq. 1})$$

Variables used in Equation 1 are intended to describe each component of a water balance for each column configuration. “Inlet” is used to indicate the volume of water the system can manage as influent while “Bypass” indicates the volume the system could not manage from a rain event which would have been diverted. The variable “Drainage” is used to indicate the volume of water leaving the system as effluent while “Seepage” is the volume of water leaving the system through mimicked infiltration. “Prediction” is used to denote the volume of water lost from the system as effluent when a prediction is sent to RTC columns prior. Finally, “Storage/ET” is the volume of water retained by the columns.

Analysis of Total Nitrogen

Water quality data for nitrogen constituents were collected for each storm event during the 6-week study period. The influent and effluent of three columns of each configuration were sampled with Total Kjeldahl Nitrogen (TKN) (mg/L) being analyzed by Microbac laboratories. The Water Quality Core Facility at the University of Tennessee was used to determine concentrations in mg/L of nitrate (NO_3^- -N), nitrite (NO_2^- -N), and ammonium (NH_4^+ -N) using ion chromatography. Using these data, as well as the recorded effluent volumes (recorded in L), the mass of each nitrogen constituent (in mg) in the inlet and outlet stormwater applications was calculated.

N Constituent Mass Calculations:

$$\text{TON} = \text{TKN} - \text{NH}_4^+ \text{-N} \quad (\text{Eq.2})$$

$$\text{TN} = \text{TKN} + \text{NO}_3^- \text{-N} + \text{NO}_2^- \text{-N} \quad (\text{Eq. 3})$$

$$\text{TN} = \text{TON} + \text{NH}_4^+ \text{-N} + \text{NO}_3^- \text{-N} + \text{NO}_2^- \text{-N} \quad (\text{Eq. 4})$$

As Total Nitrogen is typically the water quality variable of interest for regulatory purposes, it was the focus of this analysis. The overall mass balance was calculated for of Total Nitrogen (TN) using Equation 6.

N Mass Calculations:

$$\text{“Removed” (mg)} = \text{Retained (mg)} + \text{Infiltrated (mg)} \quad (\text{Eq. 5})$$

$$\text{“Removed” (mg)} = \text{Total Influent (mg)} - \text{Bypass (mg)} - \text{Outflow (mg)} - \text{Prediction (mg)} \quad (\text{Eq. 6})$$

The variable “Retained” refers to the mass of TN retained within the system while “Infiltrated” is the mass assumed to be lost by the system and removed by subsurface processes. “Total Influent” refers to the total mass of nitrogen in the Influent of every storm event while “Bypass” is the mass of nitrogen that was diverted from the system. “Outflow” and “Prediction” are intended to be the masses of Nitrogen in the effluent either as drainage, or the drainage caused by predictions that the RTC columns received. The variables Infiltrated, Outflow, and Prediction are assumed to use the outlet concentration, while the Influent and Bypass variables use inlet concentrations in the overall mass balance. Note that there is some error regarding Infiltrated and Prediction concentrations as these values were not measured. Some preliminary testing using double and half concentrations for Infiltrated and Prediction values do not show differences in the overall mass balance greater than 0.5 mg.

Statistical Analysis

Statistical analysis was performed using R version 4.0.3. Normality was assessed using the both the Shapiro-Wilk and Kolmogorov-Smirnov test which found both normal and non-normal distribution depending on the variable of interest. Further analysis was completed using non-parametric statistical analysis to allow consistent treatment of data. A Wilcoxon sign rank test was used to determine if significant differences occurred between replicates while a Wilcoxon rank-sum test was used to compare treatment types and determine significant differences between column types. Finally, the Spearman’s Rank correlation coefficient was used to determine the influence of antecedent dry periods and storm size on effluent volumes and storage. Volume reduction was calculated as the difference between influent and effluent volumes. The influent

volumes were calculated as Bypass subtracted from Inflow volume while the effluent volumes reflecting Drainage. Statistical difference was determined using a 0.05 significance level.

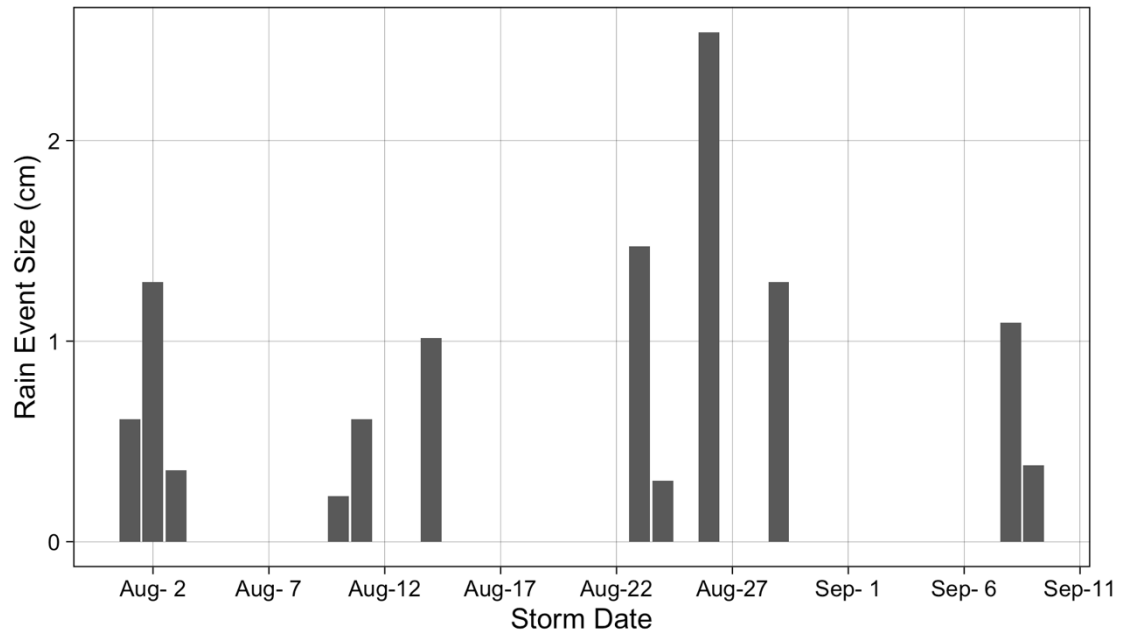
Results and Discussion

Hydrologic data collected over the six-week study period were analyzed using a water balance in which water balance variables are compared to determine shifts in hydrologic pathways for each treatment type. Each configuration's ability to take on incoming runoff volumes, reduce runoff loads, and storage potentials are discussed as a measure of performance. Figure 2.2 compares the hydrologic partitioning of each configuration for every storm while Figure 2.3 and Table 2.1 make comparisons based on averages over the study period.

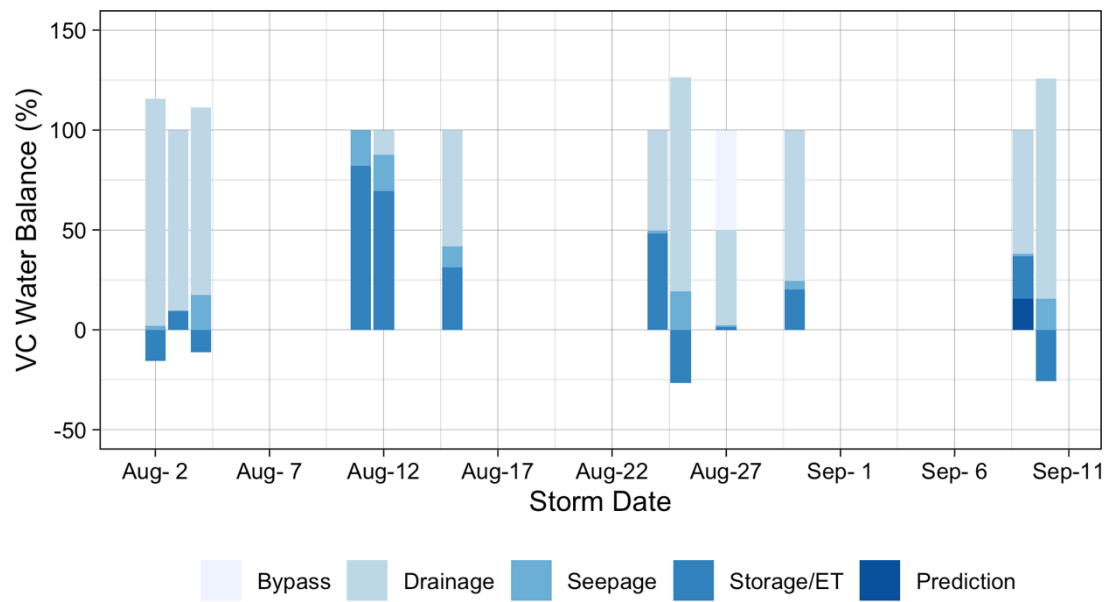
Water Balance

Influent and Bypass

As noted above, columns were dosed with stormwater volumes equivalent to runoff from historical weather data. All columns were able to manage simulated runoff volumes for most events, however one event (2.54 cm, Storm 9) resulted in overflow from the system (designated as Bypass herein - Figure 2.2). The FD configuration had the smallest Bypass percentage (2%) while the IWS configuration had five times the amount (11%). The FD configuration was likely able to take on more water volume than IWS because the outlet structures on IWS systems allow water retention while those on FD systems allow more continuous flow out of the system. This difference in outlet creates more available storage space in FD configurations. Comparison of VC and SM configurations show similar results to IWS, with SM performing slightly better. This difference could be attributed to the SM control system allowing more open drainage for longer

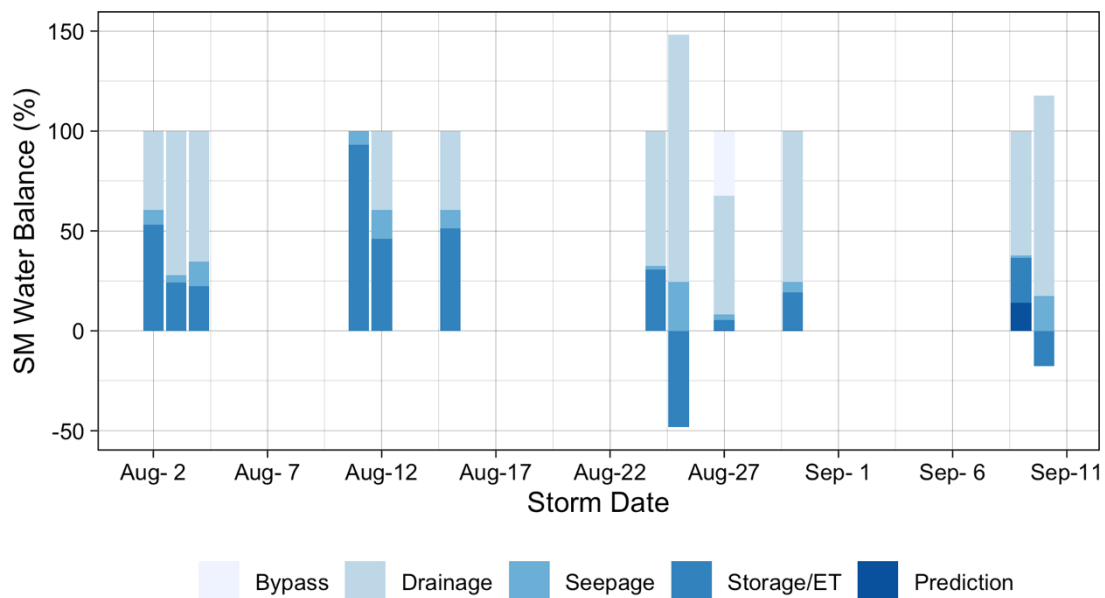


(a)

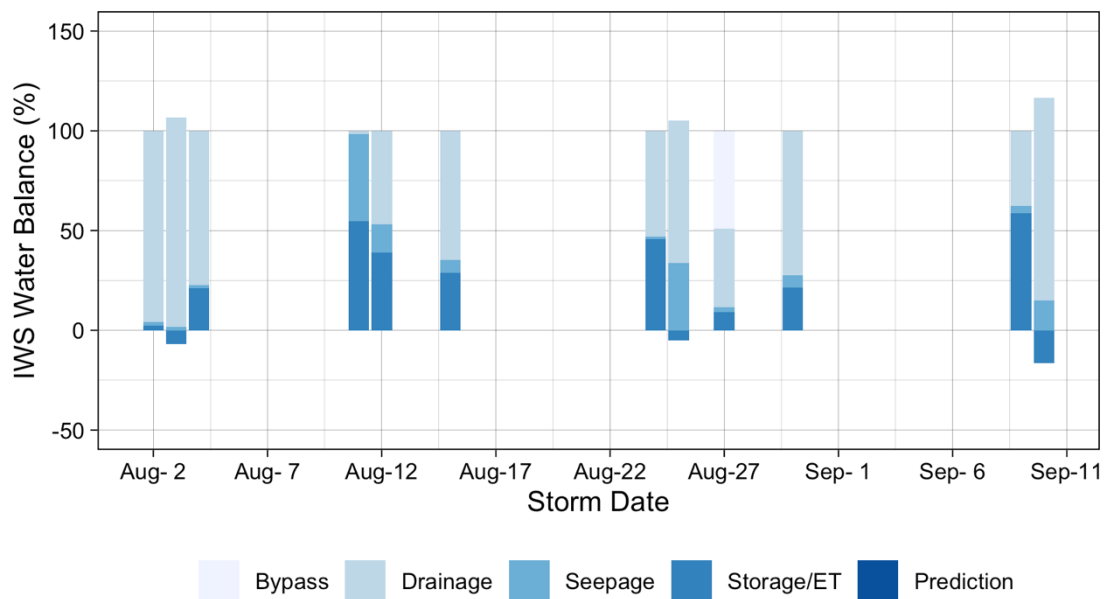


(b)

Figure 2.2 (a) Graph of the size of each rain event in cm. All other figures (b)-(e) are the water balance for each storm event for each configuration over the 6-week study period

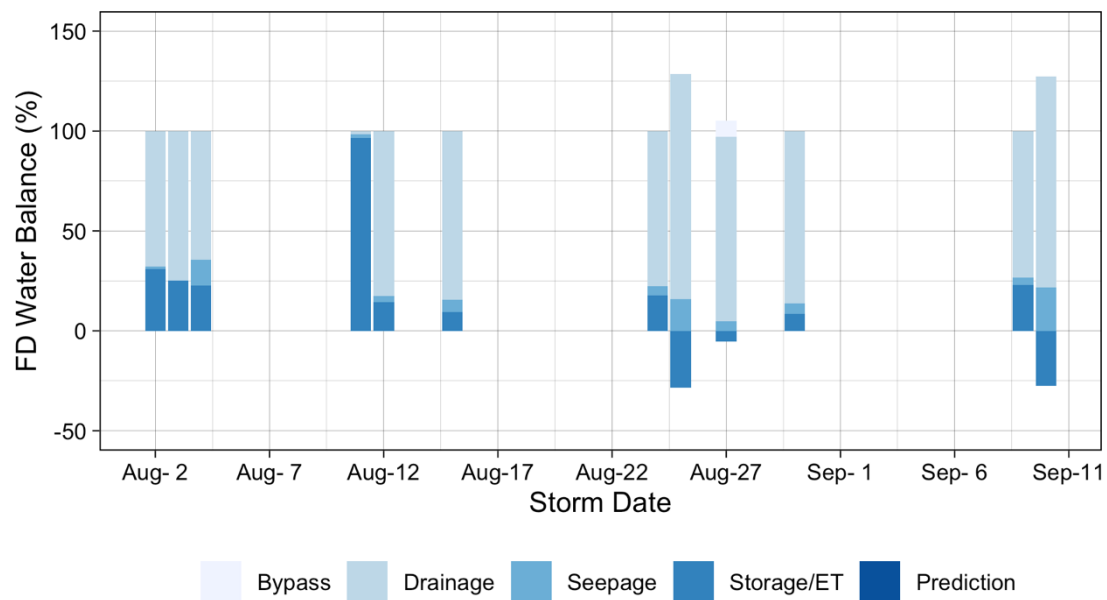


(c)



(d)

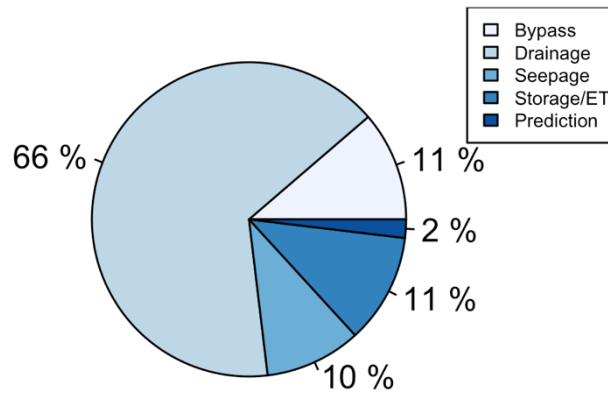
Figure 2.2 Continued



(e)

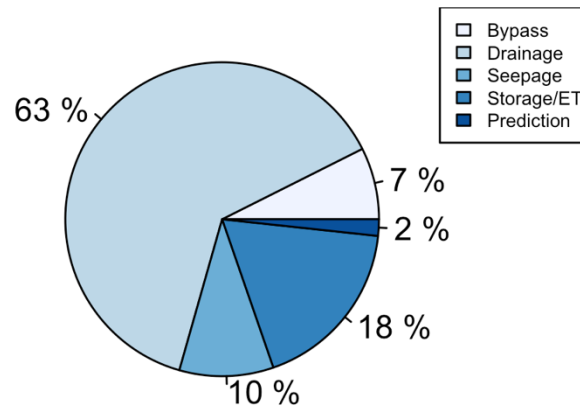
Figure 2.2 Continued

VC



(a)

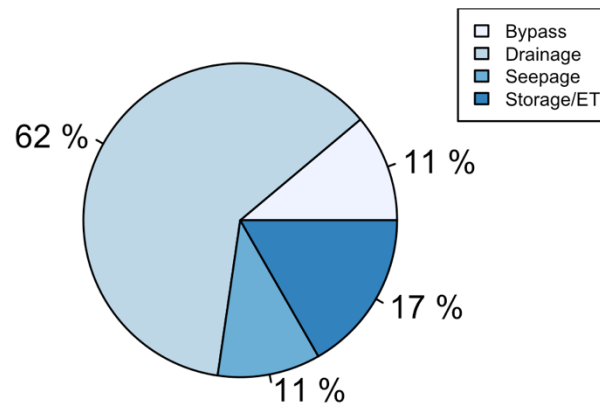
SM



(b)

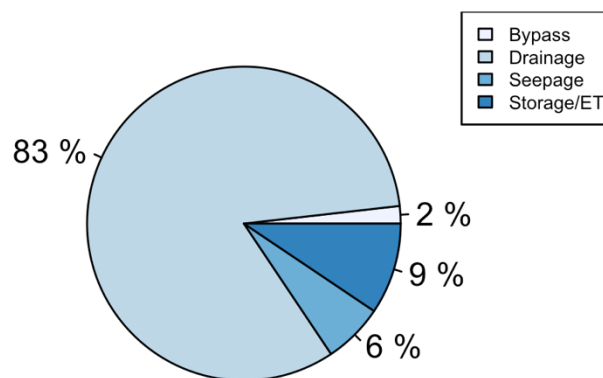
Figure 2.3 Water tracking for each column type based on water balance of the 6-week study period

IWS



(c)

FD



(d)

Figure 2.3 Continued

Table 2.1 Summary of water balance components (%) with standard deviations.

Configuration	Bypass	Drainage	Prediction	Seepage	Storage/ET
VC	11.3 (0.005)	65.6 (0.055)	1.9 (0.013)	9.9 (0.034)	11.3 (0.083)
SM	7.4 (0.051)	63.2 (0.075)	1.7 (0.015)	9.7 (0.039)	18.0 (0.057)
IWS	11.1 (0.000)	61.6 (0.090)	0.0 (0.00)	10.6 (0.072)	16.7 (0.046)
FD	1.8 (0.000)	82.5 (0.035)	0.0 (0.007)	6.2 (0.056)	9.4 (0.064)

periods of time during stormwater application. This allowed more available storage space, and thus less Bypass than VC systems.

Storage/ET

The volume retained was determined by quantifying the amount of stormwater not accounted for in seepage or effluent, effectively equating to Storage and/or ET (Figure 2.2). Although ET rates were not explicitly determined, previous work by (Li et al. 2009) states that 19% of inflow would be lost through evapotranspiration (ET), which is within range of that noted for SM and IWS.

ET is known to vary based on many characteristics of the soil, plants, and environment that measurements are taken in (Ebrahimian et al. 2019; Winston et al. 2016). In examination of the data, FD had substantially less Storage and ET than SM and IWS. This is because the FD configuration is designed to pass water through the system and typically retains less water (Persaud et al. 2019). The RTC treatments showed variable performance, with SM having the greatest storage/ET of all treatments (18%) and VC having storage/ET that was similar to the FD treatment (11% and 9% respectively). This result was unexpected as the VC configuration was designed to mimic IWS in storage capacity, yet IWS showed a higher percentage (16%). The more dynamic nature of the VC control scheme causes rapid release and storage as control valves are opened and closed more frequently.

Individual storm analysis of Storage/ET show more variable results with some storms resulting in negative storage, particularly following larger storms (larger than 1cm). The IWS configuration has the least negative storage which is indicative of better retention. All columns do show trends of negative storage following large, clustered storms which could be a result of an insufficient processing time. An exceeding amount of water would result in more drainage and

further push stored volumes of water out of a system to accommodate incoming stormwater. Systems were then unable to regulate before the 24-hr period intended for processing which ultimately resulted in negative storage (stormwater release) from the system in these events. The system would require greater recovery times to maintain an optimal water storage level. Due to the nature of stormwater application in the study, there is a limit to the volume stored as there is not a continuous flow of simulated stormwater entering the system to allow for gradual recovery. Allowances for differences in intensity may allow VC systems to regulate themselves in a more even manner rather than having to deal with incoming stormwater in a more immediate timeframe.

Seepage

On each column, ports were built into the bottom to simulate seepage into surrounding soils at a rate consistent with that of clay. A field study by Winston et al. (2016) shows bioretention seepage rates in clayey soils between 36 and 59% which is higher than that found in this study. However, compared to field studies, bioretention columns do not allow for lateral seepage out of the system which likely constitutes a substantial amount of loss. Seepage rates are dependent on media depth, exfiltration surface areas, and surrounding subsoils (Hathaway et al. 2014). While seepage rates were set to be the same for each column type, differences in internal conditions resulted in differences in seepage between FD and IWS systems with FD systems having half the seepage volume of IWS with 6% and 11% respectively. This difference is a result of the greater water storage seen in IWS, allowing slow draw down in between storms. Similarly, the RTC treatments allowed water to collect in the bottom of the column (due to closure of the outflow valve) which forced more seepage out of the system. Conversely, the FD configuration allowed

continuous flow from the outlet during storms, reducing water available for seepage in between events.

Changes in Seepage from storm to storm show an increase in Seepage for all column types as events are clustered (Figure 2.2). As water availability increases, an increase in Seepage occurs as events are processed. When looking at differences in Seepage following larger events (ex. Storm 9), Seepage percentage does not increase, rather Drainage increases. Seepage can then be thought of as a more stagnant factor in the hydrologic water balance.

Drainage

The FD configuration had the most drainage (83%) while all other column types performed similarly (ranging from 62 – 66%). Davis et al. (2012) compared FD and IWS bioretention and found performance at 22% and 13% drainage respectively which is far lower than the drainage rates in this study. Similarly, a study by Li et al. (2009), showed IWS resulted in less drainage than FD treatments in clay soils. There are likely multiple reasons for the high drainage rates in this study. Lateral infiltration in a field scale application contributes to greater losses than what is noted in column studies despite research such as Brown and Hunt (2011) which showed similar infiltration rates to what was used in this study. Additionally, the drainage intensity in column studies is likely much higher than that of field studies. Regardless, compared to other column types, the FD configuration encourages efficient drainage resulting in greater drainage volumes. While major differences in drainage volume were not observed between the other column types, drainage time should be considered. Qualitatively, each column type was observed to process stormwater at different rates with the IWS column having the longest time and the FD column having the shortest. The RTC columns performed in a timeframe between that of IWS and FD with

VC processing faster than SM and having faster open/close valve cycles. Differences in timing would cause differences in transfer rates to downstream water bodies. The IWS configuration requires water to pond before discharge, and this slower processing changes internal retention time and subsequently delays drainage. Similarly, the VC and SM treatments experienced more significant time delays between application and drainage as control scheme thresholds had to be met before control valves would open and allow effluent to drain out of the system. Future research should analyze these temporal dynamics which could not be quantified herein.

Analysis of individual storm dynamics show the first three storm events exhibiting excess drainage for VC and IWS systems (Figure 2.2a and Figure 2.2c). Prior to the start of the study, columns were operating based on their control designs and maintained using weekly stormwater applications. Volumes of water stored within these columns may have triggered excess drainage as the systems continued to receive stormwater at the start of the study. This trend continues throughout the study and can be noted in all column types for (Storm 8 and Storm 12). The occurrence of this excess drainage occurring in clustered rain events indicates all systems needing more than 24 hours to process stormwater as continued Drainage was observed. The physical application of rain events influences bioretention performance as longer, low intensity events produce smoother flow patterns and subsequently better hydrologic and water quality processing potentials (Sun et al. 2019). Additional work in analyzing event processing times based on storm size should be done to further understand internal flow processing of these systems. Mimicking rainfall intensity would allow more accurate analysis of hydrologic properties.

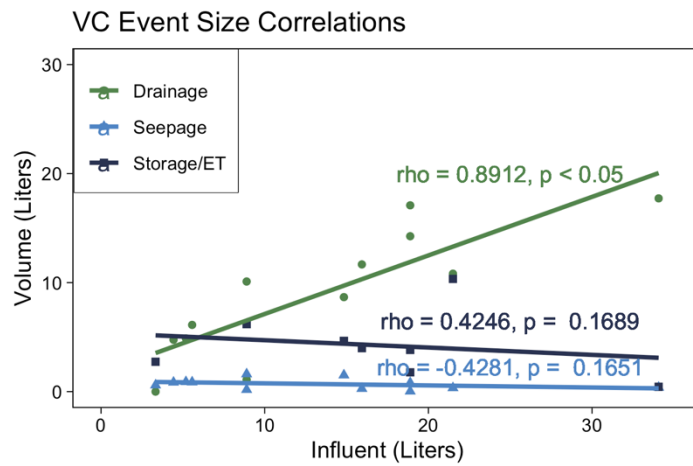
Prediction

As noted above, predictions were sent to columns 12 hours prior to an expected event (whether that event happened or not). Of the 19 predictions, 6 resulted in discharge for the RTC columns, that is, the prediction necessitated discharge to meet its programmed objective. For VC and SM configurations, the volume of effluent collected pre-event (in anticipation of the event) only accounted for 2% of the overall water balance. This was relatively low compared to expectations; thus, further study should be performed to understand rainfall prediction uncertainty and how this may propagate to RTC.

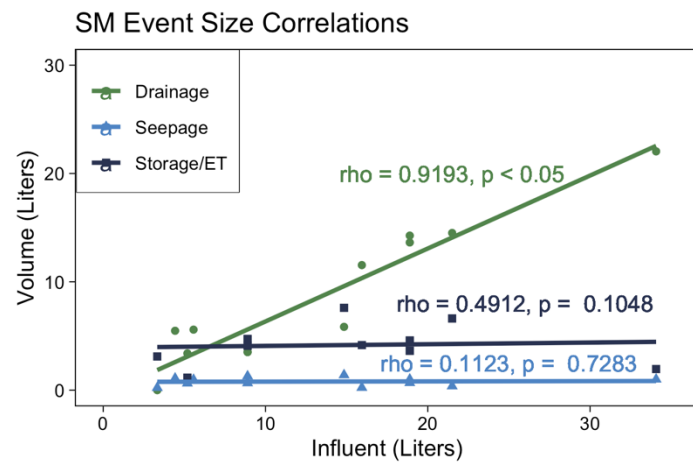
In analysis of Predictions from storm to storm, only one event (Storm 11) showed the largest amount of Prediction drainage. Had other storms been predicted with more accuracy, like for example, Storm 9, more available area within each column would be available for storage which would facilitate less Bypass. This can also be noted for clustered events as flow through a column is not fully facilitated in a 24-hour cycle through traditional methods. Allowing more predictive drainage could result in more storage. The accuracy of storm prediction should be studied further to better facilitate predictive release of water in RTC systems.

Effect of Influent Volume on Partitioning:

Investigation of the impact that event size (Influent) had on Drainage, Seepage and Storage/ET included a Spearman correlation analysis where correlation was characterized as Spearman's ρ greater than 0.5 (Figure 2.4). All configurations show significant positive correlations between Drainage and storm size with FD being more positively correlated than IWS, VC and SM. The physical nature of a FD system does not impede the flow of water through the system and the positive correlation is to be expected. This result is consistent with previous

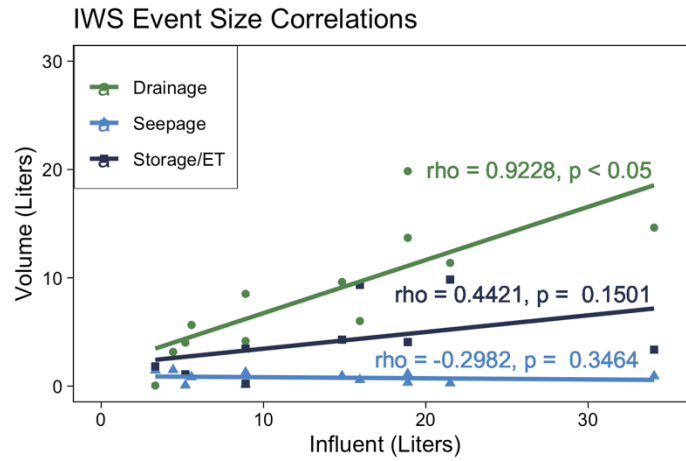


(a)

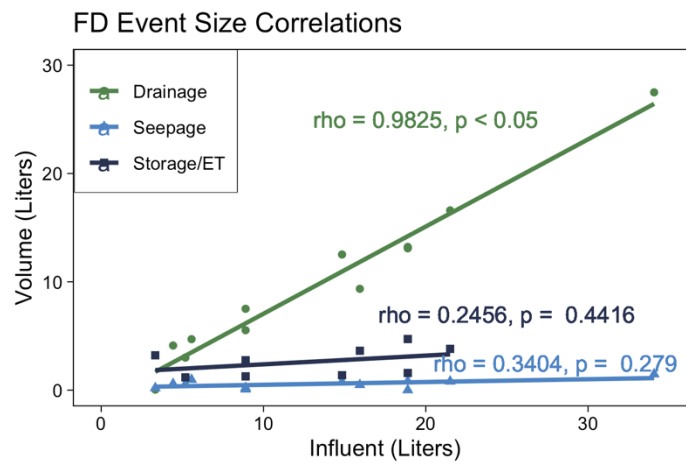


(b)

Figure 2.4 Spearman correlations comparing Drainage, Seepage and Storage/ET to Influent volumes.



(c)



(d)

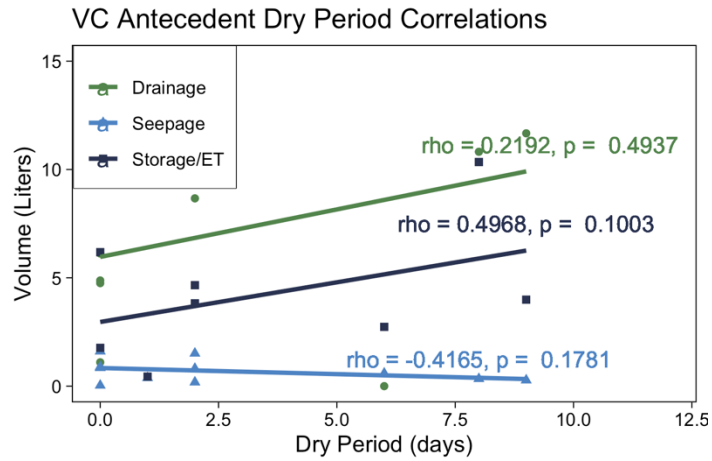
Figure 2.4 Continued

research which documented peak flow mitigation in a practice to be dependent on drainage configuration (Hunt et al. 2012). With the IWS, VC, and SM designs, the lower correlations than FD are expected as flow in these systems was more restricted. Previous research conducted by Wright et al. (2018) supports these results as bioretention cells managing high rainfall events were modeled to decrease peak flows but increased the duration of low flows which suggests more drainage does occur. Effectiveness of bioretention practices in reducing volume and peak flows may be more evident in drier periods where systems are not constantly inundated.

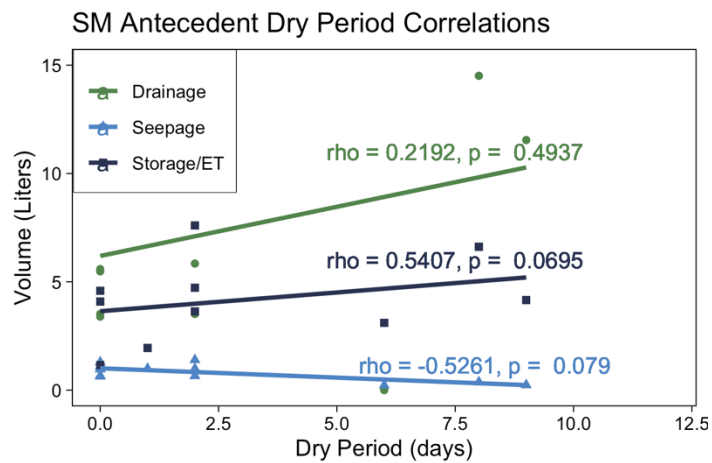
For Seepage and Storage/ET, the trends do not indicate that storm size is influential, with all Spearman's rho values being statistically insignificant ($P > 0.05$). In analysis of the physical nature of each bioretention configuration, Seepage is restricted by infiltration rates which are much lower than the rates of Drainage and so there is a finite amount of volume leaving the system through Seepage. Separate analysis conducted between Seepage and Storage/ET did not show positive Spearman correlations for any column type which is inconsistent with research done by Davis et al. (2009) which states that peak flow mitigation is dependent on infiltration rate. Seepage, which mimics infiltration into surrounding subsoils, is then determined to be a more fixed measure in bioretention cells and does not vary based on configuration or storm size, however, differences in subsoil would alter rates of infiltration. This would then impact peak flow reduction as varying infiltration rates would allow varying degrees of flow impedance.

Effect of Antecedent Dry Period on Partitioning:

Analysis of the influence of dry periods on Drainage, Seepage, and Storage/ET was also conducted using Spearman's correlations (Figure 2.5). No significant correlations were observed for Drainage and Seepage regarding antecedent dry periods, however positive correlations (rho greater than 0.5)

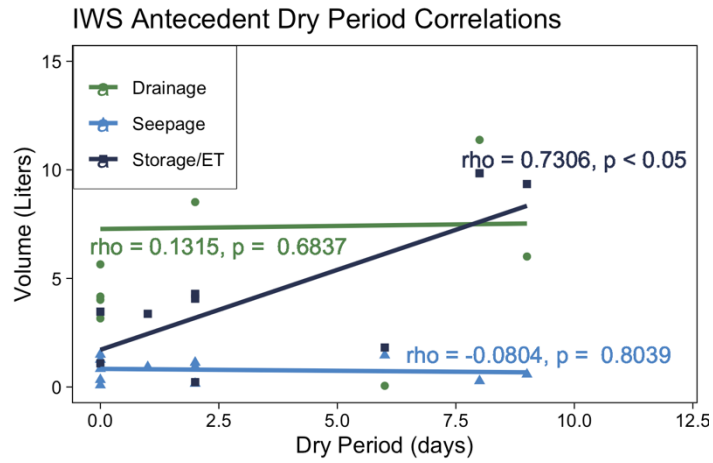


(a)

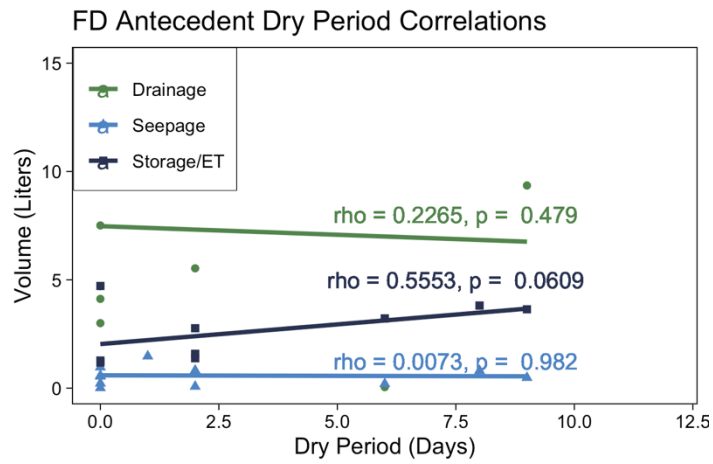


(b)

Figure 2.5 Spearman correlations comparing Drainage, Seepage, and Storage to antecedent dry periods



(c)



(d)

Figure 2.5 Continued

between storage volume and dry periods were observed for the SM, IWS, and FD configurations. This further support conclusions seen in previous research by Mangangka et al. (2015) stating that dry periods lead to lower soil moisture conditions. Available Storage space is then created when for impending influent rain events.

Analysis of correlations between dry periods and volume reduction showed positive correlations (Spearman's rho greater than 0.50) for the VC, SM, and FD configurations while the IWS configuration exhibited high correlations (Spearman's rho greater than 0.75). This result is in line with previous research in which delayed drainage and greater storage potentials are noted by IWS bioretention systems (Liu and Fassman-Beck 2017), as dry periods lead to more storage potential within each configuration. In particular, ET is continuously depleting the stored volume which can contribute to the available storage volume at the time of the next rain event.

Nitrogen Mass Balance

TN mass removal occurred in each configuration with the VC configuration performing the best and FD columns performing the worst (82.3 and 65.6% respectively in Table 2.2). The SM and IWS columns performed similarly with removal being 77.3% and 78.1% respectively. These results do not follow volume reduction trends in which IWS and FD columns were extremes for volume reduction (38.5% and 21.1% reduction respectively), while VC and SM columns performed similarly (35.6% and 32.3% reduction). Differences in nitrogen removal and volume reduction show that hydrologic losses are not the only removal pathway, and that biogeochemical processing within each treatment type affects nutrient removal. It should also be noted that the facilitation of saturated zones promotes more nitrogen removal, which is also promoted in previous research (Barrett et al. 2013; Henderson et al. 2007; Read et al. 2008). Of the two RTC designs,

Table 2.2 Volume reduction and TN removal as %. TN mean, median, Standard Deviation (Std Dev), and Relative Standard Deviation (RSD) also listed.

Type	Volume Reduction (%)	TN Mass Removal (%)	TN Median (mg)	TN Mean (mg)	TN Std Dev	TN RSD (%)
VC	35.6	82.3	6.4	7.2	5.39	75.4
SM	32.3	77.3	8.4	9.2	7.90	86.1
IWS	38.6	78.1	6.9	8.8	7.17	81.2
FD	20.1	65.6	11.8	13.9	13.02	93.8

the VC configuration performed best as it had the highest nutrient reductions which is a result of the creation of both aerobic and anaerobic environments within the system (Persaud et al. 2019). The SM column performed similarly to IWS which also indicates the facilitation of aerobic and anaerobic processing. The FD column had the least amount of nitrogen removal which was expected as the system is not designed for water retention and is primarily aerobic in processing.

As noted above, it was assumed that Prediction effluent has the same concentration as the effluent collected after a storm event (Drainage). This is a potential source of error; however, this hydrologic pathway only accounted for 2% of the water balance. Thus, even if these concentrations were slightly different, there would not be a significant change in overall effluent mass. This is an interesting observation in the context of RTC for bioretention, as it indicates preemptive water release may not greatly affect performance. In regard to the mass of nitrogen infiltrated, these values are not included in the water balance as they are assumed to be lost by the system and removed by groundwater.

Conclusions and Recommendations

A column study was conducted over a six-week period to compare hydrologic performance of RTC and traditional bioretention designs through a water balance. Traditional bioretention treatments (FD and IWS) showed different partitioning for every portion of the water balance which further highlights the impact of their design differences. FD systems were originally designed to manage stormwater volumes and reduce peak flows. Having the lowest Bypass rate of 2%, FD displayed its ability to manage larger stormwater volumes compared to IWS systems which had an 11% Bypass rate. However, in terms of storage/ET, the FD systems only had a 9% storage rate while IWS had 16%. This is an artifact of their outlet configuration and the role of a

submerged zone. As IWS was created to promote water quality benefits, the storage within the system reflects the saturated zone created when using this practice. Finally, when comparing seepage rates, IWS doubles the seepage of FD which indicates the greater groundwater recharge potential of the practice.

Of the two RTC treatments, SM treatment was better performing with similar Storage/ET to IWS, 18% and 16% respectively, as compared to the storage of VC (11%). Further, there was 7% Bypass from the system when compared to VC (11%), indicating a greater ability of this system to take on larger stormwater volumes. Compared to traditional practices, the SM treatment could lend itself to becoming a more functional design practice than IWS and FD systems. Considering the water quality benefits of this practice seen in research by Persaud et al. (2019), the SM control scheme could be a reasonably well performing system as it manages larger stormwater volumes while also promoting nitrification and denitrification. Analysis of TN in this research does show greater reductions for the VC treatments and it should be noted that the VC control in this study differs from that in previous studies. The improved TN reduction could indicate better facilitation of nitrification and denitrification within the system than that observed by Persaud et al. (2019). Compared to the SM control, the VC control lacked the storage and volume capabilities. It is noteworthy that in both VC and SM configurations, effluent caused by predicted drainage contributes to a very small percentage of the water balance. As such omitting this preemptive control could simplify RTC designs. Greater focus on environmental stressors in future optimization would allow more targeted treatment dynamics whether that be for hydrologic or water quality purposes.

Future study of field scale applications of RTC schemes should be performed to understand larger implications of their design and determine feasibility in implementation. Further, additional

studies on the drainage times of traditional and RTC system should be compared to understand the importance of internal processing time (i.e., what hydrologic delays are provided and if they are influential on receiving streams). In terms of nutrient processing, optimizing storage and release could make significant differences in effluent water quality. With constantly variable weather conditions, continuous adaptation to accommodate stormwater flows from frequent events, or moisture level maintenance during dry periods can ensure water quality standards are being met. Finally, using this technology could create a more connected network of stormwater management, where water quality, groundwater recharge, and volume reduction are optimized through the use of combined systems in an actively controlled watershed.

Acknowledgments

The authors would like to thank the University of Tennessee Civil Engineering workshop, Aaron Akin of the University of Tennessee, and the East Tennessee AgResearch and Education Center who were all instrumental in completing this work. This research was funded by the National Science Foundation (NSF Grant number 1737432) and the Institute for a Secure and Sustainable Environment at the University of Tennessee.

References

- Barrett, M. E., M. Limouzin, and D. F. Lawler. 2013. 'Effects of Media and Plant Selection on Biofiltration Performance', *Journal of Environmental Engineering-Asce*, 139: 462-70.
- Bartos, M., B. Wong, and B. Kerkez. 2018. 'Open storm: a complete framework for sensing and control of urban watersheds', *Environmental Science-Water Research & Technology*, 4: 346-58.
- Bell, C. D., J. M. Wolfand, C. L. Panos, A. S. Bhaskar, R. L. Gilliom, T. S. Hogue, K. G. Hopkins, and A. J. Jefferson. 2020. 'Stormwater control impacts on runoff volume and peak flow: A meta-analysis of watershed modelling studies', *Hydrological Processes*, 34: 3134-52.
- Brabec, E., S. Schulte, and P. L. Richards. 2002. 'Impervious surfaces and water quality: A review of current literature and its implications for watershed planning', *Journal of Planning Literature*, 16: 499-514.
- Bratieres, K., T. D. Fletcher, A. Deletic, and Y. Zinger. 2008. 'Nutrient and sediment removal by stormwater biofilters: A large-scale design optimisation study', *Water Research*, 42: 3930-40.
- Bronstert, A., D. Niehoff, and G. Burger. 2002. 'Effects of climate and land-use change on storm runoff generation: present knowledge and modelling capabilities', *Hydrological Processes*, 16: 509-29.
- Brown, R. A., and W. F. Hunt. 2011. 'Underdrain Configuration to Enhance Bioretention Exfiltration to Reduce Pollutant Loads', *Journal of Environmental Engineering-Asce*, 137: 1082-91.
- Burns, M. J., T. D. Fletcher, C. J. Walsh, A. R. Ladson, and B. E. Hatt. 2012. 'Hydrologic shortcomings of conventional urban stormwater management and opportunities for reform', *Landscape and Urban Planning*, 105: 230-40.
- CSN. 2013. "Bioretention Illustrated: A Visual Guide for Constructing, Inspecting, Maintaining and Verifying the Bioretention Practice." In. Prince William County, VA: Chesapeake Stormwater Network.

- Davis, A. P., W. F. Hunt, R. G. Traver, and M. Clar. 2009. 'Bioretention Technology: Overview of Current Practice and Future Needs', *Journal of Environmental Engineering*, 135: 109-17.
- Davis, A. P., R. G. Traver, W. F. Hunt, R. Lee, R. A. Brown, and J. M. Olszewski. 2012. 'Hydrologic Performance of Bioretention Storm-Water Control Measures', *Journal of Hydrologic Engineering*, 17: 604-14.
- Dietz, M. E., and J. C. Clausen. 2005. 'A field evaluation of rain garden flow and pollutant treatment', *Water Air and Soil Pollution*, 167: 123-38.
- Ebrahimian, A., B. Wadzuk, and R. Traver. 2019. 'Evapotranspiration in green stormwater infrastructure systems', *Science of the Total Environment*, 688: 797-810.
- Fletcher, T. D., W. Shuster, W. F. Hunt, R. Ashley, D. Butler, S. Arthur, S. Trowsdale, S. Barraud, A. Semadeni-Davies, J. L. Bertrand-Krajewski, P. S. Mikkelsen, G. Rivard, M. Uhl, D. Dagenais, and M. Viklander. 2015. 'SUDS, LID, BMPs, WSUD and more - The evolution and application of terminology surrounding urban drainage', *Urban Water Journal*, 12: 525-42.
- Hamel, P., and T. D. Fletcher. 2014. 'The impact of stormwater source-control strategies on the (low) flow regime of urban catchments', *Water Science and Technology*, 69: 739-45.
- Hathaway, J. M., R. A. Brown, J. S. Fu, and W. F. Hunt. 2014. 'Bioretention function under climate change scenarios in North Carolina, USA', *Journal of Hydrology*, 519: 503-11.
- Henderson, C., M. Greenway, and I. Phillips. 2007. 'Removal of dissolved nitrogen, phosphorus and carbon from stormwater by biofiltration mesocosms', *Water Science and Technology*, 55: 183-91.
- Hopkins, K. G., A. S. Bhaskar, S. A. Woznicki, and R. M. Fanelli. 2019. 'Changes in event-based streamflow magnitude and timing after suburban development with infiltration-based stormwater management', *Hydrological Processes*: 17.

- Hunt, W. F., A. P. Davis, and R. G. Traver. 2012. 'Meeting Hydrologic and Water Quality Goals through Targeted Bioretention Design', *Journal of Environmental Engineering-Asce*, 138: 698-707.
- Hunt, W. F., J. T. Smith, S. J. Jadlocki, J. M. Hathaway, and P. R. Eubanks. 2008. 'Pollutant removal and peak flow mitigation by a bioretention cell in urban Charlotte, NC', *Journal of Environmental Engineering-Asce*, 134: 403-08.
- Juan, A., C. Hughes, Z. Fang, and P. Bedient. 2017. 'Hydrologic Performance of Watershed-Scale Low-Impact Development in a High-Intensity Rainfall Region', *Journal of Irrigation and Drainage Engineering*, 143: 11.
- Kerkez, B., M. Daniels, S. Graves, V. Chandrasekar, K. Keiser, C. Martin, M. Dye, M. Maskey, and F. Vernon. 2016. 'Cloud Hosted Real-time Data Services for the Geosciences (CHORDS)', *Geoscience Data Journal*, 3: 4-8.
- Kerkez, B., C. Gruden, M. Lewis, L. Montestruque, M. Quigley, B. Wong, A. Bedig, R. Kertesz, T. Braun, O. Cadwalader, A. Poresky, and C. Pak. 2016. 'Smarter Stormwater Systems', *Environmental Science & Technology*, 50: 7267-73.
- Konrad, Christopher P. 2003. "Effects of Urban Development on Floods." In.: USGS.
- Li, H., L. J. Sharkey, W. F. Hunt, and A. P. Davis. 2009. 'Mitigation of Impervious Surface Hydrology Using Bioretention in North Carolina and Maryland', *Journal of Hydrologic Engineering*, 14: 407-15.
- Liu, J., D. J. Sample, C. Bell, and Y. T. Guan. 2014. 'Review and Research Needs of Bioretention Used for the Treatment of Urban Stormwater', *Water*, 6: 1069-99.
- Liu, R. F., and E. Fassman-Beck. 2017. 'Hydrologic experiments and modeling of two laboratory bioretention systems under different boundary conditions', *Frontiers of Environmental Science & Engineering*, 11: 10.

- Mangangka, I. R., A. Liu, P. Egodawatta, and A. Goonetilleke. 2015. 'Performance characterisation of a stormwater treatment bioretention basin', *Journal of Environmental Management*, 150: 173-78.
- Manka, B. N., J. M. Hathaway, R. A. Tirpak, Q. He, and W. F. Hunt. 2016. 'Driving forces of effluent nutrient variability in field scale bioretention', *Ecological Engineering*, 94: 622-28.
- Middleton, J. R., and M. E. Barrett. 2008. 'Water quality performance of a batch-type stormwater detention basin', *Water Environment Research*, 80: 172-78.
- Mullapudi, A., M. Bartos, B. Wong, and B. Kerkez. 2018. 'Shaping Streamflow Using a Real-Time Stormwater Control Network', *Sensors*, 18: 11.
- Parolari, A. J., S. Pelrine, and M. S. Bartlett. 2018. 'Stochastic water balance dynamics of passive and controlled stormwater basins', *Advances in Water Resources*, 122: 328-39.
- Persaud, P. P., A. A. Akin, B. Kerkez, D. T. McCarthy, and J. M. Hathaway. 2019. 'Real time control schemes for improving water quality from bioretention cells', *Blue-Green Systems*, 1: 55-71.
- Read, J., T. Wevill, T. Fletcher, and A. Deletic. 2008. 'Variation among plant species on pollutant removal from stormwater non biofiltration systems', *Water Research*, 42: 893-902.
- Roman, D., A. Braga, N. Shetty, and P. Culligan. 2017. 'Design and Modeling of an Adaptively Controlled Rainwater Harvesting System', *Water*, 9: 14.
- Sun, Y. W., C. Pomeroy, Q. Y. Li, and C. D. Xu. 2019. 'Impacts of rainfall and catchment characteristics on bioretention cell performance', *Water Science and Engineering*, 12: 98-107.
- TDEC. 2014. "Tennessee Permanent Stormwater Management and Design Guidance Manual." In, 141-76.
- TSM, Tennessee Stormwater Management. 2015. "Management and Design Guidance Manual." In *Tennessee Stormwater Managment*.

- Waller, L. J., G. K. Evanylo, L. A. H. Krometis, M. S. Strickland, T. Wynn-Thompson, and B. D. Badgley. 2018. 'Engineered and Environmental Controls of Microbial Denitrification in Established Bioretention Cells', *Environmental Science & Technology*, 52: 5358-66.
- Wang, M., D. Q. Zhang, S. W. Lou, Q. H. Hou, Y. J. Liu, Y. N. Cheng, J. D. Qi, and S. K. Tan. 2019. 'Assessing Hydrological Effects of Bioretention Cells for Urban Stormwater Runoff in Response to Climatic Changes', *Water*, 11: 20.
- Winston, R. J., J. D. Dorsey, and W. F. Hunt. 2016. 'Quantifying volume reduction and peak flow mitigation for three bioretention cells in clay soils in northeast Ohio', *Science of the Total Environment*, 553: 83-95.
- Wright, O. M., E. Istanbuluoglu, R. R. Horner, C. L. DeGasperi, and J. Simmonds. 2018. 'Is there a limit to bioretention effectiveness? Evaluation of stormwater bioretention treatment using a lumped urban ecohydrologic model and ecologically based design criteria', *Hydrological Processes*, 32: 2318-34.

Chapter 3 : Real-Time Control Influence on Microbial Functionality in Bioretention

P.P. Persaud¹, and J.M. Hathaway¹, D.T. McCarthy², M.F. Campa¹, T. Hazen¹

¹ Dept. of Civil and Environmental Engineering, University of Tennessee, 325 John D. Tickle Building, 851 Neyland Dr., Knoxville, TN, 37996 USA

² Environmental and Public Health Microbiology Lab (EPHM Lab), Dept. of Civil Engineering, Monash University, Melbourne, VIC, AUS

*Corresponding author: hathaway@utk.edu

Primary Research question:

What differences in microbial populations exist between real-time controlled, traditional, and internal water storage bioretention systems?

Abstract

Bioretention practices shift urban hydrology toward more natural regimes while also providing water quality improvements through physical, chemical, and biological processing. Variations in physical design components have been shown to influence water quality, with some of this influence being tracked to microbial populations in design variations such as free draining (FD) and internal water storage (IWS). A six-week column study was conducted to build on this knowledge by comparing traditional, gravity controlled bioretention to real-time control (RTC) designs. Specifically, this work aims to investigate microbial populations related to nutrient processes in bioretention and how they are affected by cyberphysical design variations, a unique application of RTC. Water samples were analyzed for nitrogen species changes from inlet to outlet, and differences in microbial communities to understand the influences that RTC has on nitrifying and denitrifying microbial populations. Spearman correlations were also calculated to determine

significant relationships between nutrient removal, microbial abundance, and environmental variables. As expected, based on previous studies, the free draining systems better promoted aerobic process, the internal water storage treatments better promoted anaerobic processes, while the RTC treatments were able to balance these two extremes. Results from this study show that the more dynamic environmental conditions created by RTC treatments improved nutrient removal which was also reflected in microbial abundances when specifically analyzing *Nitrospira* and *Pseudomonas* genera (two species known to take part in nitrification and denitrification). Furthermore, this research suggests that targeting specific processing or environmental conditions, like aerobic and anaerobic zone creation, or soil moisture conditions, can also influence removal rates and bacterial populations. Further research using functional gene data should be conducted to allow an understanding as to the activity of nitrogen processing bacteria, and thus the degree to which they are actually contributing to nitrogen removal.

Introduction

Expanding urbanization has caused an increase in impervious areas, runoff, and chemical inputs into natural systems (Konrad 2003; Li et al. 2019). To combat this, green infrastructure practices, like bioretention, have been developed to mitigate runoff and water quality impacts (Fletcher et al. 2015). Bioretention uses sand-based media to infiltrate and filter runoff, while biogeochemical processes remove pollutants such as nutrients from stormwater (TDEC 2014). Along with plants, microbes play a critical role in nutrient conversions. Of particular importance are nitrogen compounds (coming from fertilizers, fuel, leaking sewer lines, etc.) which cause eutrophication and the release of nitrous oxide (N_2O), a greenhouse gas (Cameron et al. 2013; Bosch et al. 2018; Zhang et al. 2019).

Processing through the nitrogen cycle requires microbes to perform mineralization, aerobic, and anaerobic processing and convert nitrogen compounds like ammonium (NH_4^+) and nitrate (NO_3^-) to nitrogen gas (N_2) (Zumft 1997; Madigan 2019). Incomplete denitrification and nitrate leaching have been observed in previous literature, and research attempting to rectify this issue has focused on physical aspects of bioretention design like plant presence/absence, media variations, stratification, and inclusion of a saturated zone; however, information regarding microbial processing is comparatively limited (Glaister et al. 2017; Dagenais et al. 2018; Waller et al. 2018; Chen et al. 2019; Jay et al. 2019). Plants are the primary removal mechanism for nitrogen; however, denitrifying bacteria have a greater potential to process higher concentrations of nitrogen (Laurenson et al. 2013; Morse et al. 2018). This is because plant uptake capacity can be exhausted when dealing with high nutrient loads (Payne et al. 2014). Plant die off within a practice can also add sequestered nitrogen back into the system (Chen et al. 2013). Bioretention practices have also been shown to experience incomplete denitrification leading to additional NO_3^- inputs into watersheds (Mangangka et al. 2015). As a result, additional research is needed to inform nutrient processing in bioretention, in particular, to explore methods for minimizing incomplete denitrification.

Various environmental conditions are known to promote microbial growth and activity (Passeport et al. 2009). Specific to bioretention, the design of a cell has been shown to affect bacteria more than environmental conditions (e.g., temperature and event size) would as vertical stratification, media composition, and nutrient sources promote growth of aerobic and anaerobic bacteria (Morse et al. 2018; Waller et al. 2018). The two most common design configurations are free draining (FD), and those that incorporate an internal water storage (IWS) zone. Differences between the two are that FD creates predominantly aerobic conditions, while IWS creates both

aerobic and anaerobic conditions within a cell (provided in situ soils have a low enough hydraulic conductivity to allow extended ponding) (Dietz and Clausen 2005; Li et al. 2014; Waller et al. 2018). Hydrologically, FD configurations have greater available storage but also lose more water to underdrains; conversely, IWS systems are limited to the available storage volume above the saturated zone but have relatively smaller losses to underdrains (Chapter 3, Persaud Dissertation?). Differences in losses to infiltration between the two configurations are substantially influenced by the hydraulic conductivity of surrounding soils which, as noted above, greatly affects the degree to which anaerobic conditions are supported. However, examination of previous research shows that IWS can enhance denitrifying bacteria in these systems which further influences nutrient processing (Igielski et al. 2019). Despite differences between these systems, they are both gravity-driven and perform in a static manner once constructed, that is, their function does not adapt based on conditions.

Recent advances in bioretention cell design have involved the use of real-time control (RTC) technology (Shen et al. 2020). Such dynamic, adaptable designs have the potential to improve water quality by actively balancing system conditions and creating an optimized function between the extremes of FD and IWS systems (Persaud et al. 2019). Other previous work using real-time control technology for stormwater controls has largely focused on hydrologic processes with the goal to help mitigate effects such as flooding (Kerkez et al. 2016; Bartos et al. 2018; Mullapudi et al. 2018). To a lesser degree, research has also been performed to investigate some aspects of water quality improvement and conservation in green infrastructure such as detention basins and rainwater harvesting systems (Middleton and Barrett 2008; Roman et al. 2017) however, using RTC in bioretention is an unexplored context thus far in literature. Furthermore,

analyzing microbial community shifts due to RTC, and understanding how this may lead to changes in water quality performance of the system has yet to be performed.

This work will build on existing literature by documenting nitrogen transformations and removal in active control designs and compared these results to more traditional, static FD and IWS bioretention designs. These observations will be supported by investigating variations in the nitrifying and denitrifying microbial communities over the six-week column study. Specifically, this research will: 1) document differences in nitrogen transformations and removal between treatments, 2) examine the variance in microbial population types between treatments, 3) identify conditions promoting denitrifying bacteria through the use of RTC, and 4) couple microbial and water quality data to understand nutrient dynamics within bioretention and how they impact effluent water quality. These outcomes will allow suggestions as to how real-time control schemes may be modified to allow enhanced microbial function within bioretention practices.

Materials and Methods

Column Design

The bioretention columns were built in the four configurations outlined in Persaud et al. (2019). In short, traditional FD and IWS columns were accompanied by two real-time control treatments, all with five replications. Soil moisture (SM) control columns were designed to maintain media in the columns at field-capacity, determined specifically for the bioretention media utilized in the study by laboratory testing in which soils were saturated and allowed to drain for 48 hours while being constantly monitored. Water removed from the soil profile was only removed by gravity and soil moisture readings after 48 hours were used for field capacity. Volume control (VC) columns were designed to maintain a saturated zone below a prescribed depth. Both RTC

treatments were programmed to preemptively empty stored water as needed to accommodate incoming rainfall and create the desired conditions post-event (e.g., allow discharge of stored water such that the incoming water would refill the column to the targeted physical condition). The design of the VC operation in the initial column study conducted in 2018 was altered to prevent the entrance of oxygen into the lower layers of the column (Persaud et al. 2019). The VC storage depth was set to 29 cm with an operating range of (+/-) 15 cm. Reducers were also affixed to the outlets of the VC columns to allow a slower release of water. This allowed the system to better adjust within the given range of storage rather than experiencing a repeated high velocity drainage.

Experimental Procedure

During the Summer of 2019, a six-week column study was completed in a climate-controlled greenhouse with an average temperature of 28°C. To allow realistic function of the RTC systems (i.e. preemptive action based on predicted rainfall), a historic 6-week period of rainfall from August 8th to September 19th, 2018, was mimicked such that both rainfall and rainfall prediction data could be gathered. Rainfall data and historic rainfall predictions were obtained from the National Oceanic and Atmospheric Administration (NOAA) archive. Rainfall data informed dosing times and volumes based on the assumption of a 1:20 ratio of practice area to watershed area (TSM 2015). Rainfall predictions were sent to RTC columns 12 hours prior to a simulated rainfall event, allowing each column to adjust to expected rainfall based on their preprogrammed algorithm. The columns also became reactive systems and adjusted during and after an event to meet their objective soil moisture or level threshold. This procedure is similar to that outlined in Persaud et al. (2019), only being altered in regard to the VC configuration operation and the rainfall time period utilized. Columns were dosed with nutrient concentrations in a

synthetic stormwater mixture outlined by Bratieres et al. (2008). The mixture was made with dechlorinated tap water and chemical additions to meet necessary concentration goals (Table 3.1).

At the start of each storm event, samples were collected from the stormwater mixing tank to serve as an “influent” sample. Artificial stormwater was dosed to the columns in three passes to allow more homogenous distribution between columns. Following each prescribed storm event, columns were allowed to process stormwater for 24 hours after the event before a sample was collected. Samples were collected from the effluent of each bioretention column, and a control collection bin in the greenhouse. The control bin was filled with autoclaved deionized water to understand background microbial influences as the greenhouse was not a sterile environment.

Water Quality Sampling

Samples were prepared for water quality analysis by filtering them through 0.45 μm Whatman disposable filters. After filtering, samples were refrigerated, and analysis was completed within 30 days of collection. Samples were analyzed for nutrient (NO_2^- -N, NO_3^- -N, and NH_4^+ -N) concentrations using ion chromatography in the University of Tennessee Water Quality Core Facility. Total Kjeldahl Nitrogen (TKN) for each sample was tested at Microbac Laboratories using EPA reference method 351.2. Total Organic Nitrogen (TON) and Total Nitrogen (TN) were also calculated mathematically using the following formulas:

$$\text{TON} = \text{TKN} - \text{NH}_4^+\text{-N}$$

$$\text{TN} = \text{TKN} + \text{NO}_3^-\text{-N} + \text{NO}_2^-\text{-N}$$

$$\text{TN} = \text{TON} + \text{NH}_4^+\text{-N} + \text{NO}_3^-\text{-N} + \text{NO}_2^-\text{-N}$$

Table 3.1 Stormwater target concentrations and contributing chemical additions

Pollutant	Target Concentration (mg/L)	Chemical
NO _x -N	0.75	KNO ₃
NH ₄ ⁺ -N	0.27	NH ₄ Cl
TDP	0.04	KH ₂ PO ₄
Cu ²⁺	0.05	CuSO ₄
Pb ²⁺	0.14	PbSO ₄
Mn ²⁺	0.25	MnCl*4H ₂ O
Zn ²⁺	0.25	ZnCl ₂
Cd ²⁺	0.0045	Cd ²⁺ Std. Soln.
Ni ²⁺	0.03	Nicotinic Acid
Fe ³⁺	1	FeSO ₄ *7H ₂ O
Cr ⁶⁺	0.025	Cr ⁶⁺ Std. Soln.

DNA Preparation and Analysis

Samples were filtered through 0.22µm Sterivex filters and stored at -20 °C. DNA extraction was conducted using Qiagen DNeasy PowerSoil Kits following methods specified by the manufacturer. An important note is the greenhouse environment was not sterile based on samples taken from a reference collection bin to represent the background microbial abundance present in the greenhouse (GHBKGD). Libraries were prepared using the NEBNext Library Quant kit from Illumina and purified using the Zymo Research Corporation Select-a-Size DNA Clean & Concentrator kit following the respective manufacturer's instructions. The Invitrogen Quant-iT Qubit dsDNA BR Assay kit was used with the Qubit Fluorometer to quantify samples per the manufacturer's instructions and sized using a Bioanalyzer 2100. Libraries were prepared for sequencing using a MiSeq Reagent Kit for metagenomics libraries and a MiSeq V2 (300 cycle) Reagent Kit according to manufacturer's instructions for 16S amplicon libraries. The PhiX Control V3 library by Illumina was used as a control.

Analysis of the entire data set was conducted to determine differences in general abundance and variability. Filtering analysis was then done to support and better explain the water quality trends based on microbes known to support nitrifying and denitrifying processes. First a list of 24 nitrifying and denitrifying bacteria were compiled based on previous literature and used to identify bacteria that were present in the data set (Pajares and Bohannon 2016; Chen et al. 2013; Daims et al. 2015; Morse et al. 2018; Zumft 1997; Payne et al. 2014; Zhou et al. 2016). The list was then pared down based on genus filtering – meaning only genera that showed abundance were included in the analysis. The final list of genera of interest showed 7 common genera, however analysis was done using the two most predominant representatives which was similar to those seen in previous literature by (Morse et al. 2018; Payne et al. 2014). This taxa representative of nitrifying and

denitrifying bacteria is summarized in results. The overall list of bacteria can be found in [supplementary material](#).

Statistical Analysis

Genomic data analysis was performed using QIIME II and the vegan and phyloseq package in R. Silva feature classifiers were used as a reference data set. Abundance data was rarefied prior to analysis and Alpha and Beta diversity analysis was done to determine differences between sample types. Alpha diversity is used to determine population variability within one sample, meaning how many different populations are present within one sample. Measures of alpha diversity range from 0 to infinity and are shown in this research as Shannon diversity. Lower Shannon diversity scores indicate a sample with fewer species present while higher scores would be indicative of rich abundance of many different types of microbiomes. Beta diversity analyzes samples against each other. More specifically, this is an indicator of how much samples vary. Samples clustered together are more similar to each other. In this research, weighted and unweighted unifracs PCoA plots are presented to show clustering trends in the data. Weighted unifracs PCoA plots take abundance into account when determining distances in clustering while unweighted unifracs plots are based only on sequence distance. While both plots are presented, the weighted unifracs PCoA is used in analysis as relative abundance and sequence length are accounted for.

Water quality data were first evaluated using Kolmogorov-Smirnov tests for normality and confirmed the non-normal distribution of data and the necessity for non-parametric tests. Microbial populations and water quality comparisons between treatment types were made using Wilcoxon sign-rank tests. Correlation analysis using Spearman's rho was also performed to determine if

correlations existed between nutrient removal, microbial abundances, soil moisture levels in the upper and lower portions of each column type, and wet and dry periods.

Results

Water quality and 16S data were collected for each storm over the 6-week study period for a total of 420 samples. Of these samples, 265 were used in water quality analysis while 155 were used for 16S analysis. Figure 3.1 shows the applied rain events over the 6-week study period. storms ranged in size from 0.23 cm to 2.54 cm with a median of 0.81 cm while dry periods in between events ranged from 0 to 9 days with a median of 1.5 days. Analysis of water quality and microbial data includes discussion about the influence of wet and dry periods.

Water Quality Results

Analysis of influent and effluent concentrations of Nitrogen species is shown in Figure 3.2 and Table 3.2. These graphs show the concentration differences between the influent and effluent for each column type and differences in performance are indicators of steps in the nitrogen cycle that are facilitated by each column configuration. For $\text{NH}_4^+\text{-N}$ Figure 3.2a, similar trends in removal as those from Persaud et al. (2019) were observed, with VC and SM columns having the greatest removal and IWS columns having the least. SM columns performed similarly to VC (82.5% and 84.7% removal respectively) while the FD configuration performed better than the IWS configuration (68.3% and 54.4% removal respectively). The performance by the traditional configurations is consistent with research by Gilchrist et al. (2014) and Passeport et al. (2009) in which quantitative $\text{NH}_4^+\text{-N}$ reductions were improved in non IWS systems.

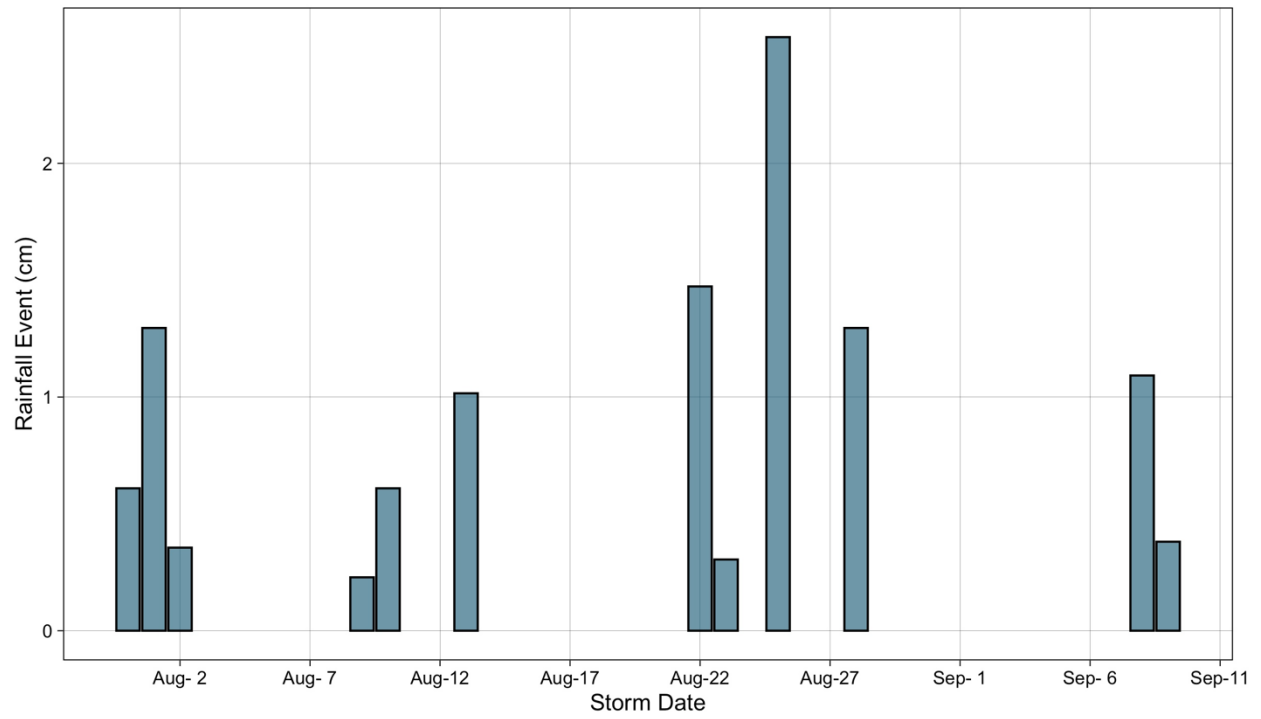
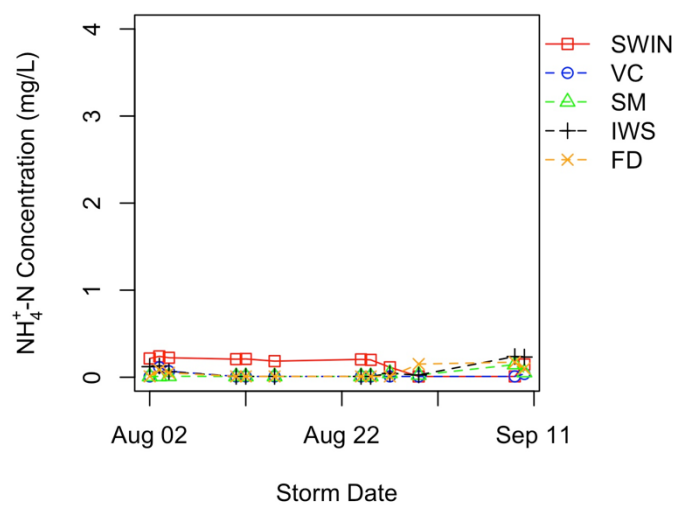
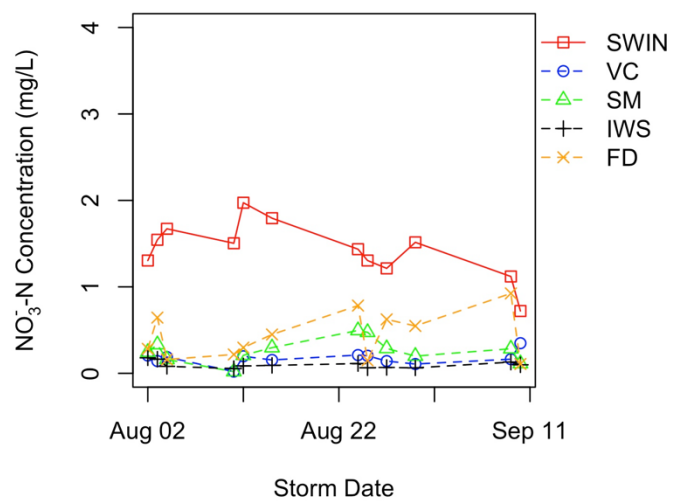


Figure 3.1 Bar plot of storm size (cm) for the 6-week study period

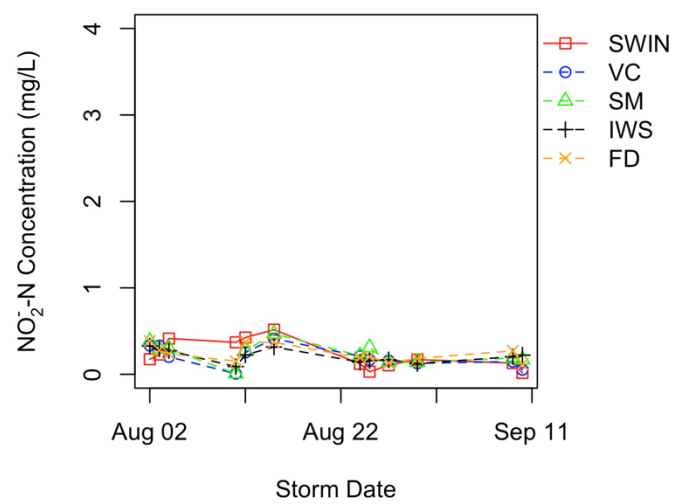


(a)

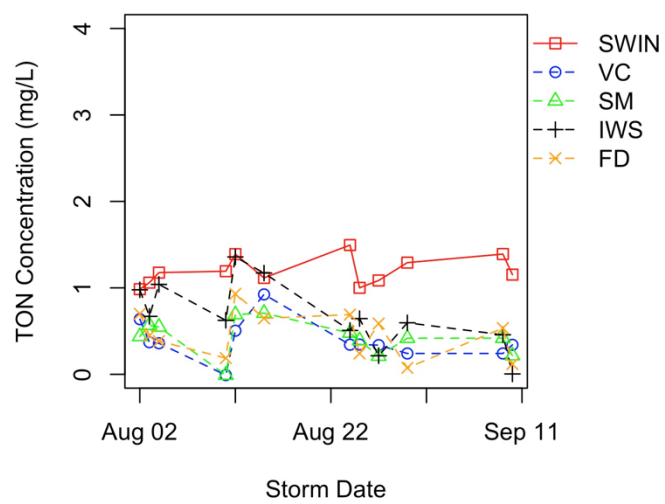


(b)

Figure 3.2 Nitrogen species concentrations for the influent and effluent of each column type.

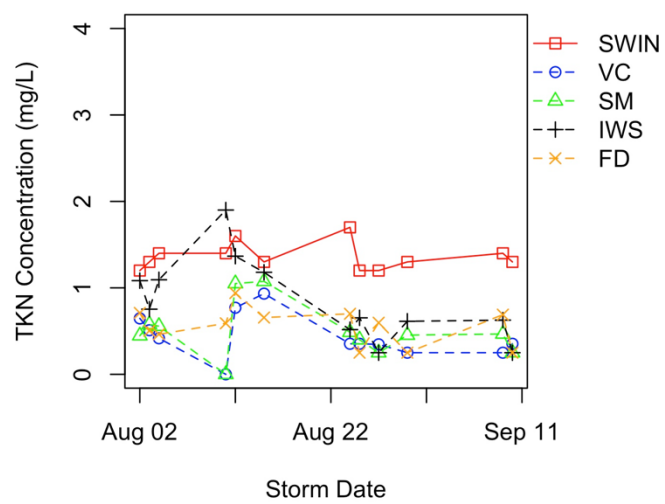


(c)

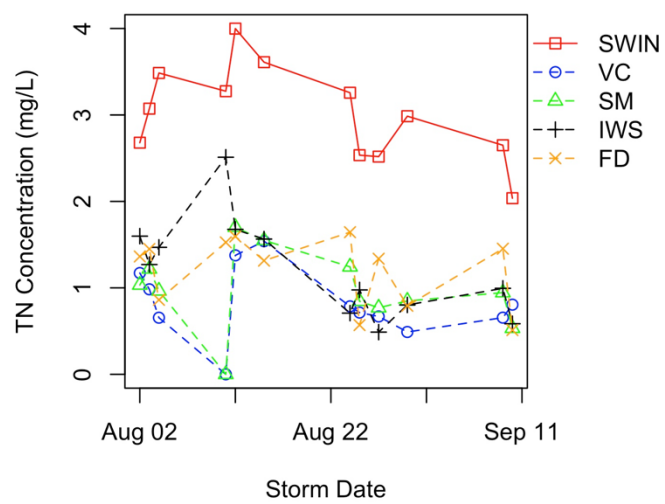


(d)

Figure 3.2 Continued



(e)



(f)

Figure 3.2 Continued

Table 3.2 Percent reductions, median effluent concentrations, mean effluent concentrations, standard deviation (Std. Dev.), and relative standard deviation (RSD) for each column type throughout the 6-week study

Pollutant	Type	Reduction (%)	Median (mg/L)	Mean (mg/L)	Std. Dev.	RSD
NH ₄ ⁺ -N	VC ^{*aΔ}	84.7	0.009	0.025	0.045	181.2
	SM ^{aΔ}	82.5	0.009	0.029	0.059	205.4
	IWS ^{bΔ}	54.4	0.009	0.075	0.097	130.4
	FD ^Δ	68.3	0.009	0.052	0.080	153.8
NO ₃ ⁻ -N	VC ^{aΔ}	87.8	0.150	0.174	0.132	76.0
	SM ^{aΔ}	81.8	0.219	0.260	0.228	87.8
	IWS ^{bΔ}	93.0	0.085	0.099	0.052	52.7
	FD ^{cΔ}	69.7	0.378	0.432	0.351	81.3
NO ₂ ⁻ -N	VC ^a	11.1	0.192	0.200	0.130	65.1
	SM	-6.8	0.198	0.240	0.151	62.7
	IWS	5.9	0.217	0.212	0.108	50.9
	FD ^b	-7.1	0.203	0.241	0.131	54.4
TKN	VC ^{aΔ}	66.0	0.250	0.462	0.265	57.3
	SM ^{aΔ}	63.2	0.500	0.513	0.354	69.1
	IWS ^{bΔ}	41.6	0.790	0.796	0.433	54.5
	FD ^{aΔ}	59.7	0.590	0.548	0.273	49.9
TON	VC ^{aΔ}	67.7	0.241	0.386	0.293	75.7
	SM ^{aΔ}	64.6	0.241	0.423	0.376	88.9
	IWS ^{bΔ}	42.3	0.685	0.689	0.485	70.4
	FD ^{aΔ}	61.3	0.501	0.462	0.316	68.4
TN	VC ^{aΔ}	68.2	0.250	0.462	0.265	57.3
	SM ^Δ	63.1	0.500	0.513	0.354	69.1
	IWS ^{bΔ}	36.9	0.790	0.796	0.433	54.5
	FD ^{bΔ}	59.5	0.590	0.548	0.273	49.9

* Letters indicates significant differences between treatments ($\alpha = 0.05$) as per Wilcoxon Sign-Rank test. No letter indicates no significant difference for that configuration.

^Δ Symbol indicates significant difference between Influent and effluent concentrations for each pollutant.

The IWS configuration had the least removal presumably because of the lack of substantial aerobic space within the columns. The FD columns were expected to perform with the greatest removal; however, this configuration showed less removal than the RTC types. This shows that the creation of more dynamic storage and release systems can promote better nutrient removal. For example, the VC configuration was intended to act similarly to IWS configuration; however, it was able to better reduce NH_4^+ -N concentrations. Facilitating a more dynamic storage area, meaning allowing wet and dry conditions to fluctuate more within the system itself, the VC column type could have triggered more microbial activity through changes in inundation. The movement of carbon throughout the system would have also triggered more microbial processing compared to the initial column study (Willard et al. 2017).

Analysis of NO_3^- -N (Figure 3.2b) data showed similar trends to those of Persaud et al. (2019). The IWS configuration exhibited the greatest NO_3^- -N removal (93%) while the FD configuration exhibited the least (69.7%). Similar to that discussed above, this is due to the differences in design as the IWS configuration creates a more defined anaerobic zone within the soil profile and is consistent with previous research comparing FD and IWS bioretention systems (Li et al. 2014; Payne et al. 2014; Manka et al. 2016; Palmer et al. 2013). The FD configuration has no water storage, and this prevents denitrification from occurring in areas other than soil microsites. VC and SM columns performed similarly to each other, which is a result of the internally fluctuating wetting and drying regimes of both designs. As noted by Persaud et al. (2019) active control systems appear to be able to balance the more extreme conditions of the FD and IWS, more effectively providing both anaerobic and aerobic conditions.

As noted above, the RTC configurations balance NH_4^+ -N and NO_3^- -N reduction, suggesting they can balance aerobic and anaerobic conditions. These columns were also allowed

to mature, which resulted in improved removal that was also more balanced between nitrification and denitrification. That is, $\text{NH}_4^+\text{-N}$ and $\text{NO}_3^-\text{-N}$ removal occurs in similar percentages. More similarity between $\text{NH}_4^+\text{-N}$ and $\text{NO}_3^-\text{-N}$ can be seen in FD treatments, however, performance is around 20% less than that of RTC treatments. The IWS treatment showed the most imbalance as $\text{NO}_3^-\text{-N}$ removal was significantly greater than $\text{NH}_4^+\text{-N}$ removal.

Comparing $\text{NO}_2^-\text{-N}$ (Figure 3.2c) with previous column study results by (Persaud et al. 2019) show a significant decrease in export and even some removal from VC and IWS configurations. VC and SM configurations exhibited 11.1% and -6.8% removal respectively, while IWS and FD configurations had 5.9% and -7.1% removal respectively. Less export of $\text{NO}_2^-\text{-N}$ could indicate more complete denitrification taking place within the systems. The greatest improvement can be noted for the IWS configurations which, because they are not subjected to real-time control, indicates improved performance as a result of increased microbial processing within the system. The overall increase in performance for all treatments and all constituents can be attributed to carbon migration and the maturation of the columns over two years.

Analysis of TON (Figure 3.2d) in this study shows organic nitrogen removal from all configurations. The IWS columns showed the smallest TON reduction (42%) while the VC, SM, and FD columns exhibited similar removal rates (67.7%, 64.6%, and 61.3%, respectively). All configurations exhibited overall nitrogen reductions with TN reductions were greatest in the RTC configurations (68.2% and 63.1% for VC and SM respectively) while reductions for the traditional FD and IWS systems were 59.5 and 36.9% respectively (Figure 3.2f). The TN removal rates in this study are consistent with removal rates seen in enrichment research by Goh et al. (2017) which reported 60-62% removal, and Barrett et al. (2013) which reported 59-79% removal. This trend

also follows that of $\text{NH}_4^+\text{-N}$ and $\text{NO}_3^-\text{-N}$ where RTC designs performed better than traditional designs, which is expected as most of the TN in stormwater is made up of nitrate.

Antecedent Conditions

Analysis conducted on water quality data involved correlations using Spearman's rho. Antecedent wet and dry conditions, soil moisture conditions, and event size are tested against removal for each constituent. Positive correlations (Spearman's rho greater than 0.50) between effluent concentrations, antecedent dry conditions were only exhibited in the FD configuration. Of the six pollutants of interest, $\text{NO}_3^-\text{-N}$ was correlated to dry periods ($\text{rho} = 0.62$). In relation to storm size, $\text{NO}_3^-\text{-N}$ was correlated in SM and FD configurations ($\text{rho} = 0.52$ and $\text{rho} = 0.84$ respectively). These positive correlations are unexpected as $\text{NO}_3^-\text{-N}$ removal is primarily thought to occur in anaerobic environments. The SM configuration does perform water storage functions however the FD configuration is constantly draining and so the high removal correlations would have to be due to processing in soil microsites.

Analysis of soil moisture conditions show more positive correlations for the 30 cm soil moisture level and TON, TKN, and TN removal in the IWS configuration ($\text{rho} = 0.87$, $\text{rho} = 0.88$, and $\text{rho} = 0.88$ respectively). The overall sustained nature of the saturated zone would result in more removal from the system when compared to other configuration types. While the positive correlation is related to the upper soil moisture sensor and not the lower sensor, it is important to note that average soil moisture in the upper portion of the IWS configuration is greater than that of upper soil moisture in other configurations.

16S Analysis

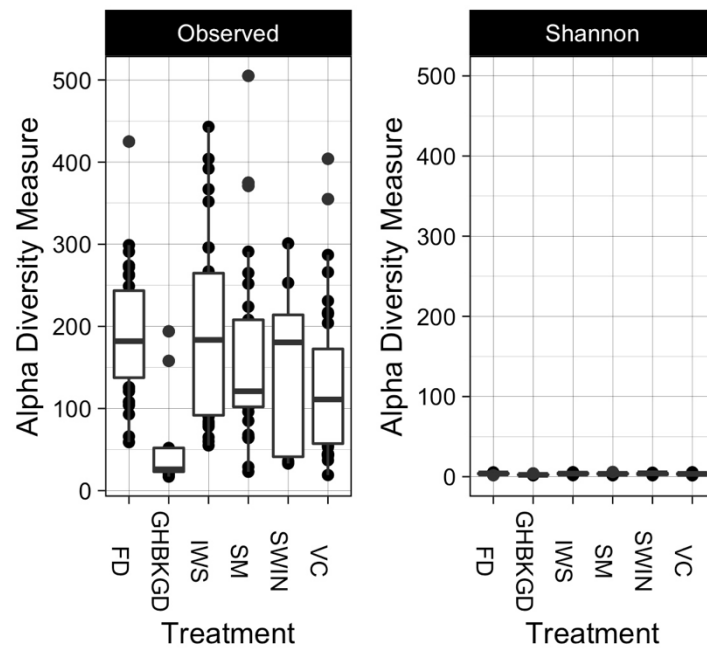
Overall analysis of microbial trends in this research included analysis of alpha and beta diversity trends. Filtering and genera selection resulted in 7 selected genera for analysis in regard to nitrification and denitrification (Table 3.3). Analysis conducted on the entire data set

Alpha Diversity Analysis

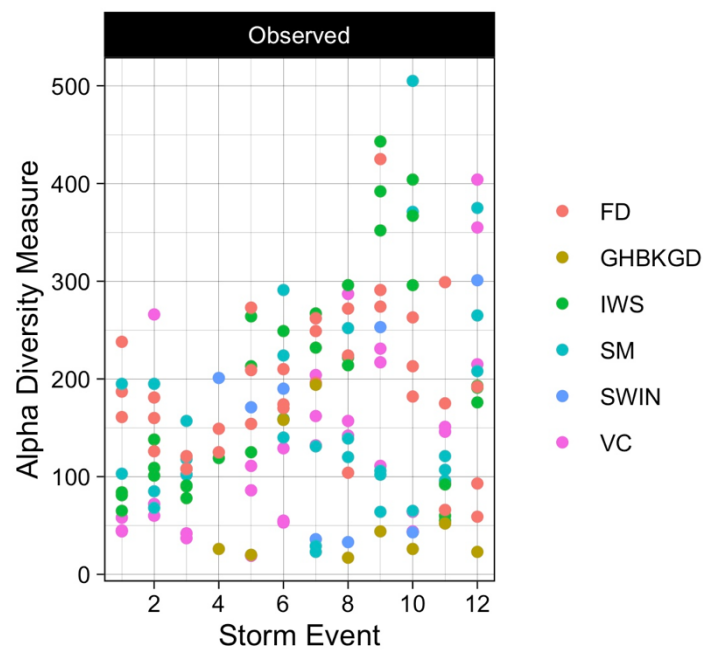
Figure 3.3a shows differences in diversity measure based on the observed richness (number of species in a sample) and the Shannon index. As expected, the GHBKGD sample type shows the least amount of diversity as these samples were taken from a supply of autoclaved deionized water left in the greenhouse. The SWIN, traditional, and RTC treatments all exhibited similar Shannon index scores which followed a similar trend when considering observed richness. This similarity between treatments indicates similar species and abundance counts. In Figure 3.3b, observed richness is plotted for each storm. While there is no definitive trend or clustering based on column type in the graph for observed richness, the diversity measure generally has an increasing trend as the study progresses, in particular between storm 1 and 10. This indicates a growing variance in abundance as the study progresses. Most notable are the diversity measures observed for storms 9 and 10, where SM and IWS columns exhibited greater diversity than other column types. It can be seen that IWS, and FD columns show a steady increase in abundance diversity measure while RTC columns seem to have more variable results. Columns were treated similarly before the beginning of the study with all being operated as if they were free-draining systems. As the hydrologic changes of each configuration were established, differences in species richness could also be allowed to develop. At the same time, there was a notable drop in diversity from storm 10 to 11.

Table 3.3 List of Nitrifying and denitrifying genus selected for analysis

Number of Taxa	Genus	Process
2	<i>Nitrosomonas</i>	Nitrification
1	<i>Nitrospira</i>	Nitrification
10	<i>Nitrospira</i>	Nitrification
7	<i>Bacillus</i>	Denitrification
2	<i>Paracoccus</i>	Denitrification
7	<i>Pseudomonas</i>	Denitrification
1	<i>Thiobacillus</i>	Denitrification



(a)



(b)

Figure 3.3 Alpha a) Box plots of Observed and Shannon Diversity b) Plot of Alpha Diversity per storm event

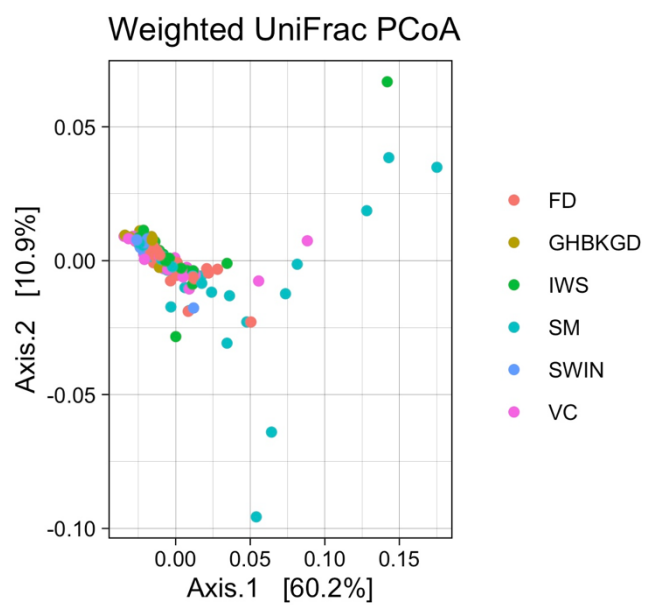
This could have been a result of dry periods impacting microbes as there was a 9- day dry period between storm event 10 and 11. While this was not seen in previous dry periods (like that which occurred between storms 6 and 7, it is also important to note that storm size would also affect microbe diversity and potential washout may have occurred as the storm events preceding the 9-day dry period between events 10 and 11 where considerably large (2.54 and 1.30 cm respectively). This would have necessitated a greater recovery by the systems.

Beta Diversity Analysis

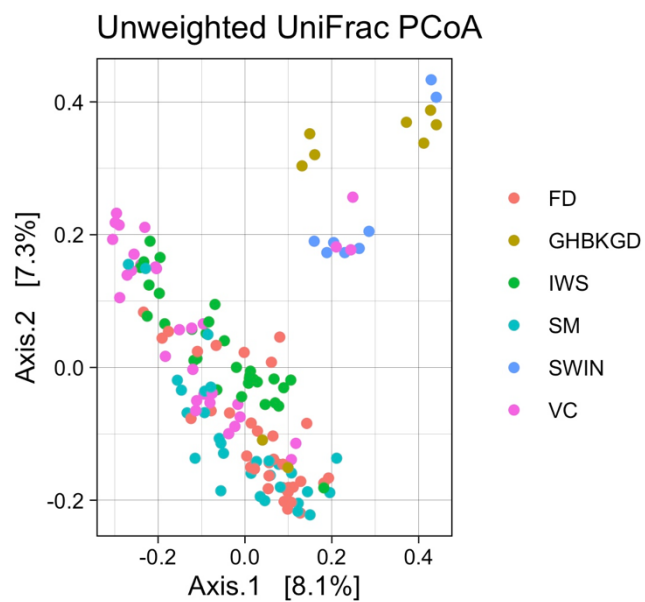
To further analyze similarities between treatments, a PCoA plot was generated (Figure 3.4a). For the majority of the treatments, there is no clear separation suggesting that all samples are similar to each other. The most dissimilarity is seen for SM samples as they are not clustered with the other data points and are spread further along axis 1. In the unweighted unifracs PCoA plot in Figure 3.4b, more separation between data points is observed, but all bioretention treatments are still grouped in the main cluster of points. Conversely, clustering of the GHBKGD and SWIN types can be seen away from samples from Bioretention columns.

Nitrifying and Denitrifying Genus

Comparisons between each configuration are also made based on abundance (Figure 3.5a and Figure 3.5b), with the selected genera from Table 3.3. As noted above, from the larger list of 24 genera, 7 were present in this data set. For more specific analysis the *Nitrospira* and *Pseudomonas* genera were chosen for discussion because of their overwhelming presence in the processed abundance data (Table 3.4).

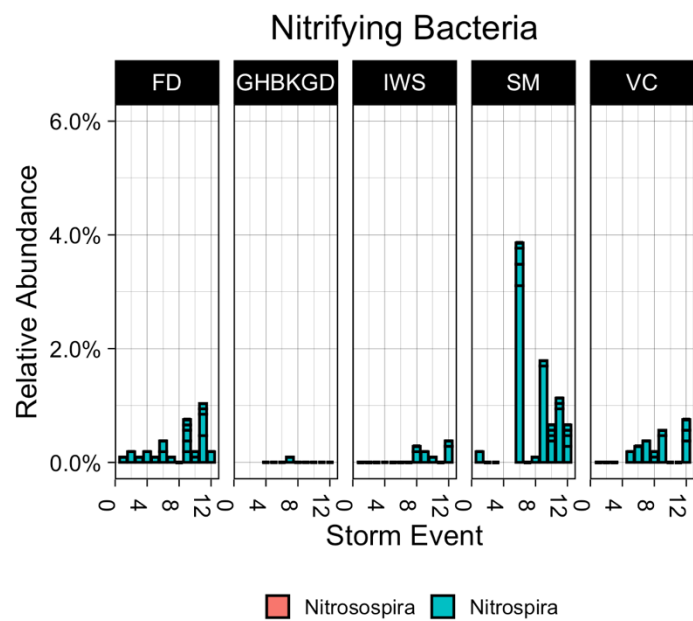


(a)

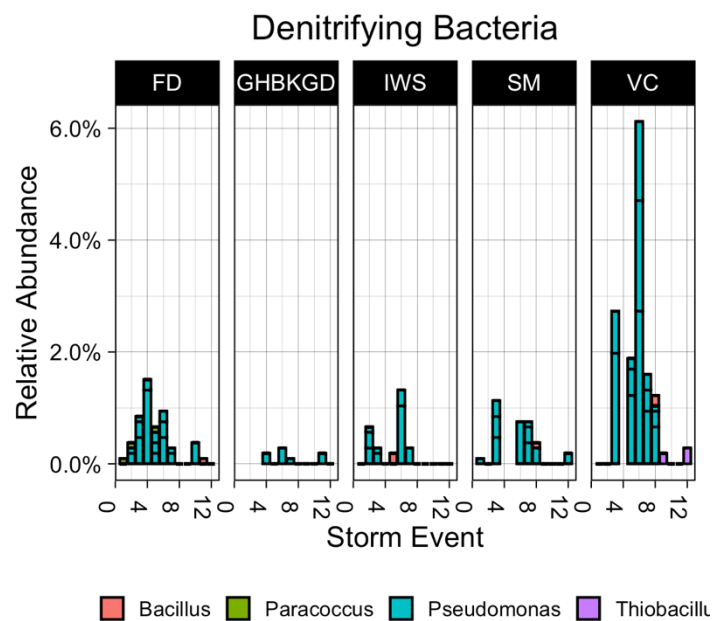


(b)

Figure 3.4 Weighted and unweighted unifrac PCoA plots for all taxa



(a)



(b)

Figure 3.5 Bar plots for relative abundance of nitrifying and denitrifying genus

Table 3.4 Average relative abundance (%) of *Nitrospira* and *Pseudomonas* over the 6-week study period

<i>Genus</i>	VC	SM	IWS	FD
<i>Nitrospira</i>	0.72	3.07	0.31	1.00
<i>Nitrosospira</i>	0.00	0.00	0.00	0.00
<i>Pseudomonas</i>	4.38	1.17	0.84	1.46
<i>Paracoccus</i>	0.00	0.00	0.00	0.09
<i>Bacillus</i>	0.06	0.03	0.06	0.03
<i>Thiobacillus</i>	0.16	0.00	0.00	0.00

Nitrification

Nitrification is a two-step process in which ammonia oxidizing bacteria (AOB) convert ammonia into nitrite and nitrite oxidizing bacteria (NOB) convert nitrite into nitrate (Xia et al. 2011). Typically found in low nitrogen soil environments (Attard et al. 2010), *Nitrospira* is a NOB (Daims et al. 2015; Zhou et al. 2016; Pajares and Bohannan 2016; Chen et al. 2013). The bar plot for nitrifying genera is primarily dominated by the *Nitrospira* genus. Dominance of the *Nitrospira* genus within bioretention was also seen in research by Morse et al. (2018) and was an expected outcome in this research because of the nutrient poor nature of bioretention media. Collectively, the SM and FD configurations show the greatest *Nitrospira* abundance followed by VC and finally the IWS configuration shows the least.

This trend in abundance does not follow the removal trends noted for $\text{NH}_4^+\text{-N}$, which indicates differences in functional gene activity within each sample. Regardless, differences in the environmental conditions created by each configuration do appear to influence the magnitude of *Nitrospira* abundance. The FD and SM configurations have the most aerobic space within the soil profile, while the IWS and VC columns have designated anaerobic space. It is likely that the aerobic space in FD and SM allowed greater abundance of the nitrifying genus. Increased abundance in the SM configuration compared to abundance seen in the FD configuration would also indicate a more favorable environment was created through employing a more dynamic treatment type. The selected soil moisture control trigger for the SM systems would have promote field capacity in the upper portion of the SM soil profile, which is ideal for microbial productivity (Barros et al. 1995).

Denitrification

Results from the analysis of denitrifying genera showed a preponderance of the genus *Pseudomonas*, a bacterial genus common in denitrification (Chen et al. 2013; Zumft 1997; Payne et al. 2014). It is common for *Pseudomonas* to be associated with plant health and be associated with soil and root interactions (Gamalero et al. 2020) which would make their presence associated with the upper portions of the soil profile. In relation to denitrification this genus would either facilitate the process in anaerobic pores in the upper portion of the soil profile or have to be found in the lower portion of the soil profile. Because denitrification is typically promoted in saturated zones that create anaerobic conditions which are typically in the lower portions of bioretention practices (Li et al. 2014; Collins et al. 2010), the role of carbon migration and availability could facilitate *Pseudomonas* growth in the lower portion of the soil profile. The maturation of these columns would have allowed carbon availability in the saturated zone which has been discussed as a limiting factor in denitrification in IWS practices as it fuels bacterial processing (Igielski et al. 2019; Thompson et al. 2008). In this research, the VC configuration has the most significant *Pseudomonas* abundance when compared to other column types. This was unexpected as the IWS configuration promoted the most NO_3^- -N removal.

Based on water quality data, it would have also been expected that the SM treatment would show similar abundances of the nitrifying genus, however, it should be noted that although bacteria may be present, this is not indicative of their functional potential. The environments created by each system should promote the growth of various types of bacteria which is consistent with the presence of the *Pseudomonas* genus in the VC configuration. This difference between IWS and VC configurations further highlights the benefit of promoting microbial growth with a more dynamic environment.

RTC as Dynamic Control Methods

The SM and VC RTC configurations showed the greatest abundance of nitrifying and denitrifying genera, respectively. These columns also show the greatest differences in abundance when considering nitrifying and denitrifying genera which further indicates that dynamic systems create more specific environments for bacterial growth. The water quality data in this study do not follow this trend as removal rates of $\text{NH}_4^+\text{-N}$ and $\text{NO}_3^-\text{-N}$ for VC and SM configurations are more similar than those of the traditional columns. These removal results would have been supported by more even abundances of *Nitrospira* and *Pseudomonas* in the RTC systems, however, this result was seen more in the traditional configurations. The dynamic nature of the RTC systems allowed a more staggered flow through the system which subsequently would have allowed more oxygen and carbon to be moved into lower parts of the soil profile. The periodic oxygen and carbon inputs may have fostered more growth while the prolonged retention in the RTC systems would have allowed anaerobic conditions to form in VC columns and denitrification to occur. Establishing a control scheme that functions based on conditions in the saturated zone of a Bioretention column would have allowed the abundance of *Pseudomonas* to grow more in this system when compared to other configurations. Creation of anoxic environments would have allowed *Pseudomonas* to undergo denitrification which it would have typically done in microsites in soil pores (Chen et al. 2019). The SM systems had less storage and control was based on soil moisture conditions in the upper portions of the soil profile. Targeting the upper portion of the soil profile in establishing a control scheme may have promoted the growth of *Nitrospira* while the storage that the system facilitated allowed denitrification to occur in the lower portion of the soil profile. These oxygen-carbon dynamics are more evident in the dynamic, RTC systems than the more stagnant, traditional systems.

The FD and IWS configurations were intended to show extremes regarding the facilitation of nitrification and denitrification and the nitrifying and denitrifying genera. The FD column types would have been expected to show more *Nitrospira*, or other nitrifying genera, than any other column type because of the volume of space within the practice allotted for nitrifying conditions. The water quality results, coupled with the abundance data show a balance between removal rates of $\text{NH}_4^+\text{-N}$ and $\text{NO}_3^-\text{-N}$, which is a similar trend to what is seen in the water quality of the RTC columns but not in the measured abundance data. When looking at data for the IWS configuration, a greater abundance of *Pseudomonas* would have been expected as this configuration exhibited the greatest $\text{NO}_3^-\text{-N}$ removal, however, a similar trend in low abundance is noted for this configuration as well. Removal exhibited in these systems, may not have been influenced by microbial contributions as much as in the RTC systems and it is important to note that other physical mechanisms for removal would have also contributed to the water quality results in this research. The low abundance exhibited by these traditional columns are not indicative of function, however, they do contribute to the growing knowledge of environments tailored for microbial development and potential.

Comparisons and Correlation Analysis

Correlation analysis was performed between results for nutrient reductions, abundance data, wet and dry periods, and average soil moisture conditions for each storm. *Pseudomonas* showed no correlations with any other variables, however correlations between *Nitrospira* abundance and nutrient reductions shows negative correlations with $\text{NO}_3^-\text{-N}$ in FD and VC configurations ($\rho = -0.76$ and -0.7 respectively) and $\text{NO}_2^-\text{-N}$ in IWS configurations ($\rho = -0.79$). In regard to nutrient correlations, TN and TKN were positively correlated in all column types (ρ

= 0.84, 0.77, 0.92, and 0.76 for VC, SM, IWS, and FD configurations respectively). When looking at soil moisture conditions, IWS configurations were the only ones to show correlations which occurred at the 30 cm soil moisture depth (Spearman's ρ = -0.81 and 0.83 for TKN and TON respectively). These correlations signify environmental conditions affecting nutrient reductions, however similar results for $\text{NH}_4^+\text{-N}$ were not seen and would have been expected as TON is calculated by subtracting $\text{NH}_4^+\text{-N}$ from TKN removal rates. No strong correlation was drawn between dry and wet periods and any other variable.

Although correlations were drawn between *Nitrospira* and nutrient reductions, analysis of functional genes would have to be done to determine the full extent of relationships present bacteria and performance. Recent literature by Deng et al. (2020) has shown that community composition influences nitrate removal and further analysis into community influence on nitrification would need to occur. Additional correlation analysis between abundance and gene expression would also provide more definite links between nutrient removal and specific bacteria.

Conclusion

A 6-week column study was conducted in which RTC and traditional bioretention columns were compared for differences in nitrogen treatment and microbial populations. For all pollutants studied in this research, RTC configurations were able to create more balanced removal than the IWS and FD configurations which performed with lower $\text{NH}_4^+\text{-N}$ (54% and 68% respectively) removal and as extremes in $\text{NO}_3^-\text{-N}$ removal (93% and 70% respectively). No significant difference in nutrient removal was noted between the SM and VC configurations. All columns also showed significant increases in removal when compared to the original 2018 column study, which is a result of column maturation.

The most notable trend in microbial populations are the increased abundances of *Nitrospira* in SM columns and *Pseudomonas* in VC columns (relative to the other columns). These genera represent bacteria that are nitrifying and denitrifying, respectively. All traditionally designed columns had comparably low abundances. This spike in abundance indicates the potential for increased function in more dynamic systems. Difference between SM and VC are indicative of differences in their respective control scheme as well. Nitrifying bacteria were more abundant in the SM configuration which was controlled by soil moisture conditions in the upper portions of the soil profile, while denitrifying bacteria is more abundant in the VC configuration which was controlled by conditions in the lower portion of the profile. Controlling based on a location within the soil profile can therefore lead to environmental conditions that are conducive to greater abundances of either nitrifying or denitrifying bacteria. When comparing this abundance data to nutrient removal, these trends are not followed and instead the VC treatment performs better than SM in both NO_3^- -N and NH_4^+ -N reduction. The VC treatment is thought to facilitate better aerobic and anaerobic environments which would have promoted more nitrification and denitrification when compared to SM which did not have as designated anaerobic zone created. Studying the stratification of RTC systems would give more insight into the microbial populations actively participating in nutrient processing.

Overall, performance between the two RTC configurations was comparable, and this research highlights the potential of more dynamic Bioretention systems in promoting the abundance of nitrifying or denitrifying bacteria. When utilizing RTC systems, targeting function in an aerobic or anaerobic environment may also influence the abundance of particular genera over others. That is, the selection of environmental conditions or area selection for sensor triggered control can influence the growth of microbial communities and their nutrient processing

capabilities. Dynamic systems are able to constantly adjust to ideal performance goals and deliver increased removal performance as a practice ages, while more static systems are limited to set metrics regardless of extenuating environmental circumstance. Further research will include metagenomic testing to determine the functional genes present and allow more in-depth analysis of the nitrogen processing pathways being facilitated in each system.

Acknowledgement

This work was funded by the National Science Foundation (NSF Grant number 1737432). The authors would like to thank University of Tennessee Civil Engineering workshop, East Tennessee AgResearch and Education Center, and the Hazen research group in facilitating data processing for this research.

References

- Attard, E., F. Poly, C. Commeaux, F. Laurent, A. Terada, B. F. Smets, S. Recous, and X. Le Roux. 2010. 'Shifts between Nitrospira- and Nitrobacter-like nitrite oxidizers underlie the response of soil potential nitrite oxidation to changes in tillage practices', *Environmental Microbiology*, 12: 315-26.
- Barrett, M. E., M. Limouzin, and D. F. Lawler. 2013. 'Effects of Media and Plant Selection on Biofiltration Performance', *Journal of Environmental Engineering-Asce*, 139: 462-70.
- Barros, N., I. Gomezorellana, S. Feijoo, and R. Balsa. 1995. 'THE EFFECT OF SOIL-MOISTURE ON SOIL MICROBIAL ACTIVITY STUDIED BY MICROCALORIMETRY', *Thermochimica Acta*, 249: 161-68.
- Bartos, M., B. Wong, and B. Kerkez. 2018. 'Open storm: a complete framework for sensing and control of urban watersheds', *Environmental Science-Water Research & Technology*, 4: 346-58.
- Bosch, D. J., M. B. Wagena, A. C. Ross, A. S. Collick, and Z. M. Easton. 2018. 'Meeting Water Quality Goals under Climate Change in Chesapeake Bay Watershed, USA', *Journal of the American Water Resources Association*, 54: 1239-57.
- Bratieres, K., T. D. Fletcher, A. Deletic, and Y. Zinger. 2008. 'Nutrient and sediment removal by stormwater biofilters: A large-scale design optimisation study', *Water Research*, 42: 3930-40.
- Cameron, K. C., H. J. Di, and J. L. Moir. 2013. 'Nitrogen losses from the soil/plant system: a review', *Annals of Applied Biology*, 162: 145-73.
- Chen, T., Y. P. Liu, B. Zhang, and L. H. Sun. 2019. 'Plant rhizosphere, soil microenvironment, and functional genes in the nitrogen removal process of bioretention', *Environmental Science-Processes & Impacts*, 21: 2070-79.

- Chen, X. L., E. Peltier, B. S. M. Sturm, and C. B. Young. 2013. 'Nitrogen removal and nitrifying and denitrifying bacteria quantification in a stormwater bioretention system', *Water Research*, 47: 1691-700.
- Collins, K. A., T. J. Lawrence, E. K. Stander, R. J. Jontos, S. S. Kaushal, T. A. Newcomer, N. B. Grimm, and M. L. C. Ekberg. 2010. 'Opportunities and challenges for managing nitrogen in urban stormwater: A review and synthesis', *Ecological Engineering*, 36: 1507-19.
- Dagenais, D., J. Brisson, and T. D. Fletcher. 2018. 'The role of plants in bioretention systems; does the science underpin current guidance?', *Ecological Engineering*, 120: 532-45.
- Daims, H., E. V. Lebedeva, P. Pjevac, P. Han, C. Herbold, M. Albertsen, N. Jehmlich, M. Palatinszky, J. Vierheilig, A. Bulaev, R. H. Kirkegaard, M. von Bergen, T. Rattei, B. Bendinger, P. H. Nielsen, and M. Wagner. 2015. 'Complete nitrification by *Nitrospira* bacteria', *Nature*, 528: 504-+.
- Deng, Qinghui, Lingling Wan, Xiaowen Li, Xiuyun Cao, Yiyong Zhou, and Chunlei Song. 2020. 'Metagenomic evidence reveals denitrifying community diversity rather than abundance drives nitrate removal in stormwater biofilters amended with different organic and inorganic electron donors', *Chemosphere*, 257: 127269.
- Dietz, M. E., and J. C. Clausen. 2005. 'A field evaluation of rain garden flow and pollutant treatment', *Water Air and Soil Pollution*, 167: 123-38.
- Fletcher, T. D., W. Shuster, W. F. Hunt, R. Ashley, D. Butler, S. Arthur, S. Trowsdale, S. Barraud, A. Semadeni-Davies, J. L. Bertrand-Krajewski, P. S. Mikkelsen, G. Rivard, M. Uhl, D. Dagenais, and M. Viklander. 2015. 'SUDS, LID, BMPs, WSUD and more - The evolution and application of terminology surrounding urban drainage', *Urban Water Journal*, 12: 525-42.
- Gamalero, E., E. Bona, V. Todeschini, and G. Lingua. 2020. 'Saline and Arid Soils: Impact on Bacteria, Plants, and Their Interaction', *Biology-Basel*, 9: 26.

- Gilchrist, S., M. Borst, and E. K. Stander. 2014. 'Factorial Study of Rain Garden Design for Nitrogen Removal', *Journal of Irrigation and Drainage Engineering*, 140: 11.
- Glaister, B. J., T. D. Fletcher, P. L. M. Cook, and B. E. Hatt. 2017. 'Interactions between design, plant growth and the treatment performance of stormwater biofilters', *Ecological Engineering*, 105: 21-31.
- Goh, H. W., N. A. Zakaria, T. L. Lau, K. Y. Foo, C. K. Chang, and C. S. Leow. 2017. 'Mesocosm study of enhanced bioretention media in treating nutrient rich stormwater for mixed development area', *Urban Water Journal*, 14: 134-42.
- Igielski, S., B. V. Kjellerup, and A. P. Davis. 2019. 'Understanding urban stormwater denitrification in bioretention internal water storage zones', *Water Environment Research*, 91: 32-44.
- Jay, J. G., M. Tyler-Plog, S. L. Brown, and F. Grothkopp. 2019. 'Nutrient, Metal, and Organics Removal from Stormwater Using a Range of Bioretention Soil Mixtures', *Journal of Environmental Quality*, 48: 493-501.
- Kerkez, B., C. Gruden, M. Lewis, L. Montestruque, M. Quigley, B. Wong, A. Bedig, R. Kertesz, T. Braun, O. Cadwalader, A. Poresky, and C. Pak. 2016. 'Smarter Stormwater Systems', *Environmental Science & Technology*, 50: 7267-73.
- Konrad, Christopher P. 2003. "Effects of Urban Development on Floods." In.: USGS.
- Laurenson, G., S. Laurenson, N. Bolan, S. Beecham, and I. Clark. 2013. 'The Role of Bioretention Systems in the Treatment of Stormwater.' in D. L. Sparks (ed.), *Advances in Agronomy, Vol 120* (Elsevier Academic Press Inc: San Diego).
- Li, C. L., M. Liu, Y. M. Hu, M. Zong, M. H. Zhao, and M. T. Walter. 2019. 'Characteristics of impervious surface and its effect on direct runoff: a case study in a rapidly urbanized area', *Water Science and Technology-Water Supply*, 19: 1885-91.

- Li, M. H., M. Swapp, M. H. Kim, K. H. Chu, and C. Y. Sung. 2014. 'Comparing Bioretention Designs With and Without an Internal Water Storage Layer for Treating Highway Runoff', *Water Environment Research*, 86: 387-97.
- Madigan, M., Bender, K., Buckley, D., Sattley, W., Stahl, D. . 2019. *Brock Biology of Microorganisms (15th Global Edition ed.)* (Pearson).
- Mangangka, I. R., A. Liu, P. Egodawatta, and A. Goonetilleke. 2015. 'Performance characterisation of a stormwater treatment bioretention basin', *Journal of Environmental Management*, 150: 173-78.
- Manka, B. N., J. M. Hathaway, R. A. Tirpak, Q. He, and W. F. Hunt. 2016. 'Driving forces of effluent nutrient variability in field scale bioretention', *Ecological Engineering*, 94: 622-28.
- Middleton, J. R., and M. E. Barrett. 2008. 'Water quality performance of a batch-type stormwater detention basin', *Water Environment Research*, 80: 172-78.
- Morse, N., E. Payne, R. Henry, B. Hatt, G. Chandrasena, J. Shapleigh, P. Cook, S. Coutts, J. Hathaway, M. T. Walter, and D. McCarthy. 2018. 'Plant-Microbe Interactions Drive Denitrification Rates, Dissolved Nitrogen Removal, and the Abundance of Denitrification Genes in Stormwater Control Measures', *Environmental Science & Technology*, 52: 9320-29.
- Mullapudi, A., M. Bartos, B. Wong, and B. Kerkez. 2018. 'Shaping Streamflow Using a Real-Time Stormwater Control Network', *Sensors*, 18: 11.
- Pajares, S., and B. J. M. Bohannan. 2016. 'Ecology of Nitrogen Fixing, Nitrifying, and Denitrifying Microorganisms in Tropical Forest Soils', *Frontiers in Microbiology*, 7: 20.
- Palmer, E. T., C. J. Poor, C. Hinman, and J. D. Stark. 2013. 'Nitrate and Phosphate Removal through Enhanced Bioretention Media: Mesocosm Study', *Water Environment Research*, 85: 823-32.

- Passeport, E., W. F. Hunt, D. E. Line, R. A. Smith, and R. A. Brown. 2009. 'Field Study of the Ability of Two Grassed Bioretention Cells to Reduce Storm-Water Runoff Pollution', *Journal of Irrigation and Drainage Engineering*, 135: 505-10.
- Payne, E. G. I., T. D. Fletcher, P. L. M. Cook, A. Deletic, and B. E. Hatt. 2014. 'Processes and Drivers of Nitrogen Removal in Stormwater Biofiltration', *Critical Reviews in Environmental Science and Technology*, 44: 796-846.
- Persaud, P. P., A. A. Akin, B. Kerkez, D. T. McCarthy, and J. M. Hathaway. 2019. 'Real time control schemes for improving water quality from bioretention cells', *Blue-Green Systems*, 1: 55-71.
- Roman, D., A. Braga, N. Shetty, and P. Culligan. 2017. 'Design and Modeling of an Adaptively Controlled Rainwater Harvesting System', *Water*, 9: 14.
- Shen, P. F., A. Deletic, K. Bratieres, and D. T. McCarthy. 2020. 'Real time control of biofilters delivers stormwater suitable for harvesting and reuse', *Water Research*, 169: 10.
- TDEC. 2014. "Tennessee Permanent Stormwater Management and Design Guidance Manual." In, 141-76.
- Thompson, A. M., A. C. Paul, and N. J. Balster. 2008. 'Physical and hydraulic properties of engineered soil media for bioretention basins', *Transactions of the Asabe*, 51: 499-514.
- TSM, Tennessee Stormwater Management. 2015. "Management and Design Guidance Manual." In *Tennessee Stormwater Managment*.
- Waller, L. J., G. K. Evanylo, L. A. H. Krometis, M. S. Strickland, T. Wynn-Thompson, and B. D. Badgley. 2018. 'Engineered and Environmental Controls of Microbial Denitrification in Established Bioretention Cells', *Environmental Science & Technology*, 52: 5358-66.

- Willard, L. L., T. Wynn-Thompson, L. H. Krometis, T. P. Neher, and B. D. Badgley. 2017. 'Does It Pay to be Mature? Evaluation of Bioretention Cell Performance Seven Years Postconstruction', *Journal of Environmental Engineering*, 143: 10.
- Xia, W. W., C. X. Zhang, X. W. Zeng, Y. Z. Feng, J. H. Weng, X. G. Lin, J. G. Zhu, Z. Q. Xiong, J. Xu, Z. C. Cai, and Z. J. Jia. 2011. 'Autotrophic growth of nitrifying community in an agricultural soil', *Isme Journal*, 5: 1226-36.
- Zhang, J. Y., R. P. Singh, Y. Z. Liu, and D. F. Fu. 2019. 'Design and operation of submerged layer in bioretention for enhanced nitrate removal', *Journal of Water Supply Research and Technology-Aqua*, 68: 744-56.
- Zhou, X., D. Fornara, M. Ikenaga, I. Akagi, R. F. Zhang, and Z. J. Jia. 2016. 'The Resilience of Microbial Community under Drying and Rewetting Cycles of Three Forest Soils', *Frontiers in Microbiology*, 7: 12.
- Zumft, W. G. 1997. 'Cell biology and molecular basis of denitrification', *Microbiology and Molecular Biology Reviews*, 61: 533-+.

Summary & Conclusion

This dissertation analyzes the implications of using real-time control (RTC) in bioretention systems. Column studies were employed to allow robust testing across a range of treatments. Two RTC configurations were tested against traditional bioretention designs for water quality, hydrology, and microbial functioning. The experimental methodology uses historic rainfall and predictions to dictate the operation of RTC systems and allow preemptive responses to impending events. Overall, RTC provides more dynamic operation which readily adapts to environmental changes and accomplishes targeted performance goals. This research focuses mainly on nitrogen processing as previous bioretention studies have reported variable results, and at times, nitrogen export from the systems.

Chapter 1 focuses on water quality performance with nutrient, metal, and suspended solid removal being analyzed. All column types were able to sufficiently remove metals and TSS however the SM treatment had the largest removal (86%). This column type also had the highest ammonium removal (43%) and the second highest nitrate removal (74%). When comparing SM to VC treatments, differences in nutrient removal were minimal. The FD and IWS columns performed at opposite extremes of the RTC columns with FD facilitating aerobic processing and IWS facilitating more anaerobic processing. As such, the IWS columns had the greatest NO_3^- -N removal while the FD had the greatest NH_4^+ -N removal. This was logical as the IWS columns had a greater soil moisture than all other column types for both the upper and lower sensor readings while the FD columns had the lowest. The VC and SM columns remained in a soil moisture range between the readings of IWS and FD. Overall, the two RTC columns balanced both NO_3^- -N and NH_4^+ -N removal more evenly creating a balance between treatments. This study highlights the

potential for RTC systems to balance aerobic and anaerobic processing to better manage influent nitrogen concentrations.

Chapter 2 analyzed the hydrologic implications of using RTC. A water balance of each system determined the amount of influent managed by each column type as well as partitioning to storage/evapotranspiration (ET), drainage, and infiltration. Over the course of the study, SM treatments performed similarly to IWS in regard to storage/ET (18% and 16% of the inflow partitioned to this pathway, respectively) while VC and FD treatments only exhibited 11% and 9% storage/ET. SM systems were also able to manage greater influent volumes as they only had 7% Bypass compared to 11% by VC columns. The FD columns produced the greatest amount of effluent (83%) while SM treatments performed similarly to IWS (63% and 62% Drainage). Overall, the SM treatment performed best for reducing Bypass and increasing storage/ET within a practice. These results coupled with the WQ performance seen in Chapter 1 show the benefit of using RTC to strike a balance between traditional treatments to satisfy performance goals. Finally, the drainage that resulted from predictions sent to RTC columns comprises only 1-2% of the water balance. Thus, removing the predictive element and having RTC operate in a more reactive manner would simplify future RTC design without compromising performance. Additional analysis in this chapter included an assessment of total nitrogen (TN) mass removal to determine if removal was a result of effluent volume reduction or biogeochemical processes. The VC and SM treatments performed similarly to IWS (82%, 77%, and 78% removal respectively) and better than the FD treatment (66% removal). This result did not follow volume reduction patterns as IWS had the greatest volume reduction, but VC had the greatest TN removal. From this, it can be said that mass removal of TN is influenced by more than just volume reduction and further investigation into removal of nitrogen species will more accurately specify channels of reduction.

Chapter 3 investigated both nitrogen species concentrations and microbes present in the effluent of each column type. Research in this chapter was conducted through a 6-week column study in which influent and effluent samples were taken in a similar manner to what was done in Chapter 1. Similar to results seen in Chapter 1, RTC systems were able to balance removal rates while IWS and FD systems performed at anaerobic and aerobic extremes, respectively. The IWS and FD columns displayed 93% and 70% removal for NO_3^- -N, and 54% and 68% removal for NH_4^+ -N, respectively. The VC and SM systems did not have any significant difference in removal and exhibited 82-88% removal for NO_3^- -N and 83-85 % removal for NH_4^+ -N, respectively. All columns also exhibited twice the removal rate seen in Chapter 1 which is theorized as being due to column maturation. Overall abundance in microbial species showed more *Nitrospira* in SM columns and *Pseudomonas* in VC columns. With *Nitrospira* being indicative of nitrification and *Pseudomonas* indicating denitrification, and as evidenced by the water quality data, the SM columns promoted more nitrification while the VC columns promoted more denitrification. This finding is a result of the control schemes used and how each targets a different environmental variable. SM columns targeted soil moisture conditions in the upper portions of the media which would facilitate nitrification as there is more oxygen in that layer. The VC columns were more focused on the volume of water held in the lower portions of the column and thus promoted anaerobic environments for denitrification to occur. The presence of these species in IWS and FD columns is considerably lower which was unexpected. This ultimately leads to the conclusion that the more dynamic nature of the RTC treatments promotes better microbial processing. Furthermore, specific processing may be targeted based on the location of environmental sensors within a system.

Collectively, all three chapters contribute to the understanding that RTC has greater potential for optimizing treatment and allowing adaptability over time. Further, this targeting of specific processing can be accomplished while simultaneously maintaining or improving other performance parameters (such as hydrologic goals). The potential for RTC to be used in other green infrastructure practices should also be researched, as these systems allow a more direct method of setting and controlling performance in green infrastructure. Specifically, in bioretention, controlling systems based on dissolved oxygen or other water quality measurements could provide more targeted control and performance. In other practices, testing other soil types or designing control based on surface water level would be useful, particularly in ponding-based systems. Using turbidity readings would also be another method of new control. Using environmental factors in designing new control schemes is practice specific, however, there are a variety of ways RTC can be altered for specific use.

Limitations to this work should be considered and addressed in future research. The insights herein are based solely on lab-scale work with hydrologic performance based on historic rainfall amounts, but not rainfall intensity. Rapid inundation of a system could result in more sudden flow patterns when compared to a more gradual application approach. That is, application in a shorter time (1-2 hours) frame, rather than a rain event occurring over the course of an entire day could influence a designs overall hydrologic performance. The predictive nature of application should also be studied further as shorter or longer predictive times may cause systems to react to incoming rainfall differently; however, this must be balanced by potential increased uncertainty in precipitation predictions with longer forecasts. Microbial assessments only included the abundance of species within the effluent of each column type; however, the functional genes present within the columns should also be studied to determine more specific portions of the

nitrogen cycle being targeted in various areas of the soil profile. Furthermore, this research was conducted in lab-scale studies and so field-scale and watershed-scale analysis should also be completed to measure the efficacy of RTC and the larger implications that using this technology can have. This is particularly important in a hydrologic context as flows through stormwater networks and differences in infrastructure can also change performance expectations of a bioretention practice and the watershed as a whole.

Overall, the research completed in this dissertation highlights the benefits of using RTC in creating more dynamic bioretention practices. By creating more actively wet and dry areas within a practice these systems allow more targeted removal by promoting diverse environments. These systems have the potential to satisfy both water quality and hydrology performance objectives while deeper understanding of the microbial reactions to RTC allows an understanding of how sensor placement affects performance objectives.

References

- ASCE. 2017. "2017 Infrastructure Report Card: A Comprehensive Assessment of America's Infrastructure." In. Reston, VA.
- Attard, E., F. Poly, C. Commeaux, F. Laurent, A. Terada, B. F. Smets, S. Recous, and X. Le Roux. 2010. 'Shifts between Nitrospira- and Nitrobacter-like nitrite oxidizers underlie the response of soil potential nitrite oxidation to changes in tillage practices', *Environmental Microbiology*, 12: 315-26.
- Barrett, M. E., M. Limouzin, and D. F. Lawler. 2013. 'Effects of Media and Plant Selection on Biofiltration Performance', *Journal of Environmental Engineering-Asce*, 139: 462-70.
- Barros, N., Gomezorellana, I., Feijoo, S., & Balsa, R. (1995). THE EFFECT OF SOIL-MOISTURE ON SOIL MICROBIAL ACTIVITY STUDIED BY MICROCALORIMETRY. *Thermochimica Acta*, 249, 161-168. doi:10.1016/0040-6031(95)90686-x
- Bartos, M., B. Wong, and B. Kerkez. 2018. 'Open storm: a complete framework for sensing and control of urban watersheds', *Environmental Science-Water Research & Technology*, 4: 346-58.
- Bell, C. D., J. M. Wolfand, C. L. Panos, A. S. Bhaskar, R. L. Gilliom, T. S. Hogue, K. G. Hopkins, and A. J. Jefferson. 2020. 'Stormwater control impacts on runoff volume and peak flow: A meta-analysis of watershed modelling studies', *Hydrological Processes*, 34: 3134-52.
- BenDor, T. K., V. Shandas, B. Miles, K. Belt, and L. Olander. 2018. 'Ecosystem services and US stormwater planning: An approach for improving urban stormwater decisions', *Environmental Science & Policy*, 88: 92-103.
- Bi, E. G., F. Monette, and J. Gasperi. 2015. 'Analysis of the influence of rainfall variables on urban effluents concentrations and fluxes in wet weather', *Journal of Hydrology*, 523: 320-32.

- Blecken, G. T., Zinger, Y., Deletic, A., Fletcher, T. D., Hedstrom, A., & Viklander, M. (2010). Laboratory study on stormwater biofiltration Nutrient and sediment removal in cold temperatures. *Journal of Hydrology*, 394(3-4), 507-514. doi:10.1016/j.jhydrol.2010.10.010
- Blecken, G. T., Y. Zinger, A. Deletic, T. D. Fletcher, and M. Viklander. 2009a. 'Impact of a submerged zone and a carbon source on heavy metal removal in stormwater biofilters', *Ecological Engineering*, 35: 769-78.
- Blecken, G. T., Zinger, Y., Deletic, A., Fletcher, T. D., & Viklander, M. (2009). Influence of intermittent wetting and drying conditions on heavy metal removal by stormwater biofilters. *Water Research*, 43(18), 4590-4598. doi:10.1016/j.watres.2009.07.008
- Bosch, D. J., M. B. Wagena, A. C. Ross, A. S. Collick, and Z. M. Easton. 2018. 'Meeting Water Quality Goals under Climate Change in Chesapeake Bay Watershed, USA', *Journal of the American Water Resources Association*, 54: 1239-57.
- Brabec, E., S. Schulte, and P. L. Richards. 2002. 'Impervious surfaces and water quality: A review of current literature and its implications for watershed planning', *Journal of Planning Literature*, 16: 499-514.
- Bratieres, K., Fletcher, T. D., Deletic, A., & Zinger, Y. (2008). Nutrient and sediment removal by stormwater biofilters: A large-scale design optimisation study. *Water Research*, 42(14), 3930-3940. doi:10.1016/j.watres.2008.06.009
- Bronstert, A., D. Niehoff, and G. Burger. 2002. 'Effects of climate and land-use change on storm runoff generation: present knowledge and modelling capabilities', *Hydrological Processes*, 16: 509-29.
- Brown, R. A., and W. F. Hunt. 2011. 'Underdrain Configuration to Enhance Bioretention Exfiltration to Reduce Pollutant Loads', *Journal of Environmental Engineering-Asce*, 137: 1082-91.

- Bullerjahn, G. S., R. M. McKay, T. W. Davis, D. B. Baker, G. L. Boyer, L. V. D'Anglada, G. J. Doucette, J. C. Ho, E. G. Irwin, C. L. Kling, R. M. Kudela, R. Kurmayer, A. M. Michalak, J. D. Ortiz, T. G. Otten, H. W. Paerl, B. Q. Qin, B. L. Sohngen, R. P. Stumpf, P. M. Visser, and S. W. Wilhelm. 2016. 'Global solutions to regional problems: Collecting global expertise to address the problem of harmful cyanobacterial blooms. A Lake Erie case study', *Harmful Algae*, 54: 223-38.
- Burns, M. J., T. D. Fletcher, C. J. Walsh, A. R. Ladson, and B. E. Hatt. 2012. 'Hydrologic shortcomings of conventional urban stormwater management and opportunities for reform', *Landscape and Urban Planning*, 105: 230-40.
- Cadenasso, M. L., S. T. A. Pickett, P. M. Groffman, L. E. Band, G. S. Brush, M. E. Galvin, J. M. Grove, G. Hagar, V. Marshall, B. P. McGrath, J. P. M. Oneil-Dunne, W. P. Stack, and A. R. Troy. 2008. 'Exchanges across land-water-scape boundaries in urban systems - Strategies for reducing nitrate pollution', *Year in Ecology and Conservation Biology 2008*, 1134: 213-32.
- Cameron, K. C., H. J. Di, and J. L. Moir. 2013. 'Nitrogen losses from the soil/plant system: a review', *Annals of Applied Biology*, 162: 145-73.
- Chandrasena, G. I., Shirdashtzadeh, M., Li, Y. L., Deletic, A., Hathaway, J. M., & McCarthy, D. T. (2017). Retention and survival of E. coli in stormwater biofilters: Role of vegetation, rhizosphere microorganisms and antimicrobial filter media. *Ecological Engineering*, 102, 166-177. doi:10.1016/j.ecoleng.2017.02.009
- Chaudry, M. A., & Zwolsman, J. J. G. (2008). Seasonal dynamics of dissolved trace metals in the scheldt estuary: Relationship with redox conditions and phytoplankton activity. *Estuaries and Coasts*, 31(2), 430-443. doi:10.1007/s12237-007-9030-7
- Chen, H. J., and H. Chang. 2014. 'Response of discharge, TSS, and E. coli to rainfall events in urban, suburban, and rural watersheds', *Environmental Science-Processes & Impacts*, 16: 2313-24.

- Chen, T., Y. P. Liu, B. Zhang, and L. H. Sun. 2019. 'Plant rhizosphere, soil microenvironment, and functional genes in the nitrogen removal process of bioretention', *Environmental Science-Processes & Impacts*, 21: 2070-79.
- Chen, X. L., E. Peltier, B. S. M. Sturm, and C. B. Young. 2013. 'Nitrogen removal and nitrifying and denitrifying bacteria quantification in a stormwater bioretention system', *Water Research*, 47: 1691-700.
- Collins, K. A., Lawrence, T. J., Stander, E. K., Jontos, R. J., Kaushal, S. S., Newcomer, T. A., . . . Ekberg, M. L. C. (2010). Opportunities and challenges for managing nitrogen in urban stormwater: A review and synthesis. *Ecological Engineering*, 36(11), 1507-1519. doi:10.1016/j.ecoleng.2010.03.015
- Colmer, T. D. (2003). Long-distance transport of gases in plants: a perspective on internal aeration and radial oxygen loss from roots. *Plant Cell and Environment*, 26(1), 17-36. doi:10.1046/j.1365-3040.2003.00846.x
- Cook, E. R., C. A. Woodhouse, C. M. Eakin, D. M. Meko, and D. W. Stahle. 2004. 'Long-term aridity changes in the western United States', *Science*, 306: 1015-18.
- CSN. 2013. "Bioretention Illustrated: A Visual Guide for Constructing, Inspecting, Maintaining and Verifying the Bioretention Practice." In. Prince William County, VA: Chesapeake Stormwater Network.
- Dagenais, D., J. Brisson, and T. D. Fletcher. 2018. 'The role of plants in bioretention systems; does the science underpin current guidance?', *Ecological Engineering*, 120: 532-45.
- Daims, H., E. V. Lebedeva, P. Pjevac, P. Han, C. Herbold, M. Albertsen, N. Jehmlich, M. Palatinszky, J. Vierheilig, A. Bulaev, R. H. Kirkegaard, M. von Bergen, T. Rattei, B. Bendinger, P. H. Nielsen, and M. Wagner. 2015. 'Complete nitrification by *Nitrospira* bacteria', *Nature*, 528: 504-+.

- Davis, A. P., Shokouhian, M., Sharma, H., & Minami, C. (2006). Water quality improvement through bioretention media: Nitrogen and phosphorus removal. *Water Environment Research*, 78(3), 284-293. doi:10.2175/106143005x94376
- Davis, A. P., M. Shokouhian, H. Sharma, C. Minami, and D. Winogradoff. 2003. 'Water quality improvement through bioretention: Lead, copper, and zinc removal', *Water Environment Research*, 75: 73-82.
- Davis, A. P., W. F. Hunt, R. G. Traver, and M. Clar. 2009. 'Bioretention Technology: Overview of Current Practice and Future Needs', *Journal of Environmental Engineering*, 135: 109-17.
- Davis, A. P., R. G. Traver, W. F. Hunt, R. Lee, R. A. Brown, and J. M. Olszewski. 2012. 'Hydrologic Performance of Bioretention Storm-Water Control Measures', *Journal of Hydrologic Engineering*, 17: 604-14.
- Davis, T. W., R. Stumpf, G. S. Bullerjahn, R. M. L. McKay, J. D. Chaffin, T. B. Bridgeman, and C. Winslow. 2019. 'Science meets policy: A framework for determining impairment designation criteria for large waterbodies affected by cyanobacterial harmful algal blooms', *Harmful Algae*, 81: 59-64.
- Delpla, I., A. V. Jung, E. Baures, M. Clement, and O. Thomas. 2009. 'Impacts of climate change on surface water quality in relation to drinking water production', *Environment International*, 35: 1225-33.
- Deng, Qinghui, Lingling Wan, Xiaowen Li, Xiuyun Cao, Yiyong Zhou, and Chunlei Song. 2020. 'Metagenomic evidence reveals denitrifying community diversity rather than abundance drives nitrate removal in stormwater biofilters amended with different organic and inorganic electron donors', *Chemosphere*, 257: 127269.
- Dietz, M. E., & Clausen, J. C. (2005). A field evaluation of rain garden flow and pollutant treatment. *Water Air and Soil Pollution*, 167(1-4), 123-138. doi:10.1007/s11270-005-8266-8

- Ebrahimian, A., B. Wadzuk, and R. Traver. 2019. 'Evapotranspiration in green stormwater infrastructure systems', *Science of the Total Environment*, 688: 797-810.
- Fierer, N., & Schimel, J. P. (2002). Effects of drying-rewetting frequency on soil carbon and nitrogen transformations. *Soil Biology & Biochemistry*, 34(6), 777-787. doi:10.1016/s0038-0717(02)00007-x
- Fletcher, T. D., W. Shuster, W. F. Hunt, R. Ashley, D. Butler, S. Arthur, S. Trowsdale, S. Barraud, A. Semadeni-Davies, J. L. Bertrand-Krajewski, P. S. Mikkelsen, G. Rivard, M. Uhl, D. Dagenais, and M. Viklander. 2015. 'SUDS, LID, BMPs, WSUD and more - The evolution and application of terminology surrounding urban drainage', *Urban Water Journal*, 12: 525-42.
- Gamalero, E., E. Bona, V. Todeschini, and G. Lingua. 2020. 'Saline and Arid Soils: Impact on Bacteria, Plants, and Their Interaction', *Biology-Basel*, 9: 26.
- Gilchrist, S., M. Borst, and E. K. Stander. 2014. 'Factorial Study of Rain Garden Design for Nitrogen Removal', *Journal of Irrigation and Drainage Engineering*, 140: 11.
- Glaister, B. J., Fletcher, T. D., Cook, P. L. M., & Hatt, B. E. (2017). Interactions between design, plant growth and the treatment performance of stormwater biofilters. *Ecological Engineering*, 105, 21-31. doi:10.1016/j.ecoleng.2017.04.030
- Goh, H. W., K. S. Lem, N. A. Azizan, C. K. Chang, A. Talei, C. S. Leow, and N. A. Zakaria. 2019. 'A review of bioretention components and nutrient removal under different climates'
- Guo, Y. P., & Adams, B. J. (1998). Hydrologic analysis of urban catchments with event-based probabilistic models - 1. Runoff volume. *Water Resources Research*, 34(12), 3421-3431. doi:10.1029/98wr02449
- HamiltonTaylor, J., Davison, W., & Morfett, K. (1996). A laboratory study of the biogeochemical cycling of Fe, Mn, Zn and Cu across the sediment-water interface of a productive lake. *Aquatic Sciences*, 58(3), 191-209. doi:10.1007/bf00877508

- Hamel, P., and T. D. Fletcher. 2014. 'The impact of stormwater source-control strategies on the (low) flow regime of urban catchments', *Water Science and Technology*, 69: 739-45.
- Hathaway, J. M., & Hunt, W. F. (2012). Indicator Bacteria Performance of Storm Water Control Measures in Wilmington, North Carolina. *Journal of Irrigation and Drainage Engineering*, 138(2), 185-197. doi:10.1061/(asce)ir.1943-4774.0000378
- Hathaway, J. M., R. A. Brown, J. S. Fu, and W. F. Hunt. 2014. 'Bioretention function under climate change scenarios in North Carolina, USA', *Journal of Hydrology*, 519: 503-11.
- Hatt, B. E., A. Deletic, and T. D. Fletcher. 2007. 'Stormwater reuse: designing biofiltration systems for reliable treatment', *Water Science and Technology*, 55: 201-09.
- Hatt, B. E., T. D. Fletcher, and A. Deletic. 2007. 'Treatment performance of gravel filter media: Implications for design and application of stormwater infiltration systems', *Water Research*, 41: 2513-24.
- . 2009a. 'Hydrologic and pollutant removal performance of stormwater biofiltration systems at the field scale', *Journal of Hydrology*, 365: 310-21.
- . 2009b. 'Pollutant removal performance of field-scale stormwater biofiltration systems', *Water Science and Technology*, 59: 1567-76.
- Henderson, C., Greenway, M., & Phillips, I. (2007). Removal of dissolved nitrogen, phosphorus and carbon from stormwater by biofiltration mesocosms. *Water Science and Technology*, 55(4), 183-191. doi:10.2166/wst.2007.108
- Hogan, D. M., S. T. Jarnagin, J. V. Loperfido, and K. Van Ness. 2014. 'Mitigating the Effects of Landscape Development on Streams in Urbanizing Watersheds', *Journal of the American Water Resources Association*, 50: 163-78.

- Hopkins, K. G., A. S. Bhaskar, S. A. Woznicki, and R. M. Fanelli. 2019. 'Changes in event-based streamflow magnitude and timing after suburban development with infiltration-based stormwater management', *Hydrological Processes*: 17.
- Houdeshel, C. D., C. A. Pomeroy, and K. R. Hultine. 2012. 'Bioretention Design for Xeric Climates Based on Ecological Principles', *Journal of the American Water Resources Association*, 48: 1178-90.
- Hsieh, C. H., Davis, A. P., & Needelman, B. A. (2007). Nitrogen removal from urban stormwater runoff through layered bioretention columns. *Water Environment Research*, 79(12), 2404-2411. doi:10.2175/106143007x183844
- Hunt, W. F., A. P. Davis, and R. G. Traver. 2012. 'Meeting Hydrologic and Water Quality Goals through Targeted Bioretention Design', *Journal of Environmental Engineering-Asce*, 138: 698-707.
- Hunt, W. F., Smith, J. T., Jadlocki, S. J., Hathaway, J. M., & Eubanks, P. R. (2008). Pollutant removal and peak flow mitigation by a bioretention cell in urban Charlotte, NC. *Journal of Environmental Engineering-Asce*, 134(5), 403-408. doi:10.1061/(asce)0733-9372(2008)134:5(403)
- Igielski, S., B. V. Kjellerup, and A. P. Davis. 2019. 'Understanding urban stormwater denitrification in bioretention internal water storage zones', *Water Environment Research*, 91: 32-44.
- Jay, J. G., M. Tyler-Plog, S. L. Brown, and F. Grothkopp. 2019. 'Nutrient, Metal, and Organics Removal from Stormwater Using a Range of Bioretention Soil Mixtures', *Journal of Environmental Quality*, 48: 493-501.
- Juan, A., C. Hughes, Z. Fang, and P. Bedient. 2017. 'Hydrologic Performance of Watershed-Scale Low-Impact Development in a High-Intensity Rainfall Region', *Journal of Irrigation and Drainage Engineering*, 143: 11.

- Kerkez, B., M. Daniels, S. Graves, V. Chandrasekar, K. Keiser, C. Martin, M. Dye, M. Maskey, and F. Vernon. 2016. 'Cloud Hosted Real-time Data Services for the Geosciences (CHORDS)', *Geoscience Data Journal*, 3: 4-8.
- Kerkez, B., C. Gruden, M. Lewis, L. Montestruque, M. Quigley, B. Wong, A. Bedig, R. Kertesz, T. Braun, O. Cadwalader, A. Poresky, and C. Pak. 2016. 'Smarter Stormwater Systems', *Environmental Science & Technology*, 50: 7267-73.
- Kirkham, M. B. (2005). *Field Capacity, Wilting Point, Available Water, and the Non-Limiting Water Range*. Amsterdam: Elsevier Science Bv.
- Kobielska, P. A., A. J. Howarth, O. K. Farha, and S. Nayak. 2018. 'Metal-organic frameworks for heavy metal removal from water', *Coordination Chemistry Reviews*, 358: 92-107.
- Konrad, Christopher P. 2003. "Effects of Urban Development on Floods." In.: USGS.
- Larsen, T. A., Hoffmann, S., Luthi, C., Truffer, B., & Maurer, M. (2016). Emerging solutions to the water challenges of an urbanizing world. *Science*, 352(6288), 928-933. doi:10.1126/science.aad8641
- Laurenson, G., Laurenson, S., Bolan, N., Beecham, S., & Clark, I. (2013). The Role of Bioretention Systems in the Treatment of Stormwater. In D. L. Sparks (Ed.), *Advances in Agronomy, Vol 120* (Vol. 120, pp. 223-274). San Diego: Elsevier Academic Press Inc.
- Le Coustumer, S., Fletcher, T. D., Deletic, A., Barraud, S., & Poelsma, P. (2012). The influence of design parameters on clogging of stormwater biofilters: A large-scale column study. *Water Research*, 46(20), 6743-6752. doi:10.1016/j.watres.2012.01.026
- Lee, E. Y., Noh, S. R., Cho, K. S., & Ryu, H. W. (2001). Leaching of Mn, Co, and Ni from manganese nodules using an anaerobic bioleaching method. *Journal of Bioscience and Bioengineering*, 92(4), 354-359. doi:10.1263/jbb.92.354
- Levin, L. A., and A. S. Mehring. 2015. 'Optimization of bioretention systems through application of ecological theory', *Wiley Interdisciplinary Reviews-Water*, 2: 259-70.

- Li, C. L., M. Liu, Y. M. Hu, M. Zong, M. H. Zhao, and M. T. Walter. 2019. 'Characteristics of impervious surface and its effect on direct runoff: a case study in a rapidly urbanized area', *Water Science and Technology-Water Supply*, 19: 1885-91.
- Li, H., L. J. Sharkey, W. F. Hunt, and A. P. Davis. 2009. 'Mitigation of Impervious Surface Hydrology Using Bioretention in North Carolina and Maryland', *Journal of Hydrologic Engineering*, 14: 407-15.
- Li, M. H., Swapp, M., Kim, M. H., Chu, K. H., & Sung, C. Y. (2014). Comparing Bioretention Designs With and Without an Internal Water Storage Layer for Treating Highway Runoff. *Water Environment Research*, 86(5), 387-397. doi:10.2175/106143013x13789303501920
- Liu, A., P. Egodawatta, Y. T. Guan, and A. Goonetilleke. 2013. 'Influence of rainfall and catchment characteristics on urban stormwater quality', *Science of the Total Environment*, 444: 255-62.
- Liu, J., D. J. Sample, C. Bell, and Y. T. Guan. 2014. 'Review and Research Needs of Bioretention Used for the Treatment of Urban Stormwater', *Water*, 6: 1069-99.
- Liu, R. F., and E. Fassman-Beck. 2017. 'Hydrologic experiments and modeling of two laboratory bioretention systems under different boundary conditions', *Frontiers of Environmental Science & Engineering*, 11: 10.
- Lovley, D. R., Holmes, D. E., & Nevin, K. P. (2004). Dissimilatory Fe(III) and Mn(IV) reduction. In R. K. Poole (Ed.), *Advances in Microbial Physiology, Vol. 49* (Vol. 49, pp. 219-286). London: Academic Press Ltd-Elsevier Science Ltd.
- Luo, Y. H., X. P. Yue, Y. Q. Duan, A. J. Zhou, Y. J. Gao, and X. Zhang. 2020. 'A bilayer media bioretention system for enhanced nitrogen removal from road runoff', *Science of the Total Environment*, 705: 8.
- Madigan, M., Bender, K., Buckley, D., Sattley, W., Stahl, D. . 2019. *Brock Biology of Microorganisms (15th Global Edition ed.)* (Pearson).

- Mangangka, I. R., A. Liu, P. Egodawatta, and A. Goonetilleke. 2015. 'Performance characterisation of a stormwater treatment bioretention basin', *Journal of Environmental Management*, 150: 173-78.
- Manka, B. N., Hathaway, J. M., Tirpak, R. A., He, Q., & Hunt, W. F. (2016). Driving forces of effluent nutrient variability in field scale bioretention. *Ecological Engineering*, 94, 622-628. doi:10.1016/j.ecoleng.2016.06.024
- McPhillips, L., Goodale, C., & Walter, M. T. (2018). Nutrient Leaching and Greenhouse Gas Emissions in Grassed Detention and Bioretention Stormwater Basins. *Journal of Sustainable Water in the Built Environment*, 4(1), 10. doi:10.1061/jswbay.0000837
- Meerow, S., and J. P. Newell. 2017. 'Spatial planning for multifunctional green infrastructure: Growing resilience in Detroit', *Landscape and Urban Planning*, 159: 62-75.
- Middleton, J. R., and M. E. Barrett. 2008. 'Water quality performance of a batch-type stormwater detention basin', *Water Environment Research*, 80: 172-78.
- Mishra, A. K., and V. P. Singh. 2010. 'A review of drought concepts', *Journal of Hydrology*, 391: 204-16.
- Morse, N., Payne, E., Henry, R., Hatt, B., Chandrasena, G., Shapleigh, J., . . . McCarthy, D. (2018). Plant-Microbe Interactions Drive Denitrification Rates, Dissolved Nitrogen Removal, and the Abundance of Denitrification Genes in Stormwater Control Measures. *Environmental Science & Technology*, 52(16), 9320-9329. doi:10.1021/acs.est.8b02133
- Mullapudi, A., Bartos, M., Wong, B., & Kerkez, B. (2018). Shaping Streamflow Using a Real-Time Stormwater Control Network. *Sensors*, 18(7), 11. doi:10.3390/s18072259
- Mullapudi, A., Wong, B. P., & Kerkez, B. (2017). Emerging investigators series: building a theory for smart stormwater systems. *Environmental Science-Water Research & Technology*, 3(1), 66-77. doi:10.1039/c6ew00211k
- NCDEQ, N. C. D. o. E. Q. (2018). Stormwater Design Manual. *North Carolina Environmental Quality*.

<https://files.nc.gov/ncdeq/Energy+Mineral+and+Land+Resources/Stormwater/BMP+Manual/C-2%20%20Bioretention%201-19-2018%20FINAL.pdf>

- Nealson, K. H., & Saffarini, D. (1994). IRON AND MANGANESE IN ANAEROBIC RESPIRATION - ENVIRONMENTAL SIGNIFICANCE, PHYSIOLOGY, AND REGULATION. *Annual Review of Microbiology*, 48, 311-343. doi:10.1146/annurev.mi.48.100194.001523
- O’Gorman, Paul A. 2015. 'Precipitation Extremes Under Climate Change', *Current Climate Change Reports*, 1: 49-59.
- Pajares, S., and B. J. M. Bohannan. 2016. 'Ecology of Nitrogen Fixing, Nitrifying, and Denitrifying Microorganisms in Tropical Forest Soils', *Frontiers in Microbiology*, 7: 20.
- Palmer, E. T., C. J. Poor, C. Hinman, and J. D. Stark. 2013. 'Nitrate and Phosphate Removal through Enhanced Bioretention Media: Mesocosm Study', *Water Environment Research*, 85: 823-32.
- Parolari, A. J., Pelrine, S., & Bartlett, M. S. (2018). Stochastic water balance dynamics of passive and controlled stormwater basins. *Advances in Water Resources*, 122, 328-339. doi:10.1016/j.advwatres.2018.10.016
- Passeport, E., W. F. Hunt, D. E. Line, R. A. Smith, and R. A. Brown. 2009. 'Field Study of the Ability of Two Grassed Bioretention Cells to Reduce Storm-Water Runoff Pollution', *Journal of Irrigation and Drainage Engineering*, 135: 505-10.
- Payne, E. G. I., Fletcher, T. D., Cook, P. L. M., Deletic, A., & Hatt, B. E. (2014). Processes and Drivers of Nitrogen Removal in Stormwater Biofiltration. *Critical Reviews in Environmental Science and Technology*, 44(7), 796-846. doi:10.1080/10643389.2012.741310
- Peralta, A. L., S. Ludmer, and A. D. Kent. 2013. 'Hydrologic history influences microbial community composition and nitrogen cycling under experimental drying/wetting treatments', *Soil Biology & Biochemistry*, 66: 29-37.

- Persaud, P. P., A. A. Akin, B. Kerkez, D. T. McCarthy, and J. M. Hathaway. 2019. 'Real time control schemes for improving water quality from bioretention cells', *Blue-Green Systems*, 1: 55-71.
- Placella, S. A., E. L. Brodie, and M. K. Firestone. 2012. 'Rainfall-induced carbon dioxide pulses result from sequential resuscitation of phylogenetically clustered microbial groups', *Proceedings of the National Academy of Sciences of the United States of America*, 109: 10931-36.
- Pulleman, M., & Tietema, A. (1999). Microbial C and N transformations during drying and rewetting of coniferous forest floor material. *Soil Biology & Biochemistry*, 31(2), 275-285. doi:10.1016/s0038-0717(98)00116-3
- Read, J., T. D. Fletcher, T. Wevill, and A. Deletic. 2010. 'Plant Traits that Enhance Pollutant Removal from Stormwater in Biofiltration Systems', *International Journal of Phytoremediation*, 12: 34-53.
- Read, J., T. Wevill, T. Fletcher, and A. Deletic. 2008. 'Variation among plant species on pollutant removal from stormwater non biofiltration systems', *Water Research*, 42: 893-902.
- Roman, D., A. Braga, N. Shetty, and P. Culligan. 2017. 'Design and Modeling of an Adaptively Controlled Rainwater Harvesting System', *Water*, 9: 14.
- Salazar, A., Sulman, B. N., & Dukes, J. S. (2018). Microbial dormancy promotes microbial biomass and respiration across pulses of drying-wetting stress. *Soil Biology & Biochemistry*, 116, 237-244. doi:10.1016/j.soilbio.2017.10.017
- Schimel, J. P. (2018). Life in Dry Soils: Effects of Drought on Soil Microbial Communities and Processes. In D. J. Futuyma (Ed.), *Annual Review of Ecology, Evolution, and Systematics*, Vol 49 (Vol. 49, pp. 409-432). Palo Alto: Annual Reviews.
- Shen, P. F., A. Deletic, K. Bratieres, and D. T. McCarthy. 2020. 'Real time control of biofilters delivers stormwater suitable for harvesting and reuse', *Water Research*, 169: 10.

- Steffen, M. M., T. W. Davis, R. M. L. McKay, G. S. Bullerjahn, L. E. Krausfeldt, J. M. A. Stough, M. L. Neitzey, N. E. Gilbert, G. L. Boyer, T. H. Johengen, D. C. Gossiaux, A. M. Burtner, D. Palladino, M. D. Rowe, G. J. Dick, K. A. Meyer, S. Levy, B. E. Boone, R. P. Stumpf, T. T. Wynne, P. V. Zimba, D. Gutierrez, and S. W. Wilhelm. 2017. 'Ecophysiological Examination of the Lake Erie Microcystis Bloom in 2014: Linkages between Biology and the Water Supply Shutdown of Toledo, OH', *Environmental Science & Technology*, 51: 6745-55.
- Sukenik, A., O. Hadas, A. Kaplan, and A. Quesada. 2012. 'Invasion of Nostocales (cyanobacteria) to subtropical and temperate freshwater lakes - physiological, regional, and global driving forces', *Frontiers in Microbiology*, 3: 9.
- Sun, Y. W., C. Pomeroy, Q. Y. Li, and C. D. Xu. 2019. 'Impacts of rainfall and catchment characteristics on bioretention cell performance', *Water Science and Engineering*, 12: 98-107.
- Tang, N. Y., & Tian, L. (2016). Nitrogen removal by three types of bioretention columns under wetting and drying regimes. *Journal of Central South University*, 23(2), 324-332. doi:10.1007/s11771-016-3077-1
- Tauhid, F. A., and H. Zawani. 2018. 'Mitigating Climate Change Related Floods in Urban Poor Areas: Green Infrastructure Approach', *Journal of Regional and City Planning*, 29: 98-112.
- TDEC. 2014. "Tennessee Permanent Stormwater Management and Design Guidance Manual." In, 141-76.
- Thompson, A. M., A. C. Paul, and N. J. Balster. 2008. 'Physical and hydraulic properties of engineered soil media for bioretention basins', *Transactions of the Asabe*, 51: 499-514.
- TSM, T. S. M. (2015). Management and Design Guidance Manual. *Tennessee Stormwater Managment*. Retrieved from <https://tnpermanentstormwater.org/manual.asp#>
- USEPA. 2002. "Federal Water Pollution Control Act." In, edited by USEPA. Washington, D.C.
- . 2009. 'TMDL Program Results Analysis Fact Sheet', USEPA Office of Water.

———. 2010. "TMDL Program Results Analysis Fact Sheet." In. Washington, DC: EPA Office of Water.

———. 2015. "Central Tenets of the National Pollutant Discharge Elimination System (NPDES) Permitting

Program." In, edited by USEPA. Washington, D.C.

USEPA, United States Environmental Protection Agency. 2016. "National Summary of State Information." In *USEPA*.

Vangestel, M., Merckx, R., & Vlassak, K. (1993). MICROBIAL BIOMASS RESPONSES TO SOIL DRYING AND REWETTING - THE FATE OF FAST-GROWING AND SLOW-GROWING MICROORGANISMS IN SOILS FROM DIFFERENT CLIMATES. *Soil Biology & Biochemistry*, 25(1), 109-123. doi:10.1016/0038-0717(93)90249-b

Waller, L. J., Evanylo, G. K., Krometis, L. A. H., Strickland, M. S., Wynn-Thompson, T., & Badgley, B. D. (2018). Engineered and Environmental Controls of Microbial Denitrification in Established Bioretention Cells. *Environmental Science & Technology*, 52(9), 5358-5366. doi:10.1021/acs.est.7b06704

Wang, J., L. H. C. Chua, and P. Shanahan. 2017. 'Evaluation of pollutant removal efficiency of a bioretention basin and implications for stormwater management in tropical cities', *Environmental Science-Water Research & Technology*, 3: 78-91.

Wang, J. L., Y. L. Zhao, L. Q. Yang, N. N. Tu, G. P. Xi, and X. Fang. 2017. 'Removal of Heavy Metals from Urban Stormwater Runoff Using Bioretention Media Mix', *Water*, 9: 22.

Wang, M., D. Q. Zhang, S. W. Lou, Q. H. Hou, Y. J. Liu, Y. N. Cheng, J. D. Qi, and S. K. Tan. 2019. 'Assessing Hydrological Effects of Bioretention Cells for Urban Stormwater Runoff in Response to Climatic Changes', *Water*, 11: 20.

- Willard, L. L., T. Wynn-Thompson, L. H. Krometis, T. P. Neher, and B. D. Badgley. 2017. 'Does It Pay to be Mature? Evaluation of Bioretention Cell Performance Seven Years Postconstruction', *Journal of Environmental Engineering*, 143: 10.
- Willems, P. 2013. 'Revision of urban drainage design rules after assessment of climate change impacts on precipitation extremes at Uccle, Belgium', *Journal of Hydrology*, 496: 166-77.
- Winston, R. J., J. D. Dorsey, and W. F. Hunt. 2016. 'Quantifying volume reduction and peak flow mitigation for three bioretention cells in clay soils in northeast Ohio', *Science of the Total Environment*, 553: 83-95.
- Wright, O. M., E. Istanbuluoglu, R. R. Horner, C. L. DeGasperi, and J. Simmonds. 2018. 'Is there a limit to bioretention effectiveness? Evaluation of stormwater bioretention treatment using a lumped urban ecohydrologic model and ecologically based design criteria', *Hydrological Processes*, 32: 2318-34.
- Xia, W. W., C. X. Zhang, X. W. Zeng, Y. Z. Feng, J. H. Weng, X. G. Lin, J. G. Zhu, Z. Q. Xiong, J. Xu, Z. C. Cai, and Z. J. Jia. 2011. 'Autotrophic growth of nitrifying community in an agricultural soil', *Isme Journal*, 5: 1226-36.
- Yan, Q., James, B. R., & Davis, A. P. (2017). Lab-Scale Column Studies for Enhanced Phosphorus Sorption from Synthetic Urban Stormwater Using Modified Bioretention Media. *Journal of Environmental Engineering*, 143(1), 13. doi:10.1061/(asce)ee.1943-7870.0001159
- Yang, Y., and T. F. M. Chui. 2018. 'Optimizing surface and contributing areas of bioretention cells for stormwater runoff quality and quantity management', *Journal of Environmental Management*, 206: 1090-103.

- Zhang, J. Y., R. P. Singh, Y. Z. Liu, and D. F. Fu. 2019. 'Design and operation of submerged layer in bioretention for enhanced nitrate removal', *Journal of Water Supply Research and Technology-Aqua*, 68: 744-56.
- Zhou, X., Fornara, D., Ikenaga, M., Akagi, I., Zhang, R. F., & Jia, Z. J. (2016). The Resilience of Microbial Community under Drying and Rewetting Cycles of Three Forest Soils. *Frontiers in Microbiology*, 7, 12. doi:10.3389/fmicb.2016.01101
- Zumft, W. G. 1997. 'Cell biology and molecular basis of denitrification', *Microbiology and Molecular Biology Reviews*, 61: 533-+.

Appendices

Appendix A: Functional Genes Triggered by Real-time Control in Bioretention Practices: An Analysis of Stratification and Water Quality Implications

Research Question:

What functional genes are present in the layers of traditional and RTC bioretention practices? How can design changes target specific microbial functions

Introduction:

Eutrophication can occur when excess nitrogen enters surface waters via urban stormwater (Cadenasso et al. 2008). For example, stormwater has been identified as a contributor to the degradation of valuable resources such as the Chesapeake Bay, leading to large, multi-state efforts to ameliorate these effects. Bioretention practices have proven effective for reducing nitrogen compounds in stormwater through biotic processing, however complete nitrogen reduction (all the way through to nitrogen gas) is not always realized, with incidents of nitrate leaching having been documented due to nitrate mobilization from soils, and microbial conversions of ammonia to nitrate that do not progress to denitrification due to a lack of anoxic conditions (Manka et al. 2016; Davis et al. 2006; Hsieh et al. 2007; Blecken et al. 2010). To combat this, efforts have taken place to enhance biological processing in bioretention by promoting more aerobic and anaerobic zones within the same practice through the creation of internal water storage. This has been effective in reducing effluent nitrate concentrations so that systems are no longer exporting (Hunt et al. 2008; Brown and Hunt 2011), however, a specific optimal internal storage depth has not been defined. Further, such systems may introduce hydrologic deficiencies, depending on in situ soils, by reducing storage capacity.

In an examination of biological processing, studies indicate plants, soil moisture, temperature, and microbial communities contribute to the treatment of nitrogen species (Passeport et al. 2009; Blecken et al. 2010). Nitrification and denitrification in the nitrogen cycle require sequential aerobic and anaerobic environments to convert NH_4^+ to $\text{N}_{2(\text{g})}$ although there is the potential for the formation of $\text{N}_2\text{O}_{(\text{g})}$, which is a greenhouse gas (Collins et al. 2010). Understanding the consequences of emitting greenhouse gasses and the threat of eutrophication from incomplete processing has prompted research into the inner workings of bioretention systems. Studies tracking the transformations of nitrogen compounds have primarily dealt with differences in influent and effluent concentrations through “black box” approaches and related these differences to various design properties like plants, soil type, and vertical stratification (Henderson et al. 2007; Read et al. 2008; Read et al. 2010; Chen et al. 2019). While the influence of these parameters has been observed, more process-based studies may further elucidate the pathways being triggered by design type. The few studies that have focused on targeting microbial functioning within bioretention have focused on denitrifying bacteria to understand conditions that result in incomplete processing in traditional practices (Waller et al. 2018; Morse et al. 2018). In particular, microbial communities targeted included *Acidobacteria*, *Clostridia*, *Betaproteobacteria* and *Alphaproteobacteria* because of the functional genes (*amoA*, *narG*, *nirS*, *nirK*, *nosZ*, and *norB*) that facilitate nitrification and denitrification (Chen et al. 2019; Chen et al. 2013). In these studies, recommendations for future design called for optimizing internal water storage as the practice consistently reduced nitrate levels in bioretention effluent (Igielski et al. 2019). It should also be noted that prior research examining bioretention through column studies have not included an avenue to mimic seepage, which will alter the soil moisture dynamics within all treatment types and affect microbial functioning.

Using control technologies to enhance bioretention design could allow advances beyond IWS; it could allow enhanced microbial processing while balancing hydrologic benefits. Real-time Control (RTC) is an emerging adaptation in green infrastructure that could further promote microbial processing when conditions like soil moisture are targeted in an optimized design (Kerkez et al. 2016). The technology has been used to optimize watershed hydrology (Mullapudi et al. 2018), improve water harvesting systems (Roman et al. 2017), and improve water quality in detention basins (Middleton and Barrett 2008). As an example, research investigating water quality effects in detention basins, have shown reductions in total suspended solids (TSS) due to longer retention times (Middleton and Barrett 2008). Further, previous work by Persaud et al. (2019) has shown water quality applications of RTC specifically related to bioretention practices where nitrogen processing, while not complete, showed more balance between nitrification and denitrification. Understanding the implications of using RTC in bioretention to promote nitrification and denitrification will also require an analysis of the functional genes triggered by resulting system changes.

To provide further understanding of the effects of RTC systems on microbial communities, a full understanding of the genes expressed in various bioretention schemes should be developed to understand the processing pathways being triggered by different treatment types. A six-week column study was conducted in which comparisons between genes and bioretention design variations are made. This research will focus on 1) understanding the functional genes triggered by the use of RTC systems in bioretention, 2) analyzing the internal conditions that can result in incomplete denitrification, and 3) suggesting further optimization parameters to use in future RTC configurations. This study will further the capability of bioretention practices to manage nitrogen species in stormwater.

Methods:

This research was conducted over a six-week period during which a column study was performed to compare free draining (FD), internal water storage (IWS), and RTC bioretention practices. The RTC designs were based on internal soil moisture (SM) and volume management (VC). This research builds off of the column studies outlined in Chapters 3 and 4 by including metagenomic analysis. The goal of this work is to study the functional genes triggered in the upper and lower portions of soil media that allow microbial species to perform nitrification and denitrification and understand how different treatment types can trigger specific genes. This will allow a more detailed understanding of the specific pathways for nitrification and denitrification in bioretention practices.

Column Design

The columns are described in detail in Chapter 2. Briefly, each column design was replicated five times with the outlet structures being the only difference in configuration. Traditional columns were designed to function as FD and IWS systems while RTC columns operated based on specific SM and VC objectives. The soil moisture target used in this study is the field capacity of the bioretention media as this is optimal for microbial function (Barros et al. 1995). Each column was built and operated in accordance with design descriptions given in Chapter 3; however, sampling ports were installed in the upper and lower portions of the columns at 30 cm and 90 cm from the bottom of the column (Figure A.1).

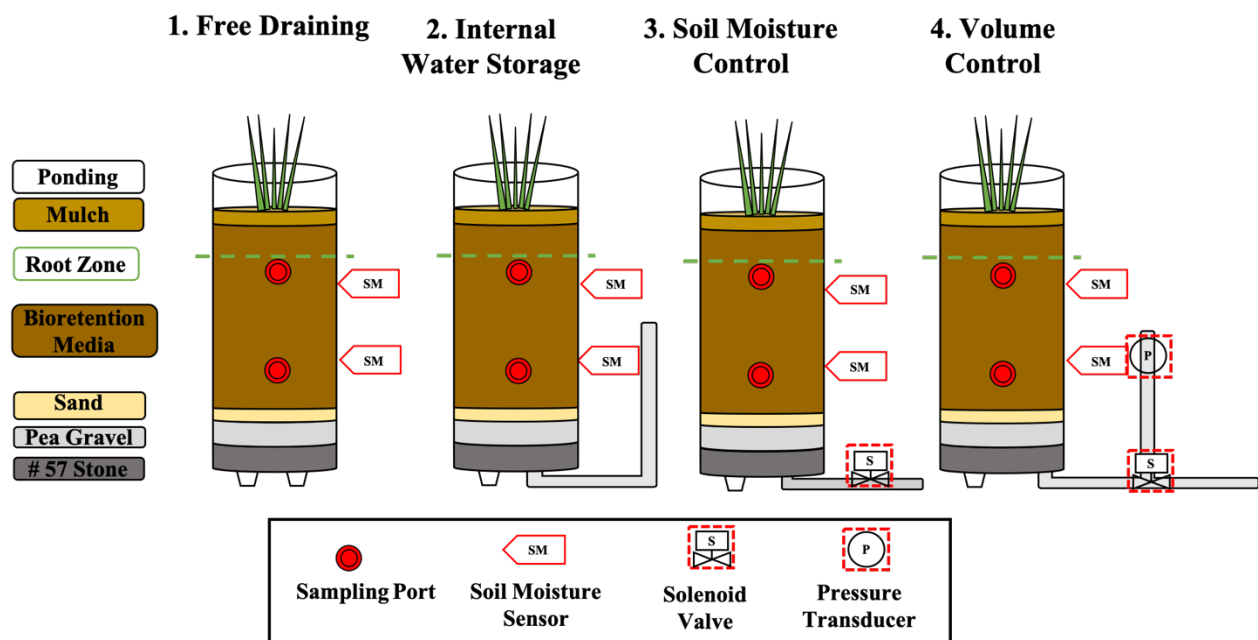


Figure A.1 Column designs that feature sampling port locations for soil core extractions following the 6-week column study.

Experimental Procedure:

Over the course of this six-week study, artificial rainfall was applied to the bioretention columns in accordance with historic rainfall from August 8th to September 19th, 2018 and matched with rainfall prediction data obtained by NOAA. These columns were intended to test the functionality of RTC compared to traditional bioretention design in an effort to further understand and optimize RTC strategies. Twelve hours prior to an event, weather predictions were sent to warn RTC columns of incoming precipitation and allow them to adjust their internal conditions in anticipation of weather (in accordance with their control schemes). Due to weather uncertainty, it should be noted that there were occasions where a prediction was sent to columns with no rain event following, as well as occasions where no prediction occurred but a rain event (and thus a stormwater application) happened. A synthetic stormwater mixture outlined by Bratieres et al. (2008) was used to dose columns based on a 1:20 sizing ratio for the practice which resulted in 12 application events over the course of this study (TSM 2015). Further description of the experimental design is provided in Chapter 2.

Sample Collection and Preparation:

For each prescribed application, water samples were taken of the influent (from the stormwater mixing tank) and effluent from each column. These samples were analyzed for nutrients and 16S analysis as described in Chapter 3. At the end of the study, soil cores were taken from the top and bottom of three columns of each treatment for a total of 24 soil samples to be used in metagenomic analysis. Soil cores from sampling ports built into the side of each column were obtained using autoclaved stainless-steel straws. Each straw was inserted into the soil media through the entire width of the column to allow representative sampling across the column cross-

section. Each sample was then stored in a cooler for transportation and kept frozen at -20 °C until DNA extraction was performed. For each soil sample, DNA extraction was completed per manufacturer's instructions using a DNeasy PowerSoil Kit by Qiagen. Samples were then processed through the University of Tennessee Center for Environmental Biotechnology for metagenomic analysis. Nextera XT library preparation kits by Illumina were used to prepare metagenomic sequencing libraries following manufacturer instructions. A MiSeq and MiSeq V3 (600 cycle) Reagent kit was used to denature and dilute libraries for sequencing. BBDuk was then used to analyze the data and determine gene counts for genes of interest (Morse et al. 2018).

Statistical Analysis:

Specific genes targeted for analysis are included in Table A.1. Data in this research was analyzed using R version 4.0.3 and all statistic testing was performed at a 95% confidence interval. The Kolmogorov-Smirnov test was used to confirm a non-normal distribution. The Wilcoxon sign rank test was then used to determine differences between similar columns and differences between samples taken from different locations in the same columns. Wilcoxon rank sum tests were then used to analyze differences between all column types.

Results:

The nitrogen cycle is consistent of fixation, ammonification, nitrification, and denitrification. Fixation is the conversion of nitrogen gas into nitrates and nitrites, which is rarely discussed or considered in bioretention literature. Ammonification is the conversion of nitrogen in its organic forms (plant or animal waste) into ammonium (NH_4^+). Nitrification then converts that ammonium into nitrites and then into nitrates. Finally, Denitrification converts nitrates back into

nitrogen gas. The individual genes analyzed in this research target the five conversions as part of nitrification and denitrification Table A.1.

Soil samples from the upper and lower portions of each column type were used in metagenomic testing to determine the functional genes found in each column (Table A.3, Figure A.2, and Figure A.3). These gene read counts are indicative of gene activity, however they are not a direct measure of activity. RNA sequencing would have allowed a view of transcription which would be more directly linked to gene activity; however, the RNA analysis method is both costly and difficult. Overall, it can be noted that samples from the upper portion of the soil profile had higher read counts than those of the lower portion for all column types except SM, which exhibited opposite trends. Differences between traditional and RTC designs also show more read counts in the traditional columns than the RTC columns. Further descriptions of each conversion are discussed based on the associated gene.

Nitrification ($\text{NH}_4^+ \rightarrow \text{NO}_3^-$):

Converting ammonium to nitrate is performed by the *amoA* gene which can be found in *Nitrospira* bacteria. Significant differences in read counts were found between the SM-T samples and those of IWS-T and FD-T in the upper portions of the columns. Differences were also found between the VC-B samples and those of IWS-B and FD-B in the lower portions of the columns. Initial analysis of the upper portion of the columns shows the SM-T and VC-T samples had less gene reads than the IWS-T and FD-T reads (6, 9, 22, and 15 reads respectively).

Table A.1 Break down of conversions in nitrification and denitrification as well as the functional genes associated with each conversion

Function	Conversion	Genes
Nitrification	$\text{NH}_4^+ \rightarrow \text{NO}_3^-$	amoA
Denitrification	$\text{NO}_3^- \rightarrow \text{NO}_2^-$	narG
Denitrification	$\text{NO}_2^- \rightarrow \text{NO}_{(\text{g})}$	nirK, nirS, nrfA
Denitrification	$\text{NO}_{(\text{g})} \rightarrow \text{N}_2\text{O}_{(\text{g})}$	norB, norC
Denitrification	$\text{N}_2\text{O}_{(\text{g})} \rightarrow \text{N}_{2(\text{g})}$	nosZ

Table A.2 Average read counts of each gene in the upper and lower portions of each column type. Column names ending in 'T' indicate the upper (TOP) portion of the column while column names ending in 'B' represent the lower (BOTTOM) portion of the column. Standard Deviations are presented in parenthesis.

Gene	VC-T	VC-B	SM-T	SM-B	IWS-T	IWS-B	FD-T	FD-B
amoA	9.33 (5.51)	1.33 (1.16)	6.00 (3.61)	6.33 (2.52)	22.33 (4.51)	10.67 (6.66)	14.67 (7.02)	9 (5.20)
narG	440.00 (336.36)	170.33 (23.97)	203.00 (68.79)	523.33 (512.76)	1179.00 (41.14)	913.33 (556.93)	1134.00 (399.54)	886.33 (146.83)
nirK	86.00 (71.58)	50.00 (8.89)	41.67 (8.33)	106.67 (87.00)	208.33 (10.69)	190.00 (101.42)	218.33 (80.11)	285.33 (131.80)
nirS	134.00 (104)	66.00 (12.77)	79.67 (25.77)	179.67 (174.10)	397.00 (15.13)	323.33 (157.47)	365.67 (132.68)	360.33 (100.48)
nrfA	13.67 (10.02)	4.67 (3.06)	10.33 (2.31)	15.33 (18.77)	33.33 (14.19)	29.67 (16.62)	31.33 (12.66)	29.00 (16.37)
norB	96.00 (68.56)	52.67 (14.05)	47.00 (9.17)	125.00 (106.62)	249.67 (4.73)	218.67 (139.38)	286.67 (121.74)	272.00 (49.52)
norC	34.67 (24.66)	13.00 (9.85)	19.00 (7.94)	50.00 (44.19)	97.67 (22.39)	56.00 (37.97)	83.00 (17.35)	88.67 (38.89)
nosZ	175.67 (150.27)	87.33 (33.08)	70.67 (22.94)	198.33 (177.25)	440.67 (31.02)	382.67 (218.88)	405.33 (12.66)	454.00 (118.58)

Table A.3 Wilcoxon Rank Sum comparisons between sampling locations across all column types and Wilcoxon Signed Rank comparisons between samples in a similar column type ($\alpha = 0.05$).

Wilcoxon Rank Sum Tests Comparing:		P-Value
VCT	SMT	0.2973
VCT	IWST	0.09391
VCT	FDT	0.1615
SMT	IWST	0.01876
SMT	FDT	0.02443
IWST	FDT	0.8633
VCB	SMB	0.1022
VCB	IWSB	0.03998
VCB	FDB	0.02443
SMB	IWSB	0.3401
SMB	FDB	0.2224
IWSB	FDB	0.7962
Wilcoxon Signed Rank Tests Comparing:		P-Value
VCT	VCB	0.003906
SMT	SMB	0.003906
IWST	IWSB	0.003906
FDT	FDB	0.7344

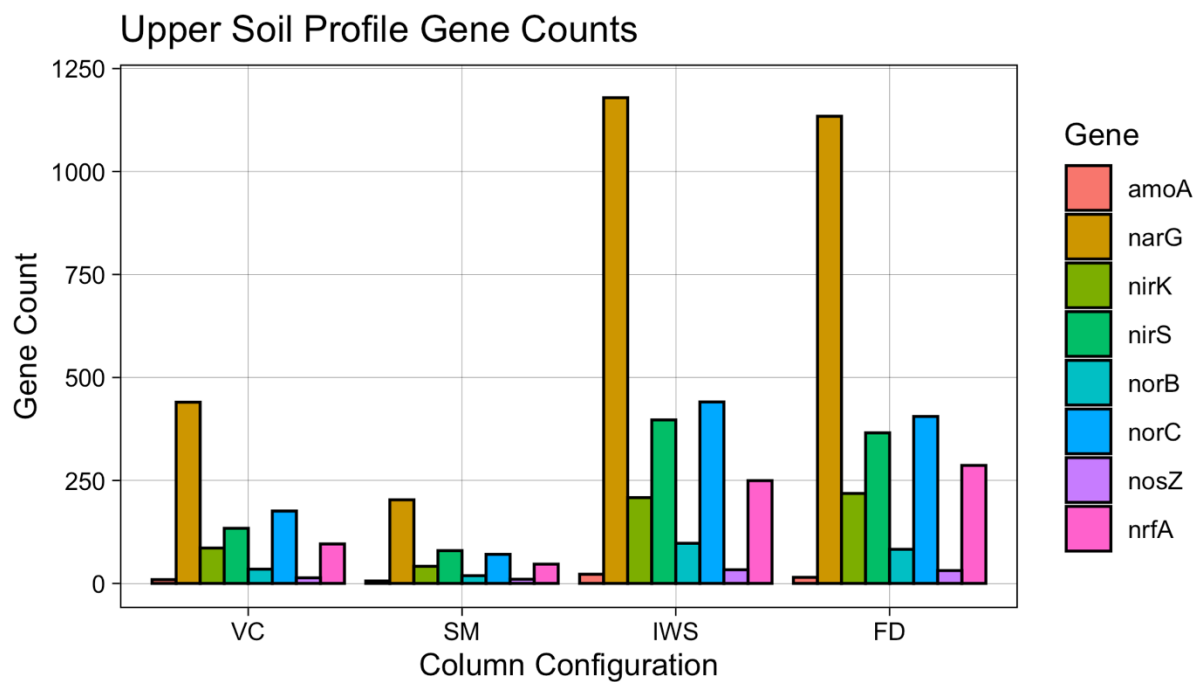


Figure A.2 Gene counts for each gene of interest in the upper portion of the soil media in each column configuration.

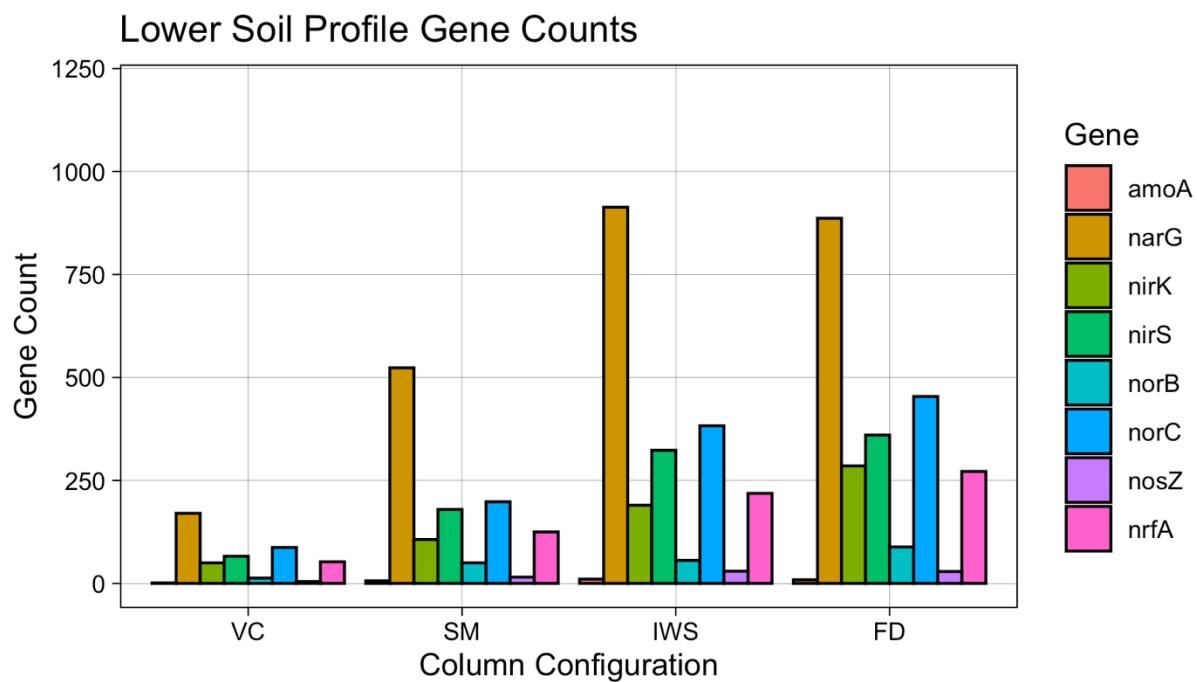


Figure A.3 Gene counts for each gene of interest in the lower portion of the soil media in each column configuration

Compared to the two traditional columns the SMT samples were expected to have the most reads as the SM columns had the most *Nitrospira* abundance and greatest percentage of NH_4^+ -N removal (seen in chapter 3). Further, the nitrification process occurs in aerobic areas which would have been facilitated in the upper portions of the soil profile in each column.

Analysis of the lower portions of the columns shows a similar trend to that of the top, with the VC-B and SM-B samples being lower in read counts than the IWS-B and FD-B samples (1, 6.33, 10, and 9 read counts respectively). While the *Nitrospira* microbe abundance for VC columns was similar to that of IWS and FD columns (Chapter 3), the NH_4^+ -N removal seen in the VC design was similar to that of the SM design column (85% and 83% removal respectively). The traditional columns did not perform as well having 54% and 68% removal of NH_4^+ -N respectively. Thus, the *amoA* gene (associated with *Nitrospira*) would have been expected to be of similar read counts, however, the difference between RTC and traditional columns shows more gene counts in traditional column types. Factors beyond microbial conversion may be responsible for the differences in removal that are seen. It's possible that in the RTC columns, nitrogen fixation or assimilation may be occurring more to contribute to NH_4^+ removal. Plant uptake would increase between the first and second column studies as roots grow and allow more removal through this method. Similar work by Morse et al. (2018) and Payne et al. (2014) showed that plants play a significant role in nutrient uptake however that research was primarily focused on denitrification.

Denitrification ($\text{NO}_3^- \rightarrow \text{NO}_2^-$):

The first step in denitrification focuses on the conversion of nitrates into nitrites through the functional activity of the *narG* gene. The upper portion of the soil profiles shows SM-T (203) and VC-T (440) are lower in read counts than IWS-T and FD-T (1179 and 1134 respectively). In the

lower portions of the profile, SM-B (523) and VC-B (170) are both lower than that of the IWS-B (913) and FD-B (886) columns, which follows the trend seen in the upper portion. The IWS columns had previously shown the most nitrate reduction which is supported in the IWS-T and IWS-B samples having the greatest read counts. The VC and SM columns were expected to express similar read counts to that of the IWS columns based on previous analyses in Chapters 2 and 3. More specifically, the abundance of *Pseudomonas* seen in chapter 3 was greatest in the VC columns which would indicate the greatest gene reads expected in the VC-B samples. The lower portions of the columns should promote more denitrification as the process is completed in anaerobic environments, and soil moisture readings were observed to be lower in this region. Lower read counts of *narG* in the bottom portion when compared to the upper portion of the columns are unexpected and indicate the conversion of nitrate into nitrite, as a result of *narG* activity, occurring more in the upper portion of the soil profile. This could be a result of denitrification occurring in soil microsites which is mostly thought to occur in FD systems as they do not store water and are primarily aerobic in nature.

Denitrification ($\text{NO}_2^- \rightarrow \text{NO}_{(\text{g})}$):

Nitrates generated from the previous step, or that entered in the artificial stormwater, are then converted to nitric oxide through the *nirK*, *nirS*, and *nrfA* genes. For all the genes associated with this conversion, the RTC columns had lower read counts than the traditional columns in both the upper and lower portions of the soil profile. Gas measurements were not taken in this study to compare with the number of gene reads seen in this data, however it can be noted that the abundance of gene counts in the traditional columns would indicate that more conversion of nitrite into nitric oxide would be occurring. It should also be noted that the SM-T had greater read counts

than SM-B and the VC-B had greater read counts than VC-T. This could indicate a greater abundance promoted due to control based on sensor location placement in varying treatment types treatment which would support the research seen in Chapter 3. When comparing these results to the traditional columns, there is still greater gene counts than that of the RTC.

Denitrification ($\text{NO} \rightarrow \text{N}_2\text{O}$):

Conversion of nitric oxide into nitrous oxide is completed by the *norB* and *norC* genes. Partial denitrification can often lead to the production of nitrous oxide, which is a greenhouse gas that has been known to be produced in some bioretention practices. Trends seen in other portions of the denitrification process are the same when considering the production of nitrous oxide. That is, the traditional columns express higher read counts than RTC columns. The higher read counts in the traditional columns indicate greater production of nitrous oxide which, if not fully converted, can contribute to greenhouse gas emissions. One difference in analysis of *norB* shows that SMB and IWSB read counts are more similar which could indicate a similar facilitation of the conversion in SMB configurations. Considerations for the final step in denitrification, which converts nitrous oxide to nitrogen gas, are important as insufficient conversion would indicate that SM columns have a similar greenhouse gas production potential to IWS columns.

Denitrification ($\text{N}_2\text{O} \rightarrow \text{N}_{2(\text{g})}$):

This conversion of nitrous oxide to nitrogen gas is important as this indicates the completion of denitrification and is facilitated by the *nosZ* gene. Lower read counts were observed in RTC columns compared to traditional columns in both upper and lower portions of the soil profile. Between the two RTC columns, the VCT had more read counts of *nosZ* than SMT did, and the

opposite was true for the samples in the lower soil profile. Between the two RTC columns, it is difficult to distinguish which column would better complete the denitrification process. Future research should include gas measurements to fully characterize each system. With the read counts being greater in the traditional columns it can be assumed that those columns better facilitate complete denitrification.

Conclusions:

Following a six-week column study, soil cores were taken from the upper and lower portion of each column configuration. Gene read counts were determined for genes associated with common conversions in the nitrification and denitrification cycles. A similar trend for each gene was observed whereby RTC columns had fewer read counts than traditional columns. This contradicted microbial abundance data found in Chapter 3. Read counts analyzed in this research are indicative of activity however more specific testing involving RNA analysis would allow a more detailed view of transcription occurring. Further analysis of gene read counts could also be done to analyze differences over time. That is, if changes in count happen between storms based on storm size or dry periods.

Specifically, regarding completing denitrification and preventing the emittance of $\text{N}_2\text{O}_{(g)}$, *norB*, *norC*, and *nosZ* gene counts in the traditional columns are higher than those in the RTC columns, which indicates more $\text{N}_2\text{O}_{(g)}$ production and conversion of $\text{N}_2\text{O}_{(g)}$ into $\text{N}_{(g)}$. Lower counts in the RTC columns should not suggest that denitrification is incomplete, however it would show that there is less denitrification occurring. Nutrient removal in these columns would then be a result of other physical or biological processes such as adsorption onto soil media or plant uptake. Thus, while this research investigated the microbial contributions to nitrogen removal, other removal

mechanisms should also be investigated. The impact of RTC on plant health was not specifically studied in this research and so analysis on differences in plant root growth and nutrient uptake should be completed in future research.

References:

- Barros, N., I. Gomezorellana, S. Feijoo, and R. Balsa. 1995. 'THE EFFECT OF SOIL-MOISTURE ON SOIL MICROBIAL ACTIVITY STUDIED BY MICROCALORIMETRY', *Thermochimica Acta*, 249: 161-68.
- Blecken, G. T., Y. Zinger, A. Deletic, T. D. Fletcher, A. Hedstrom, and M. Viklander. 2010. 'Laboratory study on stormwater biofiltration Nutrient and sediment removal in cold temperatures', *Journal of Hydrology*, 394: 507-14.
- Bratieres, K., T. D. Fletcher, A. Deletic, and Y. Zinger. 2008. 'Nutrient and sediment removal by stormwater biofilters: A large-scale design optimisation study', *Water Research*, 42: 3930-40.
- Brown, R. A., and W. F. Hunt. 2011. 'Underdrain Configuration to Enhance Bioretention Exfiltration to Reduce Pollutant Loads', *Journal of Environmental Engineering-Asce*, 137: 1082-91.
- Cadenasso, M. L., S. T. A. Pickett, P. M. Groffman, L. E. Band, G. S. Brush, M. E. Galvin, J. M. Grove, G. Hagar, V. Marshall, B. P. McGrath, J. P. M. Oneil-Dunne, W. P. Stack, and A. R. Troy. 2008. 'Exchanges across land-water-scape boundaries in urban systems - Strategies for reducing nitrate pollution', *Year in Ecology and Conservation Biology 2008*, 1134: 213-32.
- Chen, T., Y. P. Liu, B. Zhang, and L. H. Sun. 2019. 'Plant rhizosphere, soil microenvironment, and functional genes in the nitrogen removal process of bioretention', *Environmental Science-Processes & Impacts*, 21: 2070-79.
- Chen, X. L., E. Peltier, B. S. M. Sturm, and C. B. Young. 2013. 'Nitrogen removal and nitrifying and denitrifying bacteria quantification in a stormwater bioretention system', *Water Research*, 47: 1691-700.

- Collins, K. A., T. J. Lawrence, E. K. Stander, R. J. Jontos, S. S. Kaushal, T. A. Newcomer, N. B. Grimm, and M. L. C. Ekberg. 2010. 'Opportunities and challenges for managing nitrogen in urban stormwater: A review and synthesis', *Ecological Engineering*, 36: 1507-19.
- Davis, A. P., M. Shokouhian, H. Sharma, and C. Minami. 2006. 'Water quality improvement through bioretention media: Nitrogen and phosphorus removal', *Water Environment Research*, 78: 284-93.
- Henderson, C., M. Greenway, and I. Phillips. 2007. 'Removal of dissolved nitrogen, phosphorus and carbon from stormwater by biofiltration mesocosms', *Water Science and Technology*, 55: 183-91.
- Hsieh, C. H., A. P. Davis, and B. A. Needelman. 2007. 'Nitrogen removal from urban stormwater runoff through layered bioretention columns', *Water Environment Research*, 79: 2404-11.
- Hunt, W. F., J. T. Smith, S. J. Jadlocki, J. M. Hathaway, and P. R. Eubanks. 2008. 'Pollutant removal and peak flow mitigation by a bioretention cell in urban Charlotte, NC', *Journal of Environmental Engineering-Asce*, 134: 403-08.
- Igielski, S., B. V. Kjellerup, and A. P. Davis. 2019. 'Understanding urban stormwater denitrification in bioretention internal water storage zones', *Water Environment Research*, 91: 32-44.
- Kerkez, B., C. Gruden, M. Lewis, L. Montestruque, M. Quigley, B. Wong, A. Bedig, R. Kertesz, T. Braun, O. Cadwalader, A. Poresky, and C. Pak. 2016. 'Smarter Stormwater Systems', *Environmental Science & Technology*, 50: 7267-73.
- Manka, B. N., J. M. Hathaway, R. A. Tirpak, Q. He, and W. F. Hunt. 2016. 'Driving forces of effluent nutrient variability in field scale bioretention', *Ecological Engineering*, 94: 622-28.
- Middleton, J. R., and M. E. Barrett. 2008. 'Water quality performance of a batch-type stormwater detention basin', *Water Environment Research*, 80: 172-78.

- Morse, N., E. Payne, R. Henry, B. Hatt, G. Chandrasena, J. Shapleigh, P. Cook, S. Coutts, J. Hathaway, M. T. Walter, and D. McCarthy. 2018. 'Plant-Microbe Interactions Drive Denitrification Rates, Dissolved Nitrogen Removal, and the Abundance of Denitrification Genes in Stormwater Control Measures', *Environmental Science & Technology*, 52: 9320-29.
- Mullapudi, A., M. Bartos, B. Wong, and B. Kerkez. 2018. 'Shaping Streamflow Using a Real-Time Stormwater Control Network', *Sensors*, 18: 11.
- Passeport, E., W. F. Hunt, D. E. Line, R. A. Smith, and R. A. Brown. 2009. 'Field Study of the Ability of Two Grassed Bioretention Cells to Reduce Storm-Water Runoff Pollution', *Journal of Irrigation and Drainage Engineering*, 135: 505-10.
- Payne, E. G. I., T. D. Fletcher, D. G. Russell, M. R. Grace, T. R. Cavagnaro, V. Evrard, A. Deletic, B. E. Hatt, and P. L. M. Cook. 2014. 'Temporary Storage or Permanent Removal? The Division of Nitrogen between Biotic Assimilation and Denitrification in Stormwater Biofiltration Systems', *Plos One*, 9.
- Persaud, P. P., A. A. Akin, B. Kerkez, D. T. McCarthy, and J. M. Hathaway. 2019. 'Real time control schemes for improving water quality from bioretention cells', *Blue-Green Systems*, 1: 55-71.
- Read, J., T. D. Fletcher, T. Wevill, and A. Deletic. 2010. 'Plant Traits that Enhance Pollutant Removal from Stormwater in Biofiltration Systems', *International Journal of Phytoremediation*, 12: 34-53.
- Read, J., T. Wevill, T. Fletcher, and A. Deletic. 2008. 'Variation among plant species on pollutant removal from stormwater non biofiltration systems', *Water Research*, 42: 893-902.
- Roman, D., A. Braga, N. Shetty, and P. Culligan. 2017. 'Design and Modeling of an Adaptively Controlled Rainwater Harvesting System', *Water*, 9: 14.
- TSM, Tennessee Stormwater Management. 2015. "Management and Design Guidance Manual." In *Tennessee Stormwater Managment*.

Waller, L. J., G. K. Evanylo, L. A. H. Krometis, M. S. Strickland, T. Wynn-Thompson, and B. D. Badgley.

2018. 'Engineered and Environmental Controls of Microbial Denitrification in Established Bioretention Cells', *Environmental Science & Technology*, 52: 5358-66.

Vita

Originally born in Kingston, Jamaica, Padmini grew up in South Florida. After attending South Plantation High School where she participated in an Environmental Magnet, she attended the University of Florida and further pursued her interests in environmental work. She received a Bachelor of Science degree in Environmental Engineering May 2017. Prior to graduating with her undergraduate degree, Padmini made the decision to pursue a Doctor of Philosophy degree in Civil and Environmental Engineering. She chose to attend the University of Tennessee, Knoxville and focus her graduate work in water resources. Her research interests include stormwater treatment through the use of green infrastructure and the effects of using real-time control systems. After graduation, she will begin working in consulting engineering. She is grateful for all of the support she has received from her family, friends, and colleagues as she begins the next stage in her new career.