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# EVALUATING LOW IMPACT DEVELOPMENT BEST MANAGEMENT PRACTICES AS AN ALTERNATIVE TO TRADITIONAL URBAN STORMWATER MANAGEMENT

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EVALUATING LOW IMPACT DEVELOPMENT BEST MANAGEMENT PRACTICES AS AN ALTERNATIVE TO TRADITIONAL URBAN STORMWATER MANAGEMENT

A THESIS APPROVED FOR THE
SCHOOL OF CIVIL ENGINEERING AND ENVIRONMENTAL SCIENCE

BY

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For my father Mark.

I think about you every day.

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# 1.0 Introduction

## 1.1 Urbanization and Impervious Surfaces

Urbanization can be defined as a shift in population from a rural setting to urban areas and the associated adjustments society makes to accommodate the increased population densities (Annez and Buckley 2009). Urbanization plays a role in a variety of negative environmental impacts, including global climate change (e.g., city centers act as sources for carbon emissions), adverse alterations in natural biogeochemical cycles (e.g., nitrogen cycle), and substantial changes in water resource availability and quality (e.g., through construction of impermeable surfaces) (Grimm et al. 2008; USDA 2011). The most predominant alteration to the nitrogen cycle is from anthropogenic deposits of atmospheric nitrogen (IPCC 2007). The main developmental strategy that occurs when urban areas experience an increase in population is the conversion of permeable vegetated landscapes to impervious surfaces, including paved roads, driveways, parking lots, rooftops, and sidewalks (Carlson and Arthur 1998; Kaushal and Belt 2012). From 1945 to 2010 the amount of impervious surface area quadrupled in the United States (USDA 2011; USCB 2012; Martin-Mikle et al. 2015). However, this vast increase of urbanized area and associated impervious surfaces only comprises a small fraction of the total land area of the United States (~3 percent) (USCB 2012). Even with this small fraction of urban area in the United States, there is substantial evidence that the resulting impervious surface coverage affects both water quantity and water quality (EPA 1997). According to the Environmental Protection Agency (EPA 1997), pre- and post-development hydrographs demonstrate the water quantity component of these changes. Figure 1 represents the impact of urbanization on a generic basin's hydrologic regime. Post-development base flow is smaller than pre-development base flow resulting in habitat loss, higher average stream temperatures, and lower dissolved oxygen (DO) concentrations. Peak discharge is also impacted by impervious surfaces. In post-development, the peak discharge represents a short time period with elevated discharge while in pre-development the peak discharge is spread out over time, followed by a gradual regression back to base flow conditions. Besides altering the hydrologic flow regime, urbanization impacts other elements of the natural drainage system.

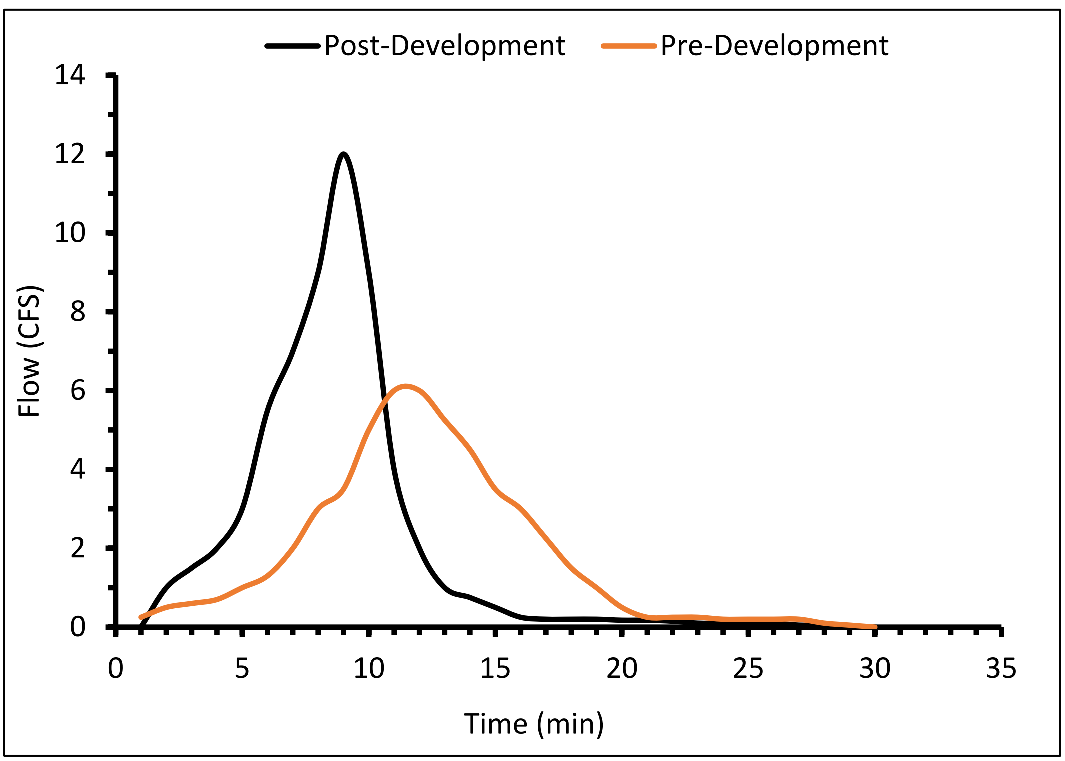


*Figure 1. Impacts of Urbanization on Stream Flow Modified from EPA (1997)*

Alterations to a basin's characteristics (e.g., permeability) results in changes in the hydrology of a basin potentially causing the flood wave to be transmitted downstream faster with less attenuation from the stream itself (Booth and Jackson 1997). According to Mollison and Holmgren (1978), the problem is the solution, "the problem is not that urban areas produce excessive quantities of stormwater. On the contrary stormwater is a resource. The problem redefined is that urban areas have a deficit of beneficial uses for the runoff they shed". With the continuous increases in impervious surface areas and resulting urban stormwater runoff, conventional methods for stormwater management (e.g., curb and gutter collection systems, drains and storm sewer conveyance and retention/detention ponds) will not sufficiently cope with larger amounts of urban stormwater volume (White and Greer 2004). In fact, the amount of runoff increases proportionally to the amount of impervious surface area within a basin, which in turn increases the peak discharge and flood magnitude (White and Greer 2004). The increase in developed urban land can be further explained by Table 1. If the watershed is only 10 percent impervious surface by area, there will be a quantifiable degradation in water quality. If the imperviousness increases to 25 percent impervious surface by area, there will be shoreline and stream channel erosion and, in turn, inadequate fish and insect habitat, and so forth.

*Table 1. Impacts from Increased Impervious Surface Area Modified from Kloss and Calarusse (2006)*

| Watershed Impervious Level | Effect |
|---|---|
| 10 percent | Degraded water quality |
| 25 percent | Inadequate fish and insect habitat along with shoreline and stream channel erosion |
| 35-50 percent | Runoff equals 30 percent of rainfall volume |
| >75 percent | Runoff equals 55 percent of rainfall volume |

Overall, impacts of urbanization and resulting increases in impervious surface area leads to a more complex urban stormwater pollutant load, decreased pollutant removal and infiltration during overland flow, and increased peak discharge rates which can expedite stream erosion (Davis 2005; Selbig and Bannerman 2008). Increases in impervious surface area as a result of urbanization results in urban systems that are highly responsive to stormwater contributions (Waldron et al. 2010). Unabated stormwater flows can substantially degrade both the geomorphology and water quality of receiving water bodies (EPA 1997). More specifically, erosion of stream banks due to increased volumes and velocities of stormwater flow leads to considerable degradation and aggradation of stream channels, along with deterioration of downstream water quality, which leads to the destruction of instream and riparian habitats, increased sedimentation rates, and nutrient loads (EPA 1997). Table 2 provides a summarized list of the resulting impacts from increased urbanization.

*Table 2. Urban Stormwater Impacts on Water Quality Modified from EPA (1997)*

| Increased Imperviousness Leads to: | Resulting Impacts | | | | |
|---|---|---|---|---|---|
| | Flooding | Habitat loss | Erosion | Channel widening | Streambed alterations |
| Increased volume | X | X | X | X | X |
| Increased peak flow | X | X | X | X | X |
| Increased stream temperature | | X | | | |
| Decreased base flow | | X | | | |
| Changes in sediment loads | X | X | X | X | X |

## 1.2 Stormwater Runoff

### 1.2.1 Traditional stormwater management and associated problems

The need for stormwater management and regulation is realized as growing urban areas produce greater quantities of stormwater of poor quality (NRC 2008; Liebman et al. 2011). The National Pollutant Discharge Elimination System (NPDES) program under the Clean Water Act (CWA) is the primary federal regulatory tool to evaluate the quality of the nation's water bodies. Initially this program served to monitor and regulate point source discharges from industrial and municipal discharges (NRC 2008; EPA 2015a). Point source pollution is relatively easy to regulate due to its discharge origins; in that the pollution source can be identified (e.g., pipe outfalls, channels, or concentrated animal feeding operations) (NRC 2008; EPA 2016a). Non-point source pollution is much more difficult to quantify, as it results from precipitation, atmospheric deposition, surface runoff, drainage, or hydrologic modification (EPA 2016a). Non-point source pollution stems from many diffuse sources, such as agricultural fields, urban and suburban residential areas, and abandoned mining operations (EPA 2016a). Due to developments through the CWA, point source pollution has become highly regulated, but non-point source pollution has degraded over 10,000 miles of rivers and streams and approximately 500,000 acres of lakes and reservoirs in Oklahoma alone, and therefore will be the focus of further discussion (EPA 2016b). To address the role of urban stormwater and its contributions to the degradation of the nation's water bodies, in 1987 Congress passed Section 402(p) of the CWA, which handed stormwater control to the NPDES program (NRC 2008). In 1990 and 1999, the EPA developed Phase I and Phase II Stormwater Rules, respectively. Phase I Stormwater Rules require NPDES permits for operators of municipal separate storm sewer systems (MS4s) serving populations over 100,000 people and for discharges associated with industry and construction sites which are five acres or larger. Phase II Stormwater Rules expanded the requirements to smaller MS4s and construction sites of one to five acres in size (NRC 2008).

With the obvious need for urban stormwater management, traditional urban and suburban stormwater management systems typically consist of curb and gutter collection systems, drains and storm sewer conveyances, and detention and retention ponds (Booth and Jackson 1997; Kloss and Calarusse 2006). The goal of traditional management is to divert stormwater runoff from urban areas as quickly as possible utilizing networks of storm drains, detention/retention ponds, and various surface water bodies to minimize the risk of local flooding (Waldron et al. 2010). The caveat to this method is that even though local flooding risks are minimized, unintended impacts to receiving waters are prevalent (e.g., alteration of stream geomorphology, increases in peak discharge rates, decreases in groundwater infiltration, introduction of more complex pollutant loads, degradation of water quality, and loss of habitat and biodiversity) (EPA 1997; Selbig and Bannerman 2008; Waldron et al. 2010). Furthermore, the designs of traditional stormwater management systems focus solely on water quantity and do not attempt to address stormwater quality.

### 1.2.2 Increased Runoff Degrades Environmental Quality

The above alterations to a basin's natural hydrologic regime also introduce increased concentrations and loads of various urban stormwater pollutants (Table 3) decreasing the quality of receiving ecosystems (Kloss and Calarusse 2006; Selbig and Bannerman 2008; Waldron et al 2010). Each of the resulting impacts (Table 2) will be discussed in detail regarding negative effects on aquatic ecosystems.

Increased imperviousness results in increases in total runoff volume, prolonged peak discharge rates, and peak discharge volume, all of which contribute to flooding in basins with high amounts of impervious surface coverage (Booth and Jackson 1997; EPA 1997). As impervious surface coverage increases, the volume and velocity of urban stormwater also increases. With the larger volume of stormwater and increased efficiency of conveying water off the surface through pipes, gutters, and man-made or straightened channels, the severity of flooding increases (Leopold 1968; Arnold and Gibbons 1996).

*Table 3. Common Urban Stormwater Pollutants and Associated Sources Modified from Kloss and Calarusse (2006)*

| Pollutant | Sources |
|---|---|
| Bacteria | Pet waste and wastewater |
| Metals | Automobiles and roof shingles |
| Nutrients | Lawns and atmospheric deposition |
| Oil and grease | Automobiles |
| Oxygen-depleting substances | Organic matter |
| Pesticides | Lawns and gardens |
| Sediment | Construction sites |
| Toxic chemicals | Automobiles and industrial facilities |
| Trash and debris | Multiple sources |

The degradation of river banks and resulting aggradation of streams from increased runoff and sediment volumes combined with the highly responsive nature of peak discharges cause several negative ecological impacts to occur (Arnold and Gibbons 1996). Widening of channels will occur and result in the loss of riparian wetlands that protect riverine systems. Loss of riparian wetlands can be devastating to a surface water ecosystem because these systems are responsible for protection from bank erosion, uptake and recycling of nutrients, and provisioning of habitat (Cowardin et al. 1979; Schueler 1992).

### 1.2.3 Urban Stormwater Pollutant Load Impacting Ecosystems

The final aspect of urban stormwater pollution that will be discussed is one that stems from the urban land uses common to watersheds with high amounts of impervious surface area. Complex pollutant loads will be transported directly to waterways through traditional stormwater management techniques common to such watersheds creating non-point pollution sources (Arnold and Gibbons 1996; Booth and Jackson 1997). Table 3 shows common pollutants and their associated sources (Kloss and Calarusse 2006). In short, bacteria can cause contamination of a habitat, while trace metals, toxic organics, and pH can all cause alterations to the species distribution. Increases in sediment volume introduced to an ecosystem can decrease available spawning areas for fish and other aquatic organisms (Ryan 1991). Finally, excess nutrients from lawns, agricultural lands, and atmospheric deposition can be the source of cultural eutrophication (Kloss and Calarusse 2006). Eutrophication occurs when the limiting nutrient (e.g., nitrogen or phosphorus) in a water body is present in quantities in excess of need, allowing for photosynthetic organisms (e.g., algae) in a water body to flourish (ODEQ 2013). Typically, in freshwater systems the most influential compound is phosphorus, even more importantly is the ratio of nitrogen to phosphorus (e.g., the Redfield ratio). The flourishing is known as an algal bloom, which could potentially be toxic for biota and humans alike depending on the species of algae present in the water body (EPA 2015a). Sometimes when these compounds are out of balance, cyanobacteria may begin to fix nitrogen and thus begin to flourish. When a bloom dies, the decomposition of the algae consumes DO, thus resulting in dramatic diurnal fluctuations of DO concentrations. Low levels of DO inhibit the survival of all aquatic species (Arnold and Gibbons 1996; Anderson et al. 2002; EPA 2015a).

## 1.3 Low Impact Development Best Management Practices

### 1.3.1 Background and History of LID BMPs

LID BMPs or Green Infrastructure (GI) are relatively new tools focusing on stormwater management created in the early 1990's (EPA 2000; Coffman 2000; Hager 2003). The theory behind these technologies is to capture and treat stormwater as close to where it falls as is possible, with the intention of creating an environment which mimics pre-development hydrologic regimes (EPA 2000; Coffman 2000; Hager 2003; Kloss and Calarusse 2006; Bedan and Clausen 2009; Waldron et al. 2010). This new urban stormwater pollution control methodology was developed based on the knowledge of natural systems in their ability to decrease urban stormwater pollutant loads (e.g., through filtration, retention, and transformation), provide a domain for various biogeochemical processes (e.g., nitrogen fixation), and increase ground water recharge (EPA 2000; Coffman 2000; Kloss and Calarusse 2006). Essentially LID BMPs attempt to model natural processes and simulate pre-development hydrology through, infiltration, retention, storage, filtration, transformation, evaporation, and detention of urban stormwater runoff (EPA 2000; Coffman 2000; Hager 2003).

Referring to Hager (2003), the five main aspects of LID integrated management include: 1) conservation and minimization, 2) conveyance, 3) storage, 4) infiltration, and 5) landscaping. Conservation and minimization can be achieved by narrowing residential streets, decreasing the impervious sidewalk area, addition of or replacement with porous pavement, and creation of concave medians. Stormwater conveyance can be accomplished by diverting water into grassed swales and disconnecting impervious areas to redirect runoff to more vegetated areas. Stormwater storage can be attained through the use of rain barrels, green roofs, and curb or subsurface storage. Infiltration of stormwater is necessary to recharge groundwater supplies (Hager 2003). Infiltration can be completed through the use of trenches and basins that allow for water to sit for a short period of time and infiltrate into the subsurface (EPA 2000). Landscaping is the final integrated management aspect. Rain gardens, slope reduction, and native ground cover are all examples of how landscaping can be used as an urban stormwater management tool (Hager 2003).

LID BMPs require a certain amount of operation and maintenance. Many times the maintenance responsibilities fall upon private land owners, who more often than not are under-educated in the technology behind the LID BMPs or the benefits that proper implementation and operation can provide to the community. Aside from concerns about who is responsible for the operation and maintenance of the LID BMPs, the public has other potential concerns when comparing LID BMPs to conventional stormwater management (EPA 2000). The first of those concerns is related to the cost of implementing LID BMPs (EPA 2009). EPA (2009) addressed this concern through a study that compared the actual cost of LID development to the estimated cost of the project using conventional stormwater management. The results of the study showed that implementing LID technology can decrease the cost of development by requiring less grading, landscaping, and paving, essentially lowering all infrastructure capital costs (EPA 2009). The second concern of the public is in regard to seasonal variation in performance. The University of New Hampshire's

(UNH) Stormwater Center (2007) investigated this issue thoroughly, finding that, “research data tells us that it’s possible to design and install systems that do an excellent job of treating pollutants in stormwater, dampening the peak flows of runoff, and reducing the volume of stormwater through infiltration”. The third issue is in regard to groundwater pollution via infiltration from LID BMPs. To ensure that infiltration from LID BMPs does not impact groundwater supplies, the design and site location must be selected to ensure that they are the best option for the local setting. Furthermore, the UNH Stormwater Center (2007) has data that suggest “infiltration practices remove pollutants found in urban stormwater below levels of concern for groundwater protection”.

Here one must emphasize the importance of evaluating any site prior to LID BMP implementation. Locations where shallow groundwater aquifers are used as a drinking water source may not be appropriate to allow for infiltration practices, so an alternative LID BMP should be selected. Additionally, infiltration of stormwater from certain land uses (e.g., salt piles, gasoline service stations, and facilities handling hazardous waste) should be avoided. These types of land uses can produce stormwater runoff containing contaminants that would not be removed via infiltration, which in turn could then pollute the groundwater drinking supply.

### 1.3.2 Common LID BMP Technologies

EPA (2015B) provided some common LID BMPs technologies (Table 4). This section focuses on describing functionality and treatment mechanisms in various LID BMP technologies. Diversion of downspouts so that stormwater is not discharged onto impervious surfaces but into grassed swales or rain barrels is beneficial for several reasons. Rain barrels are low-cost, efficient, and easily maintained retention systems that can vary in size depending on the site application, but one 50 gallon rain barrel can retain 0.24 inches of runoff for a roof of approximately 600 square feet (Coffman 2000). Grassed swales are small drainages and grassed channels with the goal to allow infiltration and conveyance of stormwater away from roads and right-of-ways (EPA 2000; Coffman 2000). According to EPA (2015B) this method of stormwater diversion can be beneficial to communities with combined sewer systems. Decreasing total stormwater volume input into sewer systems decreases the chances for overflow to occur in these combined sewer systems. Furthermore, the diversion of stormwater off of roofs and away from stormwater sewers and into rain barrels allows for rain water to be harvested. This harvested rain water can be used for irrigation, pressure washing, or in buildings to flush toilets (Dhalla and Zimmer 2010). In addition retaining the water and nutrients on site, rain barrels have the ability to decrease water utility costs and improve downstream water quality (Coffman 2000).

*Table 4. Examples of Various LID BMP Technologies Modified from EPA (2015B)*

| Practice | Description | Benefit |
|---|---|---|
| Downspout Disconnection | Divert rooftop drains to direct rain water into either vegetated area or rain barrels | Enhanced storage and potential for infiltration. Particularly beneficial to communities with a combined sewer systems |
| Rain Gardens and Bioswales | Shallow vegetated area used to collect stormwater and sediment during storm events | Retention of the storm sediment load. Aesthetic value that is exceedingly versatile |
| Green Roofs | Building rooftops planted with vegetation that are capable of absorbing and transpiring stormwater | Unique in that these can be used in areas where land area is valued. Also insulates buildings, reducing energy costs |
| Green Alleys and Streets | Implementation of permeable street LID technologies | Store stormwater for reuse and improves the citizen experience through shading and flood control |
| Land Conservation | Preserving natural areas within or near cities | Provides recreational opportunities and habitat |
| Rain water Harvesting | Collection and storage of stormwater for future use | Decreases runoff volume and provides a source for residential irrigation |
| Planter Boxes | Rain gardens with vertical walls makes these ideal for urban areas where land is valued | Allow for infiltration of stormwater from rooftops, sidewalks, parking lots, and streets |
| Permeable Pavements | Porous concrete that allows for infiltration of stormwater | Cost effective and efficient at removing stormwater from the surface |
| Green Parking | Implementation of various BMPs in parking lots | Collect and absorb stormwater, provide shade and reduce heat emitted by pavement |
| Urban Tree Canopy | Establishing an urban tree canopy | Decreases carbon footprint, trees uptake stormwater, provides structure for soil to mitigate erosion and provides habitat and shade |

Rain gardens or bioswales are shallow depressions in the landscape that have the ability to retain urban stormwater runoff from rooftops, sidewalks, and streets (Coffman 2000; Dhalla and Zimmer 2010; EPA 2015b). Retention of stormwater, removal and transformation of nutrients, and filtration of sediments are completed using various plant species and engineered soil substrates (Coffman 2000; EPA 2015b). These systems are one of the most complex type of LID BMPs, therefore many design considerations must be evaluated. Depending on local soil conditions and spatial constraints, rain gardens may be designed with an underdrain allowing for only partial infiltration to occur, or with an impermeable clay layer with an underdrain allowing only for filtration to occur (Dhalla and Zimmer 2010). Another important design consideration is to determine the proper engineered substrate to be placed into the system. The proper substrate will ensure that infiltration will actually occur (and clogging will not), that desired biogeochemical processes will take place, and that the substrate will support the planted vegetation (EPA 2000; Coffman 2000). A final design consideration that must be evaluated is the species of vegetation that will be planted. Native plant species should be used based on site-specific conditions and ecological factors. The species should be selected based upon their moisture regime, morphology, susceptibility to pests, and tolerance to common urban stormwater pollutants (EPA 2000; Dhalla and Zimmer 2010; Coffman 2014). The Maryland Department of Environmental Resources (2007) suggests that a minimum of three species of trees and shrubs be planted to ensure high diversity, minimize seasonal differences in evapotranspiration, and continue nutrient and pollutant uptake throughout the year (EPA 2000).

Impervious surfaces increase stormwater volumes and concentrations and loads of urban stormwater pollutants (Booth and Jackson 1997; EPA 1997). Permeable pavements, however, are paved surfaces that allow for, infiltration, filtration, storage, and a decrease or elimination of surface stormwater flows compared to traditional impervious pavement (Dhalla and Zimmer 2010; EPA 2015b). These systems are particularly useful when land value is high and spatial availability is low (Dhalla and Zimmer 2010; EPA 2015b). Several sources suggest for the most successful implementation these systems should be constructed in low traffic areas, such as parking lots or sidewalks and overlying highly porous soils to allow for maximum infiltration (Booth and Levitt 1999; EPA 2000; Kloss and Calarusse 2006; Dhalla and Zimmer 2010; EPA 2015b).

### 1.3.3 LID BMP Effectiveness

LID BMP or GI technology serves several purposes; the key common characteristics to these technologies are the ability to remove urban stormwater pollutants and mimic pre-development hydrologic regimes (EPA 2000; Coffman 2000; Hager 2003; Kloss and Calarusse 2006; Bedan and Clausen 2009; Waldron et al. 2010). A study in Connecticut known as the Jordan Cove Urban Watershed Project involved the comparison of control, traditional, and LID watersheds. Bedan and Clausen (2009) monitored the watersheds pre- and post-development, while the control watershed was developed several years prior to the study. The results of the study showed that total stormwater runoff from the traditional watershed increased dramatically during post-development when compared to pre-development. The increase in stormwater volumetric discharge from the watershed also increased mass exports of nitrate-nitrite ($NO_3$-$NO_2$), ammonia ($NH_3$), Total Kjeldahl Nitrogen (TKN), total phosphorus (TP), total suspended solids (TSS), Cu, and Zn. In comparison, the LID watershed experienced decreases in both total stormwater discharge and peak flow when compared to pre-development conditions. In turn, mass exports of TKN, $NH_3$, Pb, and Zn significantly decreased during the post-development conditions. A confounding note must be made in that TSS and TP concentrations and exports increased during the post-development conditions for the LID watershed runoff. This phenomenon can be attributed to the substrate that was used in the rain gardens or perhaps from fertilizer transport off residential lawns (e.g., swales) (Bedan and Clausen 2009). However this concept is not fully understood and requires further research in order to identify the problem.

The results of another study completed in the Ipswich River Basin in Massachusetts by Waldron et al. (2010), "indicate that even relatively small reductions in effective impervious area, in an area underlain by highly permeable, sandy soils, such as the LID retrofit neighborhood, can produce measureable reductions in stormwater runoff for small storms". Another study completed in Orange County, Florida by Nunn (2014), showed a 97 percent decrease in the total phosphorus load in a high density residential neighborhood through utilization of various LID BMPs (e.g., rain gardens, bioretention swales, tree filter boxes, planter boxes, and curb cuts).

Table 5 and Table 6 display percent removal efficiencies for LID BMPs addressing common urban stormwater pollutants modified from Dhalla and Zimmer (2010), EPA (2000), Coffman (2000), and Martin-Mikle et al. (2015), respectively. Percent removal efficiencies presented in Table 5 for Dhalla and Zimmer (2010) are average values from Dietz and Clausen (2005), Hunt et al. (2006a), Hunt et al. (2006b), Davis (2007a), Davis (2007b), Muthanna et al. (2007), Hunt et al. (2008), Roseen et al. (2009a), Roseen et al. (2009b), and Diblasi et al. (2009). The data presented from EPA (2000) were compiled from a laboratory and field study completed at the Beltway Plaza Mall Parking Lot, Greenbelt, Maryland. Furthermore, the percent removal efficiencies presented in Table 6 for Coffman (2000) are reported values that were averaged from CRC (1996), Davis et al. (1997), MWCG (1987), Urbonas and Stahre (1993), Yousef et al. (1985), Yu et al. (1992), and Yu et al. (1993). Finally the percent removal efficiencies presented by Martin-Mikle et al (2015) were calculated based on the International Stormwater Best Management Practice Database.

*Table 5. Percent Removal Efficiencies for Bioretention Cells Modified from Dhalla and Zimmer (2010) and EPA (2000*

| | Bioretention Percent Removals | |
|---|---|---|
| | Dhalla and Zimmer (2010) | EPA (2000) |
| Pb | 76.6 | 94 |
| Cu | 80.4 | 96 |
| Zn | 81.3 | 97.5 |
| TSS | 39.3 | --- |
| TP | -10.3 | 54 |
| TKN | 28.8 | 67.5 |
| $NH_4$ | --- | 71.5 |
| $NO_3$ | --- | 18.5 |
| TN | --- | 67.5 |

*Table 6. Percent Removal Efficiencies for Various LID BMPs Modified from Coffman (2000) and Martin-Mikle et al. (2015); BR = Bioretention, VS = Vegetated Swale, BS = Buffer Strip, IT = Infiltration Trench, PP = Porous Pavement, DP = Detention Pond, RP = Retention Pond*

| Various LID BMPs Percent Removal Efficiencies | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | | BR | VS | BS | IT | PP | DP | RP |
| Coffman (2000) | | | | | | | | |
| | TSS | --- | 47.5 | 60 | 90 | --- | --- | --- |
| | TN | 43 | 7.5 | 30 | 50 | --- | --- | --- |
| | TP | 81 | 17.5 | 30 | 50 | --- | --- | --- |
| Martin-Mikle et al. (2015) | | | | | | | | |
| | TSS | 79 | 30.5 | --- | --- | 77 | 62.5 | 80 |
| | TN | 29 | 16 | --- | --- | 21 | 0 | 27.5 |
| | TP | 21 | 0 | --- | --- | 37.5 | 20.5 | 54.5 |

## 1.4 Ecosystem Services

### 1.4.1 Background of Ecosystem Services

Ecosystems are defined by a complex set of interactions between plants, animals, microorganisms, humans, and the non-living environment (NRC 2004; MA 2005; Wan et al. 2014). These biotic and abiotic relationships are interconnected through various material cycles and energy flows (MA 2005). Ecosystems provide a suite of goods and services to people and these products are commonly known as Ecosystem Services (ES) (NRC 2004). Evaluation of ES is necessary in order to document how changes in ES impact human well-being, how changes to ecosystems may affect future generations, and what modifications can be made at various scales to improve ecosystem management and drive sustainability (NRC 2004; MA 2005).

This idea of ES has provided insight into how unrecognized goods and services provided by ecosystems benefit human societies. It is now understood that the natural environment and the systems that comprise it are a form of natural capital (NRC 2004). The recognition of the benefits provided by ecosystems is a relatively new concept (NRC 2004; MA 2005). To place a value on ES, the economic value of the goods and services provided must be known. This economic valuation of ES comes with a series of issues. One of the difficulties of placing value on ecosystem services is providing a distinct description and assessment of the links between the dynamics of natural systems, the goods and services provided, and the associated monetary values (NRC 2004). Over 100 years ago the idea of ecosystem services was realized by President Theodore Roosevelt when in 1907 he said, “The nation behaves well if it treats the natural resources as assets which it must turn over to the next generation increased and not impaired in value".

There are four predominant categories of ES: provisioning, regulating, supporting, and cultural services. Provisioning services are benefits to people that can be extracted from nature, including food, water, and timber. Regulating services are processes that moderate natural phenomena, including erosion and flood control, water quality, and climate change. Supporting services govern regular underlying natural process consisting of photosynthesis, soil formation, and nutrient cycling. Finally, cultural services provide non-material benefits that contribute to the development and cultural advancement of people. Cultural services are comprised of recreational benefits, aesthetic value, and education (Table 7) (MA 2005; EFTEC 2005; NWF 2016).

*Table 7. Functions, Processes, Goods, and Services Provided by Various Ecosystem Services Modified from NRC (2004)*

| | Ecosystem Services | |
|---|---|---|
| Functions | Ecosystem Processes | Goods and Services |
| **Regulating** | Maintenance of essential ecological processes and life support systems | |
| Gas regulation | Role of ecosystems in biogeochemical cycles | Maintenance of air quality and influence on climate |
| Climate regulation | Influence of land cover and biologically mediated processes | Maintenance of temperature and precipitation |
| Water regulation | Role of land cover in regulating runoff and river discharge | Drainage and natural irrigation |
| Water supply | Filtering, retention, and storage of freshwater | Provision of water for consumptive use |
| Nutrient regulation | Role of biota in storage and recycling of nutrients | Maintenance of productive ecosystems |
| Waste treatment | Role of vegetation and biota in removal or breakdown of nutrients and compounds | Pollution control and detoxification |
| **Supporting/Habitat** | Providing habitat (suitable living space) for wild plant and animal species | |
| Refugium | Suitable living space for wild plants and animals | Maintenance of biological and genetic diversity; Maintenance of commercially harvested species |
| Nursery | Suitable reproductive habitat | Hunting; Gathering; Aquaculture |

*Table 7. Continued*

| | | |
|---|---|---|
| **Provisioning** | Provision of natural resources | |
| Food | Conversion of solar energy into edible plants and animals | Building and manufacturing; Fuel and energy; Fodder and fertilizer |
| Raw materials | Conversion of solar energy into biomass for human construction and other uses | Improve crop resistance to pathogens and pests |
| Medicinal resources | Variety of (bio)chemical substances in, and other medicinal uses of, natural biota | Drugs and pharmaceuticals |
| Ornamental resources | Variety of biota in natural resources with (potential) ornamental use | Resources for fashion, handicraft, worship, decoration, etc. |
| **Cultural** | Providing opportunities for cognitive development | |
| Aesthetic | Attractive landscape features | Enjoyment of scenery |
| Recreation | Variety in landscapes with (potential) recreational uses | Ecotourism |
| Cultural and artistic | Variety in natural features with cultural and artistic value | Inspiration for creative activities |
| Science and education | Variety in nature with scientific and educational value | Use of nature for education and research |

### 1.4.2 The Need for Quantification of Ecosystem Goods and Services

In order to discuss the topic of valuing ES, one must first analyze what is means to value something, and then discuss the role of economic valuation. From recent philosophical discussion, two points of view regarding ecosystem values have emerged (NRC 2004). First, nonhuman species have moral standings which indicate the values of ecosystems and the resulting benefits are non-anthropogenic. The second point of view focuses on the economic approach to valuation, thus all services are anthropocentrically centered (NRC 2004). In order to evaluate the monetary values of ES, the second point of view mentioned above will be the primary focus moving forward. The purpose of economic valuation is to convert all goods and services to a comparable common metric (MA 2005; EFTEC 2005). Furthermore, economic valuation does not incorporate all sources of value (especially the intrinsic values), however, it does account for the use of environmental resources (e.g., use values) as well as their existence or even absence of use (e.g., non-use values).

In order to categorize potential sources of value, total economic value (TEV) is utilized. TEV is dependent on changes in the ecosystem goods and services being valued, the scope of the analysis, and the temporal scale (NRC 2004; MA 2005). Again this typically involves quantifying values into a common metric, which in this case are monetary values (NRC 2004). This common metric is explained further by Daily et al. (1997) and Boyd and Banzhaf (2007) in that ecosystems are socially valuable in ways that may not be immediately apparent. Furthermore, the fields of ecology and economics are currently working to develop a standardized definition of ecosystem services and their measurement (NRC 2004; MA 2005; Boyd and Banzhaf 2007). The metric provides guidance and understanding to users about allocating resources between generations, where monetary values based on market prices usually neglect the rights of future generations (Groot et al. 2012). Furthermore, this common metric is required to provide decision makers with a tool to evaluate the trade-offs and synergies between modifying ecosystem management and the social actions that change the goods and services they provide (MA 2005; Granek et al. 2009).

### 1.4.3 Methods for Evaluating Monetary Values Provided by Ecosystem Services

The above describes how values are applied to various ES; the next task is to describe how society assesses those values. Since economic valuation is anthropocentric, the values that are assigned to goods or services are based on an individual's willingness to pay (WTP) or willingness to accept (WTA). The aggregation of all individual WTP or WTA determines the societal values of specific ES. However, this value is subject to change due to variability among individuals, current income, educational level on the topic, and outlook on ES. So, it can be said that values measured through economic valuation are subject to contextual and temporal scales (NRC 2004; BenDor et al. 2015).

Evaluating the value of all goods and services provided by an ecosystem can be completed through numerous methodologies (e.g., Replacement Cost and Cost of Treatment method; Hedonic Approach, Production Function method, Stated-Preference (Contingent) method, etc.) (NRC 2004). It is said that the Replacement Cost and Cost of Treatment method can serve as a last resort "proxy" valuation estimation for ecosystem services if the following conditions are met (Shabman and Batie 1978): 1) the alternative considered provides the same services, 2) the alternative used for cost comparison should be the least cost alternative, and 3) there should be substantial evidence that the service would be demanded by society if it were provided by the least cost alternative (Shabman and Batie 1978, NRC 2004). The difficulty associated with this method comes from the need to understand the demand for the service. Surveys and questionnaires need to be completed to devise a complete knowledge base of the population and their desire (WTP or WTA) for specific ecosystem services.

The Hedonic Approach categorizes valuation based on ecosystem services provided to a particular location, for example, those affecting the value of a house in that location (NRC 2004). Value is calculated using the hedonic price equation which accounts for the size of the ecosystem, proximity to the ecosystem, and a measure of ecosystem quality as it affects the desirability of human use (Beach and Carlson 1993; NRC 2004; MA 2005). Again, a complication with this method is the determination of the overall categorization of ecosystem quality as it provides goods and services to people.

The third methodology; the Production Function method, is generally completed using a two-step approach (Barbier 1994): 1) the physical effects of changes in a biological resource or ecological services on an economic activity are determined, and 2) the impact of these environmental changes are valued in terms of the corresponding change in the market output of the relevant activity (NRC 2004).

Additionally, the issue here is understanding the complete relationships of the dynamic systems that exist between biology, ecology, and economics.

The final methodology mentioned above is the Stated Preference (Contingent) method, which can be more widely applied allowing for estimates of valuation to be completed (NRC 2004). In order to provide these estimations two conditions are necessary: 1) information must be available to describe the change in ecosystems in terms of services people care about, in order to place a value on those services, and 2) change in the ecosystem must be explained in the survey instrument in such a way that people will understand and not reject the valuation scenario (NRC 2004).

Quantifying the monetary value of various ES provided to a region will allow for a comprehensive valuation analysis of urban stormwater BMPs. This type of analysis is crucial for several reasons: 1) to provide a basis for suggestions to amend traditional residential construction in an attempt to mitigate ecosystem degradation, 2) to allow for reevaluation of economic incentives related to ecosystem destruction, and 3) to develop a payment scheme to establish ES that currently have no market value (Busch et al. 2012).

## 1.5 Problem Statement

The population within the Lake Thunderbird watershed is approximately 100,000 inhabitants and it is estimated that the population will continue to grow with contributions from Cleveland and Oklahoma Counties (ODEQ 2013). As population continues to grow within the watershed, it is likely that the amount of impervious surfaces and pollution from these non-point sources will also continue to increase (ODEQ 2013).

According to the Oklahoma Conservation Commission (OCC 2010), one of the problems in the Lake Thunderbird Watershed is directly related to excess nutrient concentrations from non-point source pollution, which results in cultural eutrophication. The rapid urbanization of the watershed and pollution from non-point sources, specifically urban stormwater, is decreasing Lake Thunderbird's ability to supply drinking water and recreation (and thus meet its designated uses) and has resulted in the lake being listed as a 303(d) impaired waterbody (ODEQ 2013). A 303(d) impaired waterbody is too polluted or degraded to meet developed water quality standards; and the law requires the development of Total Maximum Daily Loads (TMDLs) for these waters. Thus, evaluating urban stormwater quality and quantity is crucial for determining how to manage the watershed and lake in the future.

## 1.6 Purpose

The purpose of this study is three-fold, 1) determine the overall difference in storm event volumes and peak discharges attributable to LID BMPs, 2) quantify the effectiveness of LID BMPs in decreasing urban stormwater pollutant concentrations and loads, and 3) provide an evaluation of relevant ecosystem services in an attempt to evaluate the use of LID BMPs as an alternative to traditional urban stormwater management.

## 1.7 Hypotheses

The three hypotheses for this project are as follows:

1. Utilization of LID BMPs will decrease the total volume of stormwater runoff generated and the peak volumetric discharge rate for any given storm event.
2. Implementation of LID BMPs will lead to a decrease in urban stormwater runoff pollutant concentrations and loads for ammonia-nitrogen, nitrate-nitrogen, total nitrogen, total dissolved phosphorus, total phosphorus, trace metals, and total suspended solids.
3. Employment of LID BMPs for urban stormwater management will provide ecosystem services (compared to traditional stormwater management) that can result in long-term economic benefits.

## 1.8 Objectives

To evaluate the defined hypotheses three objectives will be completed:

1. Collect storm-event derived stormwater runoff quantity data from treatment (incorporating LID BMP stormwater management practices) and control (incorporating traditional stormwater management practices) watersheds of similar size and residential land use.
2. Collect storm-event derived stormwater runoff quality data from treatment (incorporating LID BMP stormwater management practices) and control (incorporating traditional stormwater management practices) watersheds of similar size and residential land use.

3. Investigate the differences in economic benefits derived from ecosystem services between treatment (incorporating LID BMP stormwater management practices) and control (incorporating traditional stormwater management practices) watersheds of similar size and residential land use.

# 2.0 Methodology

## 2.1 Watershed Background, Study Site Location, and Purpose

This project is focused on the Trailwoods residential neighborhood within the Little River watershed, part of the Lake Thunderbird watershed in Central Oklahoma. The study site is located in the Trailwoods residential subdivision in Norman, OK (N 35°15’2.29”, W 97°27’3.47”) north of Rock Creek Road. The 258-square mile Lake Thunderbird watershed is comprised of predominately agricultural and residential land uses (Vieux and Associates 2007). The primary tributary to Lake Thunderbird is the Little River, other minor tributaries include, Hog Creek, Clear Creek, Dave Blue Creek, Jim Blue Creek, Rock Creek, Moore Creek, Kitchen Creek, and Elm Creek (Figure 2 and Figure 3) (OCC 2010).

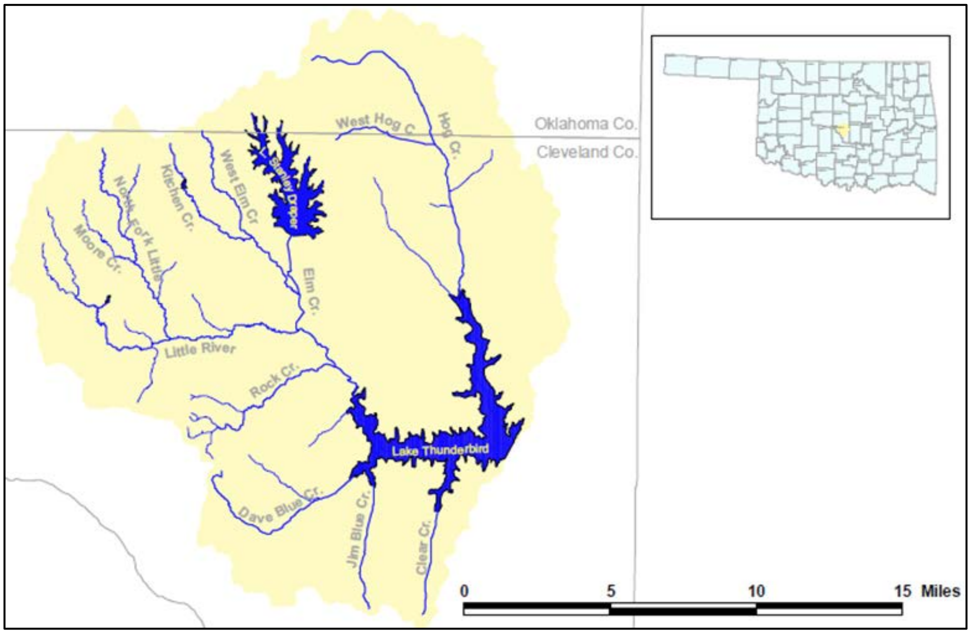


*Figure 2. Lake Thunderbird Watershed Modified from OCC (2010)*

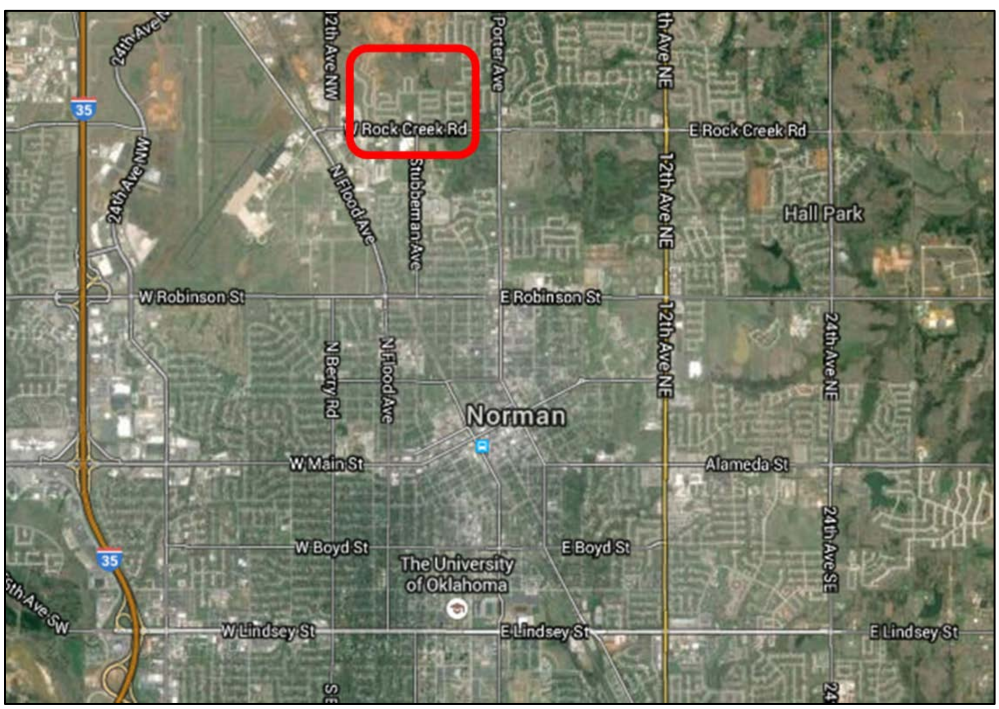


*Figure 3. Image Displaying Location of Trailwoods Study Site Relative to Norman, Oklahoma Modified From GoogleEarth*

Increased urbanization and development within the Lake Thunderbird watershed has severely impacted the water quality in the lake. It is estimated that 118,000 kg $yr^{-1}$ of total nitrogen (TN), 23,000 kg $yr^{-1}$ of TP, 200,000 kg $yr^{-1}$ of carbonaceous biochemical oxygen demand (CBOD), and 11,000 kg $yr^{-1}$ of TSS from urban stormwater runoff and agricultural production are being loaded into Lake Thunderbird (ODEQ 2013). Impacts of these loading rates (e.g., elevated turbidity, decreased DO, and excessive concentrations of chlorophyll-a) cause Lake Thunderbird to not support its designated uses for (a) Fish and Wildlife Propagation (FWP) for warm water aquatic communities and, (b) public drinking water supply (OCC 2010). Furthermore, Lake Thunderbird is considered a Sensitive Water Supply (SWS), which is defined by the Oklahoma Water Resources Board (OWRB 2011) as, "waters of the state which constitute sensitive public and private water supplies as a result of their unique physical conditions". According to these regulations, a SWS may have no new point source discharges of any contaminant after June 11, 1989 (OWRB 2011).

Stormwater management systems in the City of Norman must abide by one of two conveyance guidelines: 1) the stormwater system will convey runoff from a $Q_{10}$ precipitation event (5.88 inches/day) in a pipe network with overland flow capabilities and this combination will allow for proper management of a $Q_{100}$ precipitation event (8.75 inches/day) under fully urbanized conditions, or 2) if the full runoff volume from a $Q_{100}$ precipitation event is to be contained in a closed pipe network, a bypass system will be designed based on a 50 percent blockage of the pipe network (City of Norman 2006). Furthermore, storage and infiltration systems in the City of Norman must include, basins, ponds, infiltration trenches, dry wells, or porous paving to promote stormwater storage and resulting infiltration while also decreasing erosion and sediment transport (City of Norman 2006). The storage systems will be designed based on two specific criteria, 1) peak release rates from developments will not exceed the existing runoff that occurred before urbanization for all recurrence intervals up to and including a $Q_{100}$ precipitation event, and if improvements are to be made on any downstream channel it is required that the current floodplain storage is maintained, and 2) excess runoff due to urbanization from all precipitation events, including a $Q_{100}$ event, will be contained in the storage systems while ensuring peak discharge rates do not exceed that of pre-development conditions (City of Norman 2006). Water quality is not a topic of discussion in this regulatory report, which is concerning since a significant portion of the urban stormwater eventually ends up in Lake Thunderbird. Without addressing any of the pollution from urban stormwater it is likely the lake will remain impaired. Therefore it is important to evaluate alternatives to urban stormwater management to promote sustainability and conservation of drinking water sources such as Lake Thunderbird.

## 2.2 Study Site Description

This study is based on evaluation of the Trailwoods neighborhood to address and evaluate the impacts of LID BMPs on stormwater management. The study site is a 4.59-acre portion of the Trailwoods residential neighborhood, including a horseshoe shaped portion of the development divided into two watersheds, Trail West (TW or Treatment) and Trail East (TE or Control) (Figure 4). Design of the site controlled for

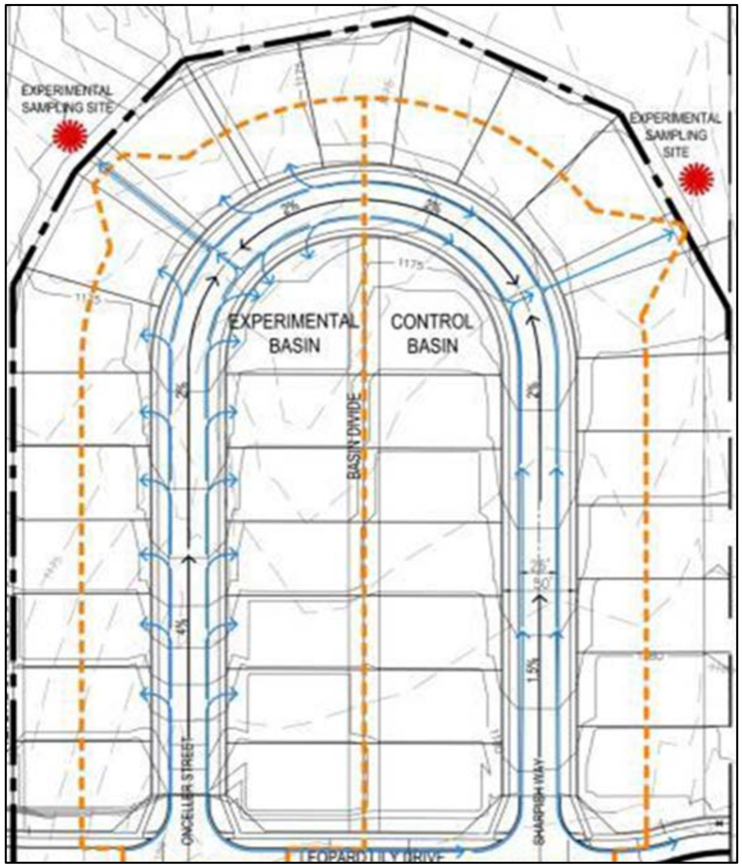


*Figure 4. Schematic of Trailwoods Study Site, Showing Anticipated Flow Paths and Division of Watersheds Modified from Coffman (2014); Experimental = TW or Treatment and Control = TE or Control*

the construction of 35 homes, sidewalks, driveways, and roads to encompass similar amounts of impervious surface area. Further considerations were given to the slope of each basin, soil composition, and types of LID BMPs to be implemented in the TW treatment portion of the residential neighborhood. TW treatment contains 18 rain gardens, 17 rain barrels, diverted downspouts, and a 120 square foot section of permeable pavement. The similarity between the two watersheds aside from the applied LID BMP technology allows for a direct comparison between the paired watersheds (Coffman 2014).

## 2.3 Study Site Design

Each rain garden was approximately 256 square feet, with a total average depth of approximately 1.75 feet. The engineered substrate was composed of 70 percent expanded clay, 20 percent sand, and 10 percent compost by volume which was acquired from Marcum's Nursery in Goldsby, Oklahoma. Xeric vegetation species were placed at the top of the basin, which transitioned to more mesic species near the basin outlets (Coffman 2014). Each residence included one fifty-gallon rain barrel placed in the front of the house and if possible near the rain gardens. These barrels were outfitted with an insect screen and a six-foot poly hose to allow for use of the captured rain water. Furthermore, when gutters were not emptied into rain barrels they were diverted into grassy swales that flowed either into rain gardens or to sub-surface piping that eventually reached the rain gardens. The final LID BMP that was utilized at the Trailwoods site is a 120 square foot section of permeable pavement, located approximately ten feet upstream of the sampling point. Permeable pavement is pavement that is more porous than ordinary asphalt or concrete, allowing for increased infiltration of urban stormwater, while also providing pore space for suspended sediment to be captured (Booth and Leavitt 1999; Dhalla and Zimmer 2010; EPA 2015b). An important note is that the City of Norman issued a variance to the traditional stormwater management methods, which allowed for the modification of infrastructure to accommodate this study.

## 2.4 Hydrologic Monitoring Methods

Stormwater runoff generated in the watersheds was diverted to concrete stormwater flumes designed for a 100 year frequency ($Q_{100}$) storm event (Table 8). Downstream of each stormwater flume, prefabricated Fiberglass reinforced polyester (FRP), 18” x 45° trapezoidal test flumes (Plasti-Fab Inc.) and ISCO model 6712 autosamplers were installed (Figure 5). The trapezoidal test flumes were designed to convey and measure flows of stormwater from $Q_2$ to $Q_{100}$ storm events. Small access points (0.25 inch from the bottom of the flume) in the side of the flume allowed for the ISCO model 730 bubbler module lines to be located in the flow path of the stormwater. The bubbler module and autosampler system were placed inside a storage safe near the trapezoidal flume in order to monitor the hydrology of the basin. The bubbler module was used to measure the water levels passing through the flume. An internal air compressor forces a known amount of air through the bubble line that is submerged in the flow channel and, the ISCO 6712 autosampler computer then records the datum (TELEDYNE ISCO 2005). Once water levels within the flume reached 0.15 feet, the autosampler was programmed to activate. The 0.15 foot activation level was selected through a back calculation of *level* based on a $Q_2$ precipitation event (Equation 1).

$$\text{Flow (CFS)} = 2.853 * (level + 0.13558)^{2.497} \qquad \text{Eq. 1}$$

*Table 8. Calculated Design Flows for Storms of Relevant Recurrence Intervals Modified from Coffman (2014)*

| Design Storms | Trail West Design Flows (CFS) | Trail East Design Flows (CFS) |
|---|---|---|
| Q2 | 6.40 | 7.38 |
| Q5 | 7.49 | 8.84 |
| Q10 | 8.53 | 10.07 |
| Q25 | 9.89 | 11.67 |
| Q50 | 11.24 | 13.27 |
| Q100 | 12.49 | 14.74 |

*Figure 5. Flume and autosampler installation at Trailwoods (TW) treatment watershed*

Equation 1 was developed by Plasti-Fab specifically for the trapezoidal flumes implemented at the Trailwoods Site. Equation 1 is needed in order to calculate discharge rates in cubic feet per second (CFS), where *level* stands for the head or depth of water (feet) in the test flume. Calculated flows were then plotted versus time to develop hydrographs for the two watersheds.

## 2.5 Stormwater Runoff Sample Collection

The samplers were programmed to rinse and purge the sample line three times before collecting a sample. For this portion of the Trailwoods study, autosampler activation triggered a sample to be taken immediately after activation, and then collect 20-mL samples for every 50 cubic feet of stormwater that passed through the flume, generating a single composite sample for a given storm event. The composite samples were then stored in five-liter Nalgene sample bottles until collection immediately after a precipitation event and within 24 hours. This sample method is considered a flow-activated storm composite sample, because samples were collected at equal intervals of flow during the event, allowing for a representative sample of stormwater runoff to be collected. When collected stormwater volume was sufficient, three different sub-samples were collected for laboratory analyses for each storm event. Samples were collected based on the Center for Restoration of Ecosystems and Watersheds (CREW) Standard Operating Procedures (SOPs) under an EPA-approved Quality Assurance Project Plan (QAPP) (OCC 2015). Nutrient sub-samples (e.g., TN, nitrate-nitrogen ($NO_3$-N), ammonia-nitrogen ($NH_3$-N), TP, and total dissolved phosphorus (TDP)) were collected in 250-mL sample bottles with zero head space. A one-liter sub-sample was taken for TSS; this sample was also taken as a zero head space sample. Finally, sub-samples for total metals (e.g., Al, As, Ca, Cd, Co, Cr, Cu, Fe, K, Mg, Mn, Na, Ni, Pb, and Zn) analyses were also collected in 250-mL sample bottles and preserved with Fisher Trace Metal Grade Nitric Acid to a pH $< 2$ until analysis was completed.

## 2.6 Water Quality Laboratory Analyses

The information in Table 9 outlines the selected water quality constituents analyzed for this project. All methods are in compliance with EPA guidelines for stormwater quality analysis.

*Table 9. Selected Laboratory Water Quality Parameters*

| Parameter | Units | Methods |
|---|---|---|
| Total suspended solids | mg/L | EPA 160.2 (1999) |
| Total nitrogen | mg/L | HACH TNT 10071 |
| Ammonia-nitrogen | mg/L | HACH TNT 10031 |
| Nitrate-nitrogen | mg/L | EPA 352.1 (1971) |
| Total phosphorus | mg/L | EPA 365.3 (1978) |
| Total dissolved phosphorus | mg/L | EPA 365.3 (1978) |
| Total metals (e.g., Al, As, Ca, Cd, Co, Cr, Cd, Fe, K, Mg, Mn, Na, Ni, Pb, and Zn) | mg/L | EPA 3015 (1994); EPA 6010C (2000) |

Analysis of TSS in urban stormwater was completed using EPA Method 160.2 (1999). This method is gravimetric and analyzed non-filterable residue in stormwater, with a range of detection from 4 to 20,000 mg/L TSS. The method uses a well-mixed sample filtered through Whatman #4 paper filter with a pore size of 20-25 micrometers (μm) retaining sediment suspended. After a known volume of water was passed the Whatman #4 paper filter, it was dried to constant mass at 100°C, allowing for determination of TSS in mg/L.

Determination of TN in urban stormwater was completed using the HACH TNT Method 10071. This method uses an alkaline persulfate digestion converting all forms of nitrogen to nitrate using a Chemical Oxygen Demand (COD) Reactor as a heat source. Addition of sodium metabisulfite eliminates any halide interferences. Nitrate then reacts with chromotropic acid under highly acidic conditions to form a yellow color that was measured at 410 nm using a HACH DR 2800 Portable Spectrophotometer (HACH 1997).

Analysis of $NH_3$-N in urban stormwater was determined by using the HACH TNT Method 10031. Ammonia compounds combine with chlorine to form monochloramine, which reacts with salicylate to form 5-aminosalicylate. This compound is oxidized in the presence of sodium nitroprusside to form a blue colored compound that was measured at a wavelength of 655 nm using a HACH DR 2800 Portable Spectrophotometer (HACH 1997).

Determination of $NO_3$-N in urban stormwater was completed using the EPA Method 352.1 (1971). This is a colorimetric method and is based upon the reaction of nitrate ions with brucine sulfate in a 13 N sulfuric acid solution at 100°C. The samples were brought up to temperature using a Fisher Scientific Isotemp water bath. Control of temperature is highly important for this measurement to allow for the proper color formation to occur. The absorbance of these samples was measured at 410 nm using a Cole Parmer 2800 UV/VIS Spectrophotometer.

Analysis of TP in urban stormwater was determined using the EPA Method 365.3 (1978). This is another colorimetric method based on the reaction of ammonium molybdate and antimony potassium tartrate in an acid medium with dilute concentrations of phosphorus to form an antimony-phospho-molybdate complex. Samples were heated for 30 minutes at 121°C using a Yamato SM200 Autoclave. Addition of ascorbic acid reduces the complex to an intensely blue-color complex, where the color of the solution is proportional to the phosphorus concentration. Absorbance was measured at 650 nm using a Cole Parmer 2800 UV/VIS Spectrophotometer.

Determination of TDP in urban stormwater was also completed utilizing the EPA Method 365.3 (1978). The only difference between this method and the TP method is that samples were first filtered through a phosphorus-free filter which had 0.45 μm pore size. Absorbance values were measured at 650 nm using a Cole Parmer 2800 UV/VIS Spectrophotometer.

Analysis of total metals in urban stormwater was completed utilizing two EPA methodologies: digestion via EPA Method 3015A (1994) and analysis via EPA Method 6010C (2000). EPA Method 3015A utilizes a preserved representative aqueous sample which used concentrated nitric acid for metal extraction. Prepared samples were then transferred to a CEM Corporation MARS Xpress Microwave System to be heated for a specific period of time (approximately 20 minutes). After cooling, samples were transferred to appropriate storage vessels until analysis could be completed. EPA Method 6010C (2000) determines various metal concentrations using inductively coupled plasma-optical emission spectrometry (ICP-OES). A Varian Vista Pro simultaneous axial ICP OES was used to measure metal emission spectra. Individual samples were nebulized, and the resulting aerosol was transported to the plasma torch, which produced element-specific emission spectra. These element-specific spectra were monitored by photosensitive devices, producing the concentration of the metal in question.

## 2.7 Quality Assurance and Quality Control

Field duplicates were collected to document the precision of the sampling process. Laboratory duplicates were also utilized in all analyses, as directed by CREW laboratory SOPs to ensure laboratory work remained consistent. Field blanks were exposed to the sample field conditions as the samples that were collected. Laboratory blanks were analyte-free solutions that were carried through the complete sample preparation and analytical procedure. With the purpose of documenting any contamination stemming from the analytical procedure, these types of samples were used in all water quality analyses and represented at least 10 percent of all analyses.

## 2.8 Statistical Analysis

Analysis of variance (ANOVA) on water quality and quantity data was completed to determine if there were any significant differences between the two basins, “assuming normal distribution of the regression residuals, equal variances, and independence” (OCC 2015). Furthermore, the evaluation of covariance (ANCOVA) aided in the determination of significant impacts realized from the LID BMP treatment

(OCC 2015). Additionally, a Spearman's rank correlation coefficient test was also performed on the data collected producing the value of Spearman's rho ($\rho$), which was then used in conjunction with a table of critical values to determine the statistical significance.

## 2.9 Evaluation of Monetary Value Provided by Ecosystems Services

Of the four broad categories that define ecosystem services (e.g., provisioning, regulating, supporting, and cultural), only provisioning and regulating services were evaluated for this study. The reasoning for only valuing the provisioning and regulating services was due to the type of data collected for study. Stormwater quality and quantity data provided the necessary values for decreased stormwater volume and percent removals of stormwater contaminants for valuation of goods and services at the Trailwoods study site. In order to value cultural or supporting services provided, it would have been necessary to complete a series of surveys determining the value residents place on aesthetic and educational value, as well as a habitat assessment to quantify the amount of habitat provided by the LID BMPs. The provisioning service evaluated delivers benefits in the form of rain water harvesting for beneficial reuse, thus decreasing potable water need of each resident. The regulating service evaluated provides benefits to people through processes that moderate natural phenomena, such as flood attenuation and removal of various stormwater pollutants.

### 2.9.1 Provisioning – Rain Barrels

Calculation of the provisioning value provided by rain barrels at the Trailwoods study site was relatively straight-forward. Assumptions that were required include, 1) 100 percent of rainfall produces 100 percent runoff from a single roof, 2) the percentage of each roof that drained into each rain barrel was assumed identical for all residences, 3) events greater than 0.24 inches completely filled the rain barrel and after the event the barrels were completely emptied, 4) events smaller than 0.24 inches only partially filled the rain barrel, which was drained before the next 0.24 inch event, and 5) residents used the captured stormwater for beneficial reuse. The first step was to determine the quantity and size of the rain barrels in the study location. At the Trailwoods study site, there were 17 fifty-gallon rain barrels attached to downspouts for each residence in the treatment watershed. Next, one must consider how much of each individual roof drains into a single rain barrel. The average surface area of the roofs in the Trailwoods neighborhood was approximately 2000 square feet. The amount of the roof that drains into the rain barrels was determined through field verification and Google Earth imagery, resulting in an estimation that approximately 30 percent of each roof contributed runoff into a rain barrel. These estimates allowed for the calculation of the approximate rainfall depth required to fill the rain barrels; that value was 0.24 inches. For the study duration, 24 precipitation events produced depths greater than 0.24 inches. During the calendar year 2015 there were 56 precipitation events that produced greater than 0.24 inch depths.

Utilizing water rate data provided by the City of Norman (2016), the price per gallon of potable water replaced by harvested rain water was calculated. The price per gallon of water was multiplied by the volume of rain water harvested for precipitation events greater than 0.24 inches for the study period. The resulting value was extrapolated to an annualized value per household and for the entire treatment watershed.

### 2.9.2 Regulating – Flood Attenuation

Calculation of the regulating ecosystem service through flood attenuation was completed using various storm event total runoff volume percent reductions (0, 25, 38, 50, 75, and 95 percent) to determine differences at varying levels of LID BMP implementation. Assuming a zero percent reduction in total discharge indicated the complete absence of LID BMPs, while a 25 percent reduction assumed a lesser amount of LID BMPs actually implemented at the Trailwoods site. The 38 percent reduction is the actual mean percent reduction in total discharge volume measured at the Trailwoods study site from May to September 2015. In addition, the 50, 75, and 95 percent reduction were predicted values if a greater number of LID BMPs were implemented at the site.

The data required to complete these analyses were acquired from several sources, including, Young et al. (1996), Brown and Schueler (1997) Narayana and Pitt (2006), City of Norman (2006), Dhalla and Zimmer (2010), Oklahoma Mesonet (2016), and Landwatch.com (2016). Overall, the monetary valuation of stormwater discharge attenuation was completed in ten steps: 1) determination of total discharge rate reduction, 2) determination of design storm and subsequent traditional BMP (e.g., retention or detention pond) design for such an event, 3) assumption of hydraulic retention time (HRT), 4) assumption of pond depth, 5) calculation of pond volume based on HRT and discharge from design storm, 6) conversion of units to input into cost equations, 7) calculation of surface area (SA) required, 8) calculation of construction cost using empirical cost equations, 9) determination of total required land costs, 10) calculation of operation and maintenance costs.

Storm event discharge rate reductions were measured at the study site from May to September 2015, comparing total discharge from the treatment watershed to that of the control watershed. The data required to size a stormwater pond for a $Q_{100}$ precipitation event and the magnitude of the event itself were collected from City of Norman (2006). Retention pond design controls, HRT and depth were referenced from Young et al. (1996) and Dhalla and Zimmer (2010). Retention basin volumes were calculated using the known HRT and inflow from a $Q_{100}$ precipitation event. Conversion from cubic feet to Mgal was required in order to calculate an approximate cost of construction and cost equations were collected from Young et al. (1996) and Brown and Schueler (1997) (Table 10). In order to determine cost of land, a surface area was calculated from the calculated volume and known depth values, and land prices for Norman, OK were acquired from Landwatch.com (2016) on June 15$^{th}$, 2016. This valuation assumed that the stormwater pond was optimally sized and designed from an engineering point-of-view. Finally, to calculate operation and maintenance costs over the life time of the retention pond, empirical data from Barr Engineering Company (2011) was utilized (0.07 USD/ cubic foot of pond volume).

*Table 10. Equations used to Determine Construction Costs for Various Stormwater Ponds; Young et al. (1996): C = Cost of Construction (USD) and V = Volume of Pond (MG); Brown and Schueler (1997): C = Cost of Construction (USD) and V = Cubic feet of $Q_{10}$ Precipitation Event*

| Source: | Cost calculated for: | Equation |
|---|---|---|
| Young et al. (1996) | Retention pond | $C = 61{,}000 * V^{0.75}$ |
| | Detention pond | $C = 55{,}000 * V^{0.69}$ |
| Brown and Schueler (1997) | General stormwater pond | $C = 12.4\, V^{0.76}$ |

### 2.9.3 Regulating – Nutrient Retention

Calculation of the regulating ecosystem service through nutrient retention provided by LID BMPs at the Trailwoods study site was completed through modification of several methodologies summarized by EPA (2007). Overall, this method was completed using a four step process outlined by EPA (2007): 1) mass removal performance (mass/year) was measured, 2) capital costs (USD) were estimated, 3) life cycle costs (USD/year) were estimated, and 4) cost effectiveness values (USD/percent removal/year) were estimated. LID BMP capital costs were estimated using data from Coffman (2014), which provided costs per square foot of LID BMPs at the study site. An assumption was made with these data that they included cost of materials and design of the LID BMPs. Construction cost data acquired from the Bureau of Labor Statistics (2013) provided mean hourly wages for construction laborers and first-line supervisors. Life cycle costs were estimated using a LID BMP design guide completed by City of Edmonton (2011) which reviewed LID BMP case studies by Capital Regional District Water Services (CRDWS 2008), Peak

(2003), Southeast Wisconsin Regional Planning Commission (SWRPC 1991), Toronto and Region Conservation Authority (TRCA 2009), and Wayne County (2001). These studies provided life cycle cost metrics that could be directly applied to the LID BMPs at the study site (Table 11).

Monitoring and collection of urban stormwater runoff from May to September 2015 allowed for removal performance to be calculated. To calculate the cost effectiveness of the LID BMPs implemented at the study site, a five step approach was used (Table 12). Numerous assumptions had to be made in order to calculate the cost effectiveness of the LID BMPs at the Trailwoods study site (Table 13). The construction time (37 days) was assumed that each rain garden took two days to install, while permeable pavement installation took only one day. Rain barrels were placed next to gutters and therefore time of installation was not considered. Revenue in this case is defined as value generated from rain barrels and overall flood attenuation from LID BMPs at the study site as calculated in the previous two sections.

*Table 11. LID BMP Life Cycle Costs Modified from City of Edmonton (2011)*

| LID BMP | Annual maintenance (USD) | Life cycle (years) |
|---|---|---|
| Rain barrels | 25 | 25-100 |
| Bioretention | 13-30/$m^3$<br>Major rehab every 20 years:<br>4-170/$m^2$ | >20 |
| Permeable pavement | 0.15-0.30/$m^2$ | >20 |

*Table 12. Five Step Approach to Calculate Cost Effectiveness of LID BMPs at the Trailwoods Study Site Modified from EPA (2007): r = Discount Rates, n = Life Time, NPV = Net Present Value, $A_{t,r}$ = Annuity Rate, EAC = Equivalent Annual Cost*

| Source | Federal Reserve Bank of St. Louis | EPA (2007) | EPA (2007) | EPA (2007) | EPA (2007) |
|---|---|---|---|---|---|
| Description | Discount Rates (r) | Net Present Value (NPV) | Annuity Rate ($A_{t,r}$) | Equivalent Annual Cost (EAC) | Cost Effectiveness |
| Equation | --- | $1\sum^{t}\left[\frac{R_n}{(1+r)^n}\right]$ | $\frac{1-\left[\frac{1}{(1+r)^t}\right]}{r}$ | $\frac{\text{NPV}}{\text{A}_{\text{t,r}}}$ | $\frac{\text{EAC}}{\text{\%Removal}}$ |
| Units | Percent | USD | Dimensionless | USD/Year | USD/Percent /Year |

*Table 13. Necessary Assumptions to Calculate Cost Effectiveness of LID BMPs*

| Category | Assumption |
|---|---|
| Overall | No costs besides costs incurred from LID BMPs were evaluated |
| Capital costs | Cost data provided by Coffman (2014) included design and material cost for LID BMPs |
| Construction costs | Only two laborers and one supervisor installed all LID BMPs at the study site working at a mean hourly wage provided by the Bureau of Labor Statistics (2013) |
| | It took 37 days to install all rain gardens and section of permeable pavement |
| Revenue | Revenue generated was constant for duration of study |
| | Only two sources of revenue exist because other sources were not measureable within the scope of this study |

## 3.0 Results and Discussion

The results of this study stem from analysis of 10 precipitation events captured between May 22nd, and September 20th, 2015. Useable stormwater hydrographs and water quality data were collected for all 10 precipitation events.

## 3.1 Precipitation Data

Precipitation data for this study were collected from the Oklahoma Mesonet Norman Station (OCS 2016), located 1.35 miles southwest of the Trailwoods study site. On a few occasions daily rainfall was slightly greater than event total rainfalls, highlighting the spatial and temporal variability of precipitation events in central Oklahoma. Overall, for the duration of this study, the magnitude of individual precipitation event ranged from 0.34 to 3.99 inches with a mean and median of 1.39 and 0.71 inches, respectively (Table 14).

*Table 14. Rainfall Data and Statistical Summary for All Storm Events Sampled in this Study*

| Event Date | Daily Rainfall (in) | Max Five-minute Rainfall Rate (in/hr) | Previous Day Rainfall (in) | Event Total Rainfall (in) |
|---|---|---|---|---|
| 5/22/2015 | 0.68 | 0.48 | 0 | 0.66 |
| 5/24/2015 | 0.90 | 0.84 | 3.38 | 3.99 |
| 6/29/2015 | 0.51 | 2.04 | 0 | 0.50 |
| 7/3/2015 | 2.61 | 2.64 | 0.86 | 3.47 |
| 7/7/2015 | 1.88 | 2.04 | 0.02 | 1.88 |
| 7/21/2015 | 0.25 | 0.96 | 0 | 0.76 |
| 8/4/2015 | 0.50 | 0.48 | 0 | 0.50 |
| 8/19/2015 | 0.52 | 0.84 | 0 | 0.52 |
| 9/8/2015 | 0.41 | 1.56 | 0 | 0.34 |
| 9/20/2015 | 1.32 | 1.20 | 0.02 | 1.32 |
| Mean | 0.96 | 1.31 | 0.43 | 1.39 |
| Median | 0.59 | 1.08 | 0 | 0.71 |
| Std. Dev. | 0.72 | 0.70 | 1.02 | 1.25 |
| Maximum | 2.61 | 2.64 | 3.38 | 3.99 |
| Minimum | 0.25 | 0.48 | 0 | 0.34 |
| Std. Error | 0.23 | 0.22 | 0.32 | 0.40 |

According to data acquired from the Oklahoma Mesonet Norman Station (OCS 2016), there were a total of 48 days with ≥ 0.01 inches of precipitation and 36 days with ≥ 0.10 inches for the study period (Table 15). Of the 122 total days of monitoring for this study, 48 days represent 39 percent of the study days, however those 48 days only produced 10 storm events, which triggered measureable sampling episodes. Six percent of these events were < 0.05 inches, 2 percent were < 0.1 inches, and 8 percent were < 0.5 inches. These precipitation events were too small to generate enough runoff to trigger the autosamplers and were not considered in these analyses.

In the entire calendar year of 2014 the total rainfall at the Norman Mesonet was 21.63 inches, while in the month of May 2015 the monthly precipitation was 23.39 inches. Overall 2015 had a total of 63.22 inches of precipitation, which was an increase of 28.44 inches or 82 percent from the long-term average of 34.67 inches. The spring of 2015 (March, April, May, and June) was an exceedingly wet season representing 103 percent of the long-term annual average precipitation for Norman, Oklahoma.

*Table 15. Monthly Rainfall Statistics at Norman Mesonet Station for Duration of Study Period*

| | Monthly Rainfall (in) | # Days with Rain ≥ 0.01 in | # Days with Rain ≥ 0.10 in | Greatest 24 Hour Total (in) |
|---|---|---|---|---|
| May | 23.39 | 19 | 15 | 4.67 |
| June | 5.95 | 7 | 7 | 1.67 |
| July | 7.46 | 11 | 7 | 2.61 |
| August | 1.74 | 5 | 4 | 0.52 |
| September | 1.98 | 6 | 3 | 1.32 |
| Totals | 40.52 | 48 | 36 | --- |

## 3.2 Water Quantity Data

### 3.2.1 Storm Hydrographs

Data for development of storm event driven hydrographs were collected directly from the ISCO 6712 autosamplers. Initially, problems with programming the autosamplers resulted in missing some precipitation events in early May 2015. However, an acceptable number of events (n=10) were collected for the duration of the study.

Overall, Table 16 summarizes the hydrologic data for the study period at the Trailwoods study site. Total runoff volume discharged from the two watersheds varied greatly depending on the magnitude of the precipitation event. For events > 1.5 inches, the TW treatment watershed consistently produced lower total runoff volumes, however when precipitation events were < 1.5 inches the LID BMPs ability to decrease total runoff volume declined. This could be attributed to the extended falling limb of the hydrographs and a return to a higher base flow, for the June 29, July 21, August 4, August 19, September 8, and September 20 precipitation events (Figures 6 and 9-13). The assumed lesser efficiency could have also been caused by an artifact of the sampling method. When stormwater passed through the flume at the toe of the TW treatment watershed ponding would occur. Ponding would occur because of elevation errors in the concrete pad on the downstream end of the test flume, in that after stormwater passed the measurement point stacking would occur which resulted in false level measurements and thus artificially larger flow rates.

*Table 16. Event Total Rainfall and Resulting Total and Peak Discharge for Study Watersheds*

| Event Date | Event Total Rainfall (in) | TE Control Total Runoff Volume (CF) | TE Control Peak Q (CFS) | TW Treatment Total Runoff Volume (CF) | TW Treatment Peak Q (CFS) |
|---|---|---|---|---|---|
| 5/22/2015 | 0.66 | 4093.17 | 0.67 | 2078.89 | 0.24 |
| 5/24/2015 | 3.99 | 42699.07 | 9.93 | 24514.59 | 4.21 |
| 6/29/2015 | 0.50 | 2107.41 | 0.67 | 3931.50 | 0.46 |
| 7/3/2015 | 3.47 | 37079.15 | 9.26 | 20119.06 | 5.13 |
| 7/7/2015 | 1.88 | 5503.26 | 1.75 | 4619.81 | 0.80 |
| 7/21/2015 | 0.76 | 8775.65 | 1.25 | 11502.65 | 0.78 |
| 8/4/2015 | 0.50 | 2311.11 | 0.36 | 2698.12 | 0.23 |
| 8/19/2015 | 0.52 | 2854.52 | 0.67 | 5534.72 | 0.57 |
| 9/8/2015 | 0.34 | 774.03 | 0.88 | 1482.07 | 0.74 |
| 9/20/2015 | 1.32 | 4341.77 | 0.73 | 4780.42 | 0.61 |
| Mean | 1.39 | 11053.91 | 2.62 | 8126.18 | 1.38 |
| Median | 0.71 | 4217.47 | 0.80 | 4700.11 | 0.67 |
| Std. Dev. | 1.25 | 14620.59 | 3.51 | 7627.50 | 1.67 |
| Maximum | 3.99 | 42699.07 | 9.93 | 24514.59 | 5.13 |
| Minimum | 0.34 | 774.03 | 0.36 | 1482.07 | 0.23 |
| Sample Size | 10 | 10 | 10 | 10 | 10 |
| Std. Error | 0.40 | 4623.44 | 1.11 | 2412.03 | 0.53 |

This issue was magnified during smaller precipitation events because there was a lesser volume of stormwater so any increases in flow from TW treatment represented more of the total stormwater volume, thus skewing the hydrologic data to make it seem as TW treatment had more stormwater passing through than TE control. This issue was most likely the reason TW treatment had larger total runoff volumes when compared to TE control for precipitation events <1.5 inches. Moreover, despite the ambiguity based on precipitation event magnitude, for the duration of the study the mean total runoff volume was 26.5 percent lower in TW treatment than TE control. This is supported by the results of a study completed in a subdivision in Prince George's County, Maryland by Cheng et al (2005), which found a 20 percent decrease in total runoff volume in the experimental basin compared to the conventional basin. Furthermore, in all ten precipitation events, peak discharge was significantly decreased (p value = 0.040) by an average of 47.3 percent. This is evident when reviewing the peaks on each hydrograph, as every time TW treatment had a lower peak discharge than TE control. These results are supported by a series of LID BMP studies including Dhalla and Zimmer (2010), NRC (2008), and Cheng et al (2005), which found decreases in peak discharge rates of 40, 42, and 40 percent, respectively.

During May and June, three precipitation events produced measurable data. These events can be categorized by multiple peaks, precipitation ranging from 0.5 to 3.99 inches, and significant differences in peak discharge rates. The event on May 24-25, 2015 was the largest event measured for this study, and displays how the peak discharges are dampened and are followed by a gradual regression back to base flow conditions. Hydrographs for this period tracked one another very well as Figure 6-8 show below. Overall, peak discharge rates were lower for the TW treatment watershed, and apart from the June 29 event, total runoff volumes were also lower. Peak discharge rates were decreased by an average of 2.12 ± 0.805 CFS (51.3 ± 4.5 percent). Excluding the June 29 event, spring total runoff volumes were decreased by 10099.39 ± 2556.73 CF (45.9 ± 1.0 percent).

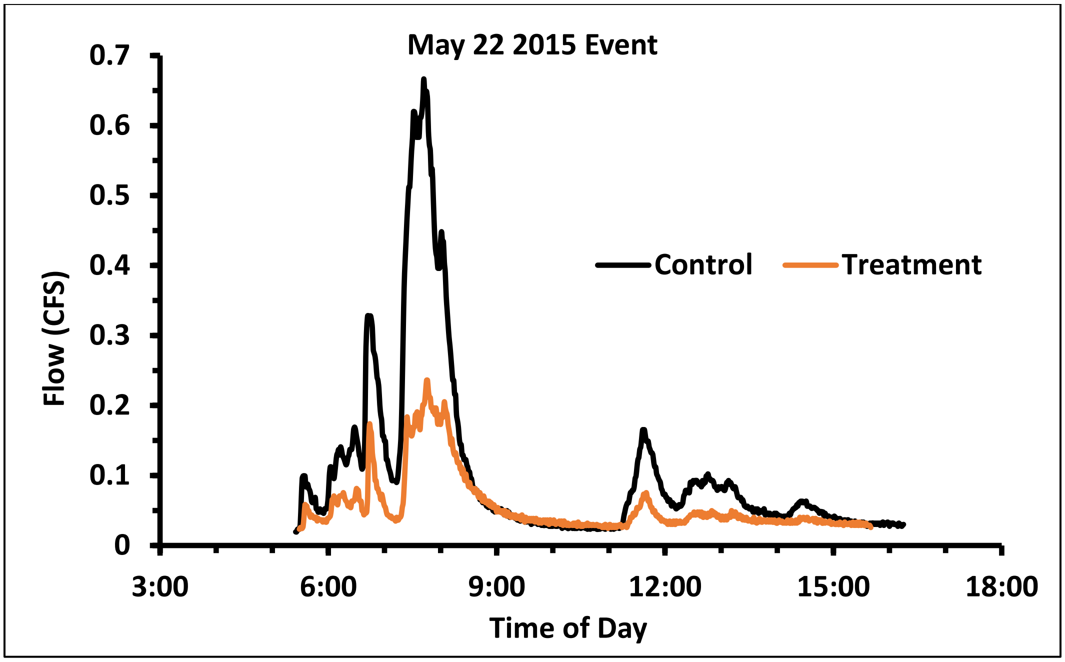


*Figure 6. TE Control and TW Treatment Watershed Hydrographs for May 22, 2015 Storm Event of 0.66 inches, Displaying Dampening of Multiple Peaks*

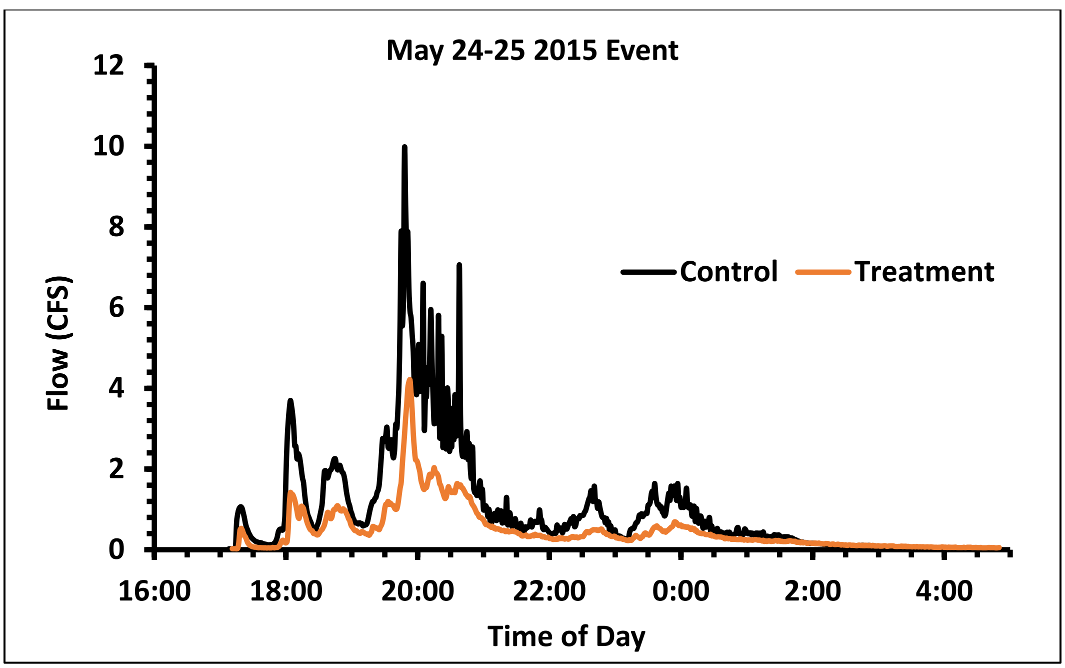


*Figure 7. TE Control and TW Treatment Watershed Hydrographs for May 24-25, 2015 Storm Event of 3.99 inches, Displaying Dampening of Multiple Peaks and Delayed Release of Stormwater from TW Watershed*

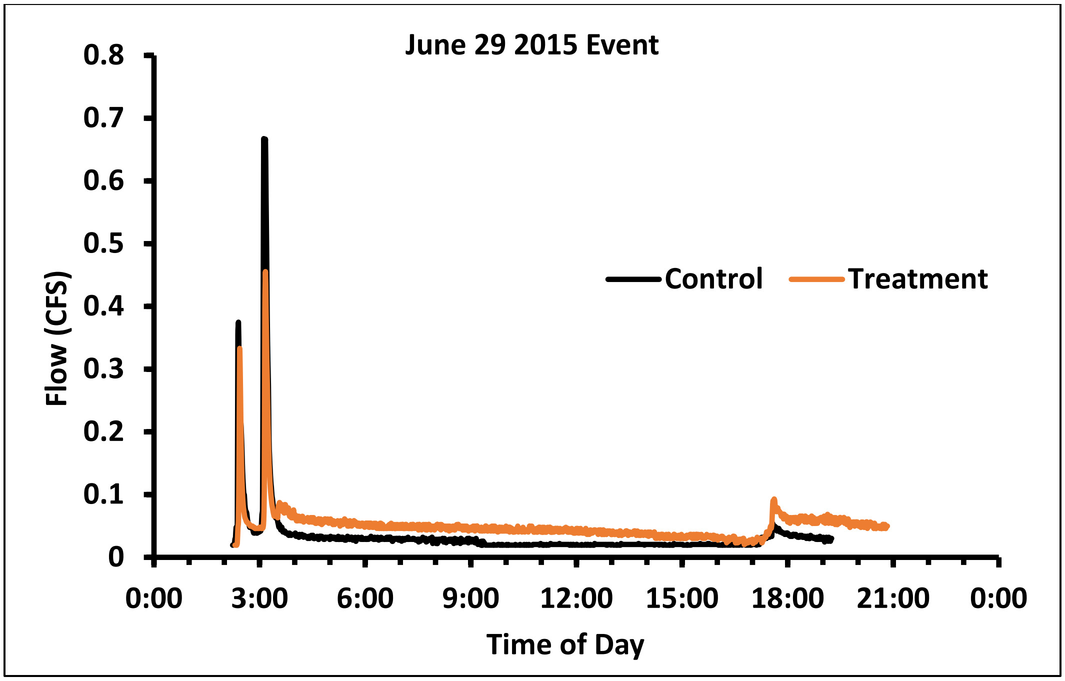


*Figure 8. TE Control and TW Treatment Watershed Hydrographs for June 29, 2015 Storm Event of 0.50 inches, Displaying Dampening of Multiple Peaks but a Return to Higher Base Flow Conditions*

During the summer of 2015, (July, August, and September) seven precipitation events produced measurable data. These events can be categorized by multiple peaks, precipitation ranging from 0.5 to 3.47 inches, and significant differences in peak discharge rates between TE control and TW treatment watersheds. The event on July 3, 2015 was the largest event captured during the summer, which again showed that peak discharges were dampened followed by a gradual regression back to base flow conditions.

Upon review of the hydrographs below (Figure 9-15), the two watersheds tracked one another well. Overall peak discharge rates were significantly lower (p value = 0.040) for the TW treatment watershed. In regard to the total runoff volume, although the LID BMPs at the Trailwoods site proved to be highly efficient in managing the spring's larger events (e.g., > 1.5 inches), but during smaller summer events (e.g., < 1.5 inches), total runoff volume was greater in the TW treatment watershed when compared to the TE control watershed, however peak discharge rates were decreased by an average of 0.86 ± 0.43 CFS (31.6 ± 4.69 percent). Summer total runoff volumes were decreased by 1557.52 ± 2024.52 CF (-25.9 ± 15.2 percent). Inability of LID BMPs to decrease total runoff volume during smaller summer events was attributed to the artifact of the sampling method discussed above. The decrease in total runoff volume is positive while the percent decrease is negative due to the magnitude of the July 3, 2015 event. This event was three to four times the size of the other summer events, which skews the decrease in volumetric units while maintaining the percent decreases.

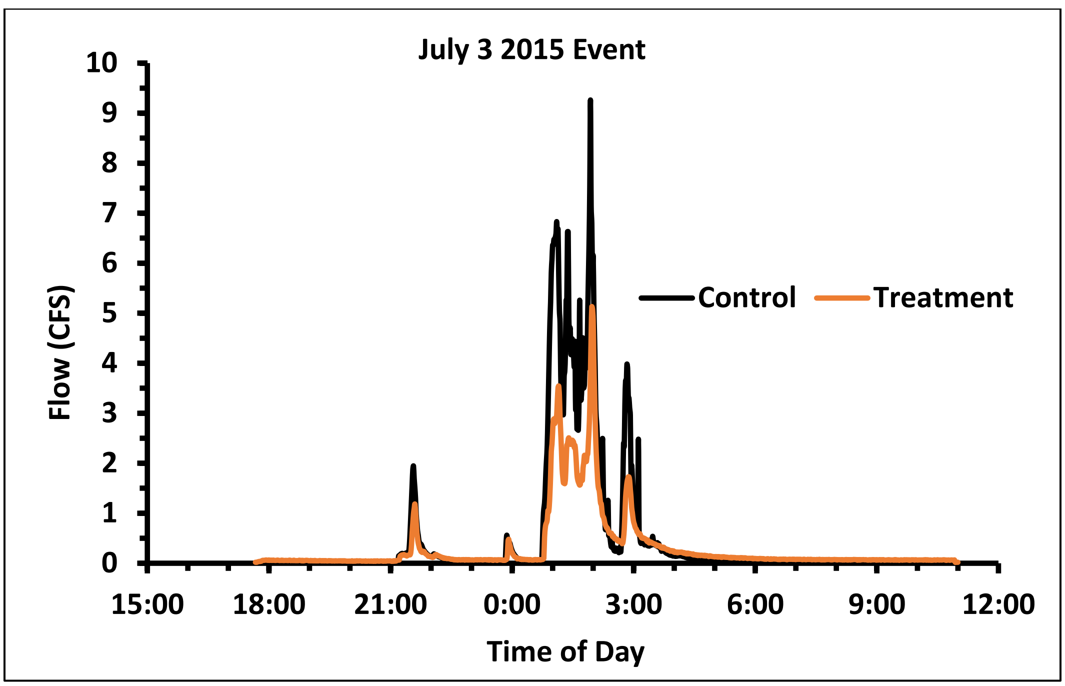


*Figure 9. TE Control and TW Treatment Watershed Hydrographs for July 3, 2015 Storm Event of 3.47 inches, Displaying Dampening of Multiple Peaks and Delayed Release from TW Treatment Watershed*

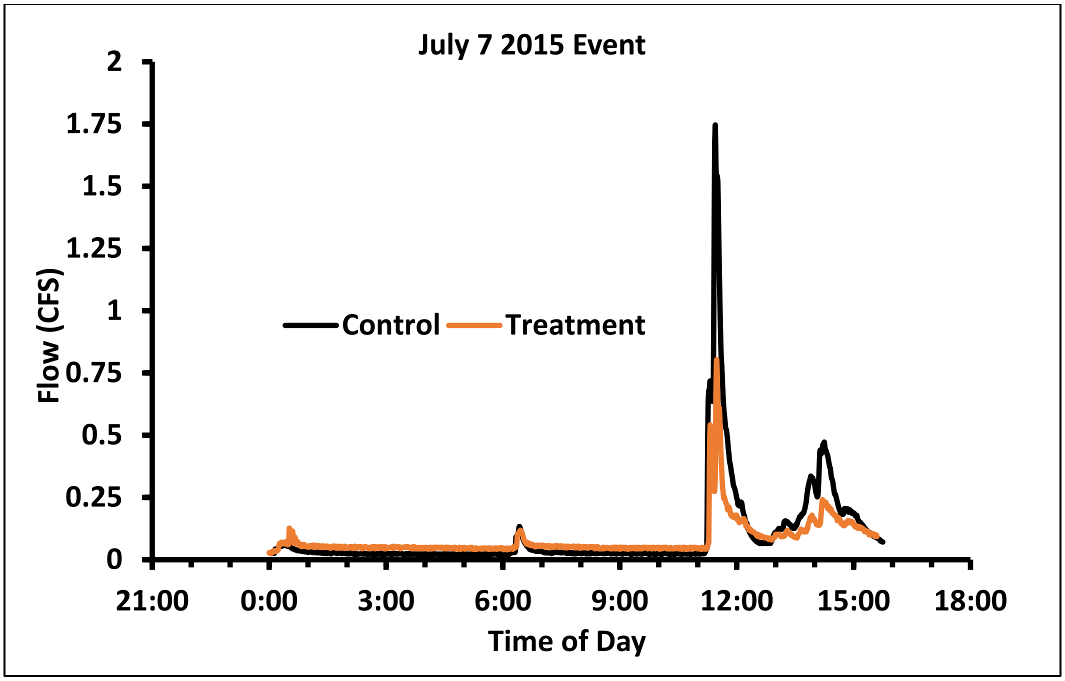


*Figure 10. TE Control and TW Treatment Watershed Hydrographs for July 7, 2015 Storm Event of 1.88 inches, Displaying Dampening of Multiple Peaks*

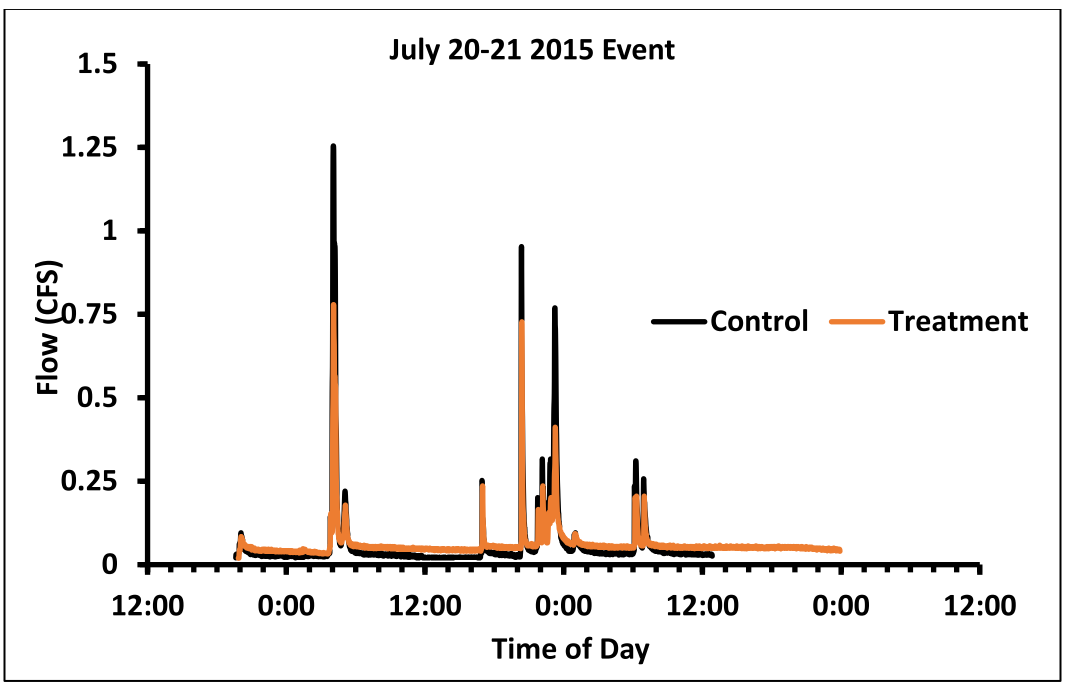


*Figure 11. TE Control and TW Treatment Watershed Hydrographs for July 20-21, 2015 Storm Event of 0.76 inches, Displaying Dampening of Multiple Peaks and Delayed Release from TW Treatment Watershed*

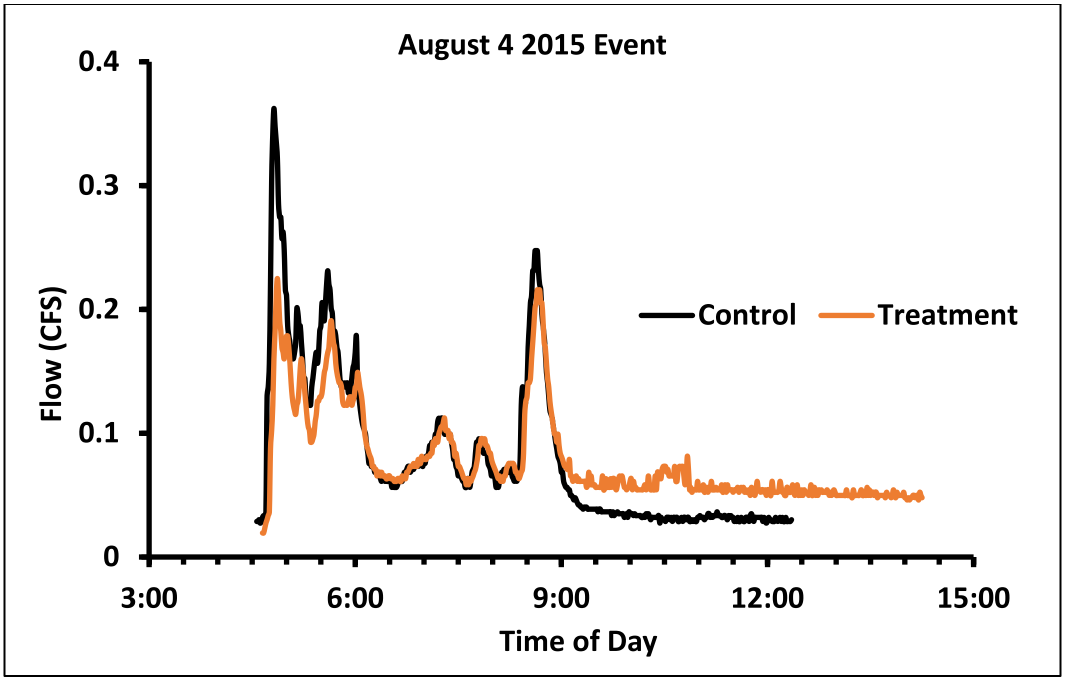


*Figure 12. TE Control and TW Treatment Watershed Hydrographs for August 4, 2015 Storm Event of 0.50 inches, Displaying Dampening of Multiple Peaks but a Return to Higher Base Flow Conditions*

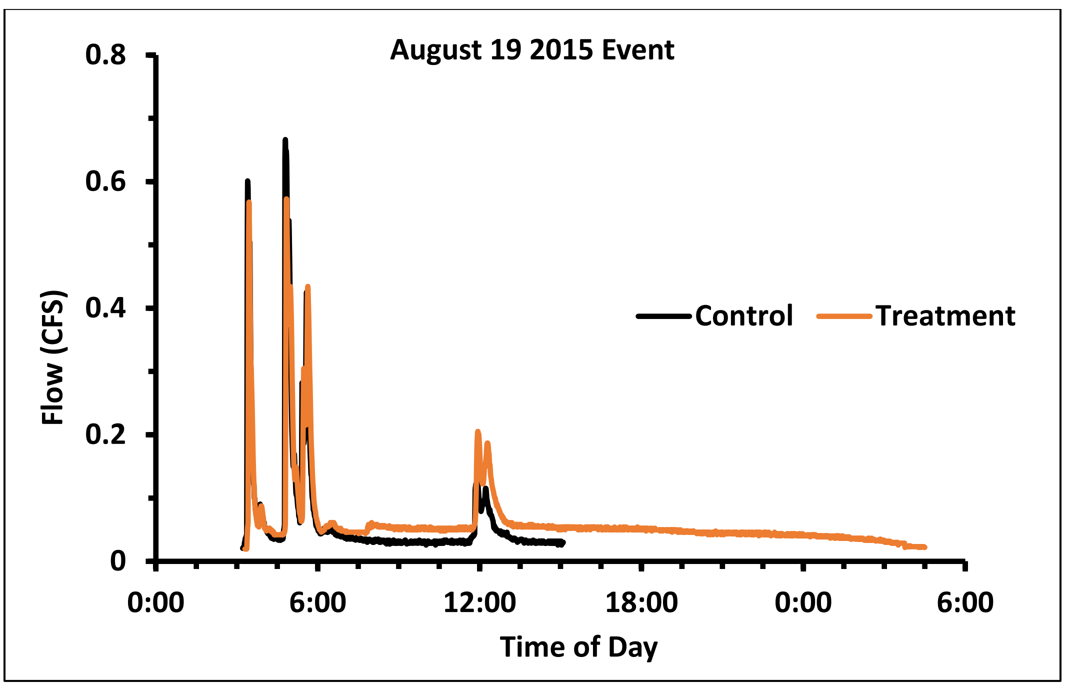


*Figure 13. TE Control and TW Treatment Watershed Hydrographs for August 19, 2015 Storm Event of 0.52 inches, Displaying Dampening of Multiple Peaks but a Return to Higher Base Flow Conditions*

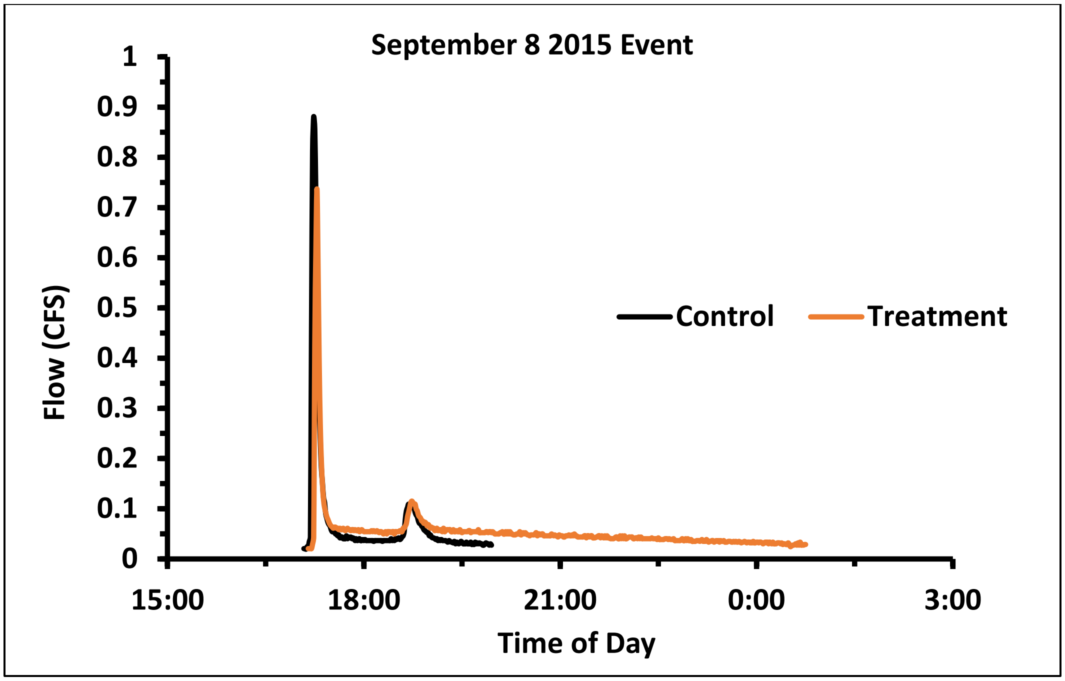


*Figure 14. TE Control and TW Treatment Watershed Hydrographs for September 8, 2015 Storm Event of 0.34 inches, Displaying Dampening of Multiple Peaks but a Return to Higher Base Flow Conditions*

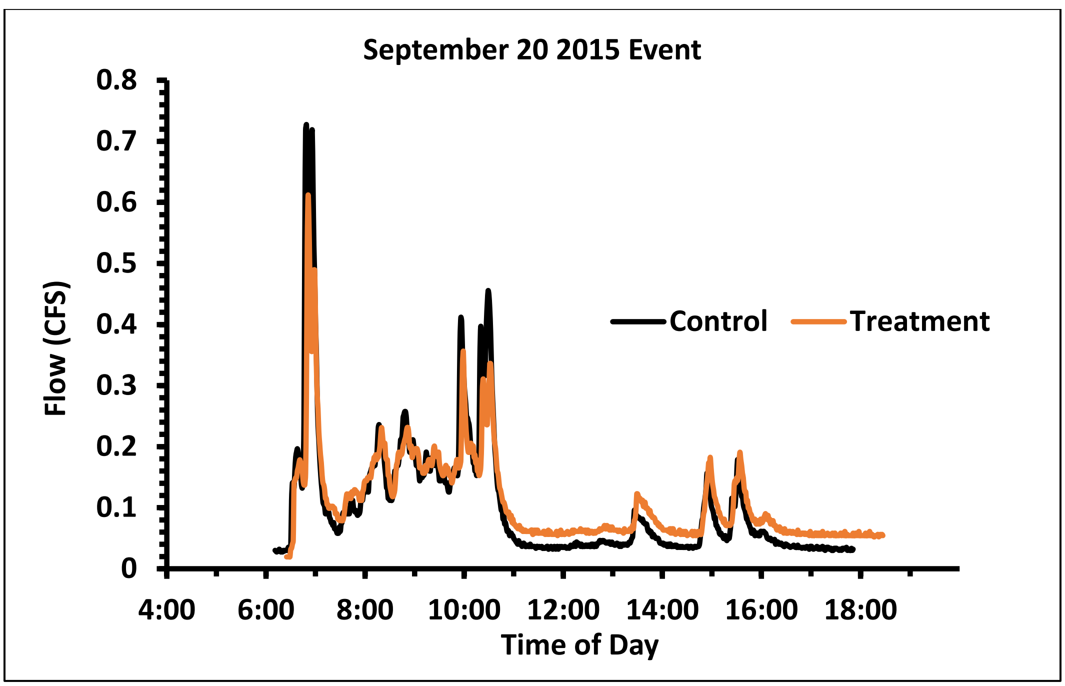


*Figure 15. TE Control and TW Treatment Watershed Hydrographs for September 20, 2015 Storm Event of 1.32 inches, Displaying Dampening of Multiple Peaks but a Return to Higher Base Flow Conditions*

### 3.2.2 Relationship between Precipitation and Volumetric Discharge Rates

An analysis of the relationship between the total event precipitation and peak discharge rates can be seen in Figure 16. There was a strong relationship for both the control and treatment watersheds, $r^2$= 0.9145 and 0.8559, respectively. Small precipitation events (< 1.5 inches) (n=7) always had a peak discharge less than 2.00 CFS, whereas the larger precipitation events (> 1.5 inches) (n=3) typically had larger peak discharge rates which ranged from 1.75 to 9.93 CFS (Table 17).

The maximum five-minute intensity of the precipitation event had no correlation with peak discharge rates. Figure 17 displays this relationship with $r^2$= 0.1071 and 0.1831 for TE control and TW treatment watersheds, respectively. This result was somewhat counterintuitive because one would think that the higher intensity for a given event would produce higher peak discharge rate for that event. When the regression slopes were compared for Figure 16-19, it was determined that the slopes for Figure 16 and Figure 18 were significantly different (p value = < 0.01), while the slopes for Figure 17 and Figure 19 were not significantly different.

*Table 17. Summary Statistics for Spearman's Rank Correlation Analysis (n=10)*

| | Event Total Precipitation vs Peak Q | Event Total Precipitation vs Total Runoff Volume | Five-Minute Intensity vs Peak Q |
|---|---|---|---|
| # of Pairs | 10 | 10 | 10 |
| Spearman ρ Value | 0.7272 | 0.8000 | 0.5818 |
| Significance | 0.01 | <0.005 | <0.05 |

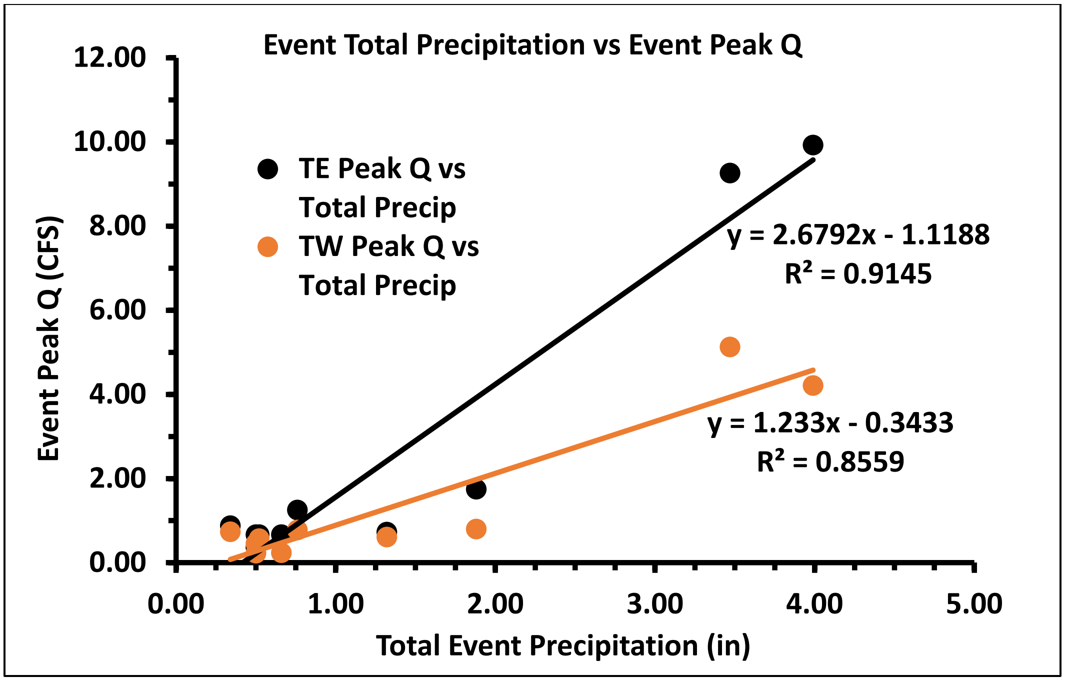


*Figure 16. Plot of Event Total Precipitation (in) versus Event Peak Q (CFS) for TE Control and TW Treatment Watersheds*

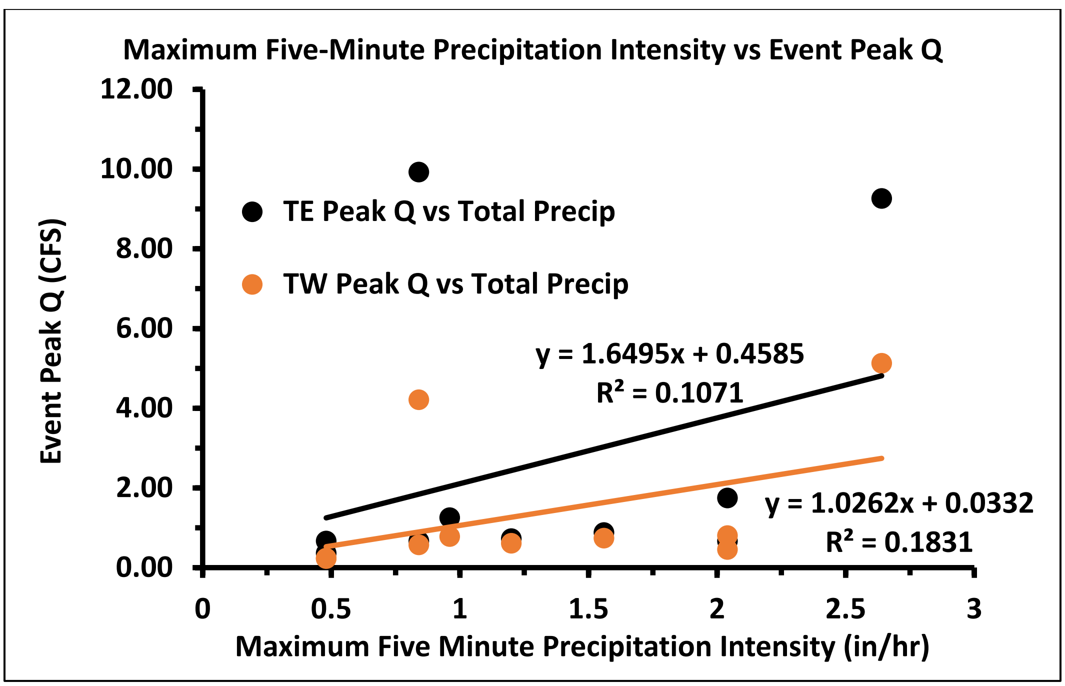


*Figure 17. Plot of Maximum Five-Minute Precipitation Intensity (in/hr) versus Event Total Runoff Volume (CF) for TE Control and TW Treatment Watersheds*

A similar analysis was completed for the event total runoff volumes to determine the relationships with event total precipitation and the maximum five-minute precipitation intensity. Figure 18 shows a strong relationship ($r^2$= 0.9055 and 0.8124) for the TE control and TW treatment watersheds, respectively. This figure also shows how the LID BMPs in TW treatment watershed decrease the total runoff volume during the study period through storage in rain barrels and rain gardens and uptake from biota for larger precipitation events (>1.5 inches).

Again, Figure 19 shows the weak relationship ($r^2$= 0.0603 and 0.042) for TE control and TW treatment watersheds, respectively between maximum five-minute precipitation intensity versus event total runoff volume.

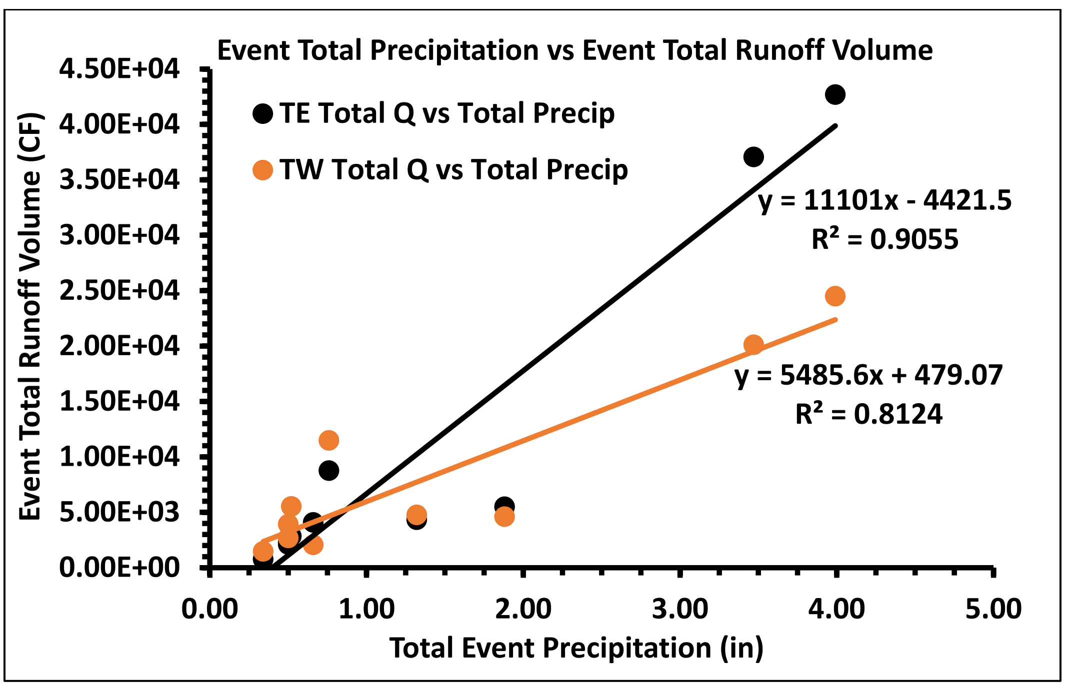


*Figure 18. Plot of Event Total Precipitation (in) versus Event Total Runoff Volume (CF) for TE Control and TW Treatment Watersheds*

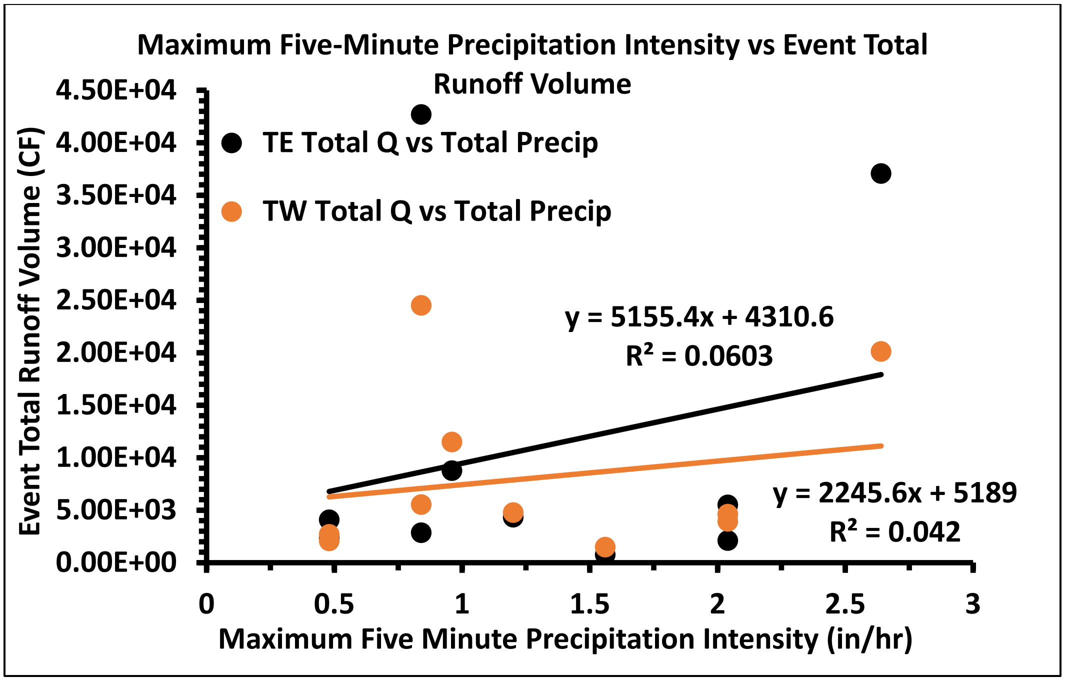


*Figure 19. Plot of Maximum Five-Minute Precipitation Intensity (in/hr) versus Event Total Runoff Volume (CF) for TE Control and TW Treatment Watersheds*

Similar conclusions can be drawn in that the maximum five-minute intensity had little impact on the total event runoff volume. This could be attributed to the fact that the average storm duration (e.g., initiation of flow to end of flow) for the study period was approximately 14 hours. Perhaps these prolonged storms do not need to have high intensity to produce significant quantities of stormwater. It is also important to note that the regression analysis in Figure 16-19 was skewed by outliers, which had much larger peak Q rates and total runoff volumes.

### 3.2.3 Additional Hydrologic Characteristics

This study evaluated several hydrologic parameters, including total runoff volume, peak discharge, runoff depth, runoff ratios, and lag time. Total runoff volume is the actual quantity of water that passed through the system for a given precipitation event. The peak discharge rate is the largest discharge rate for a given precipitation event measured every minute. Runoff depth was calculated by dividing the total runoff amount by the area of the watersheds. The runoff ratio is the runoff depth divided by the depth of precipitation for a given precipitation event. Lag time is the time difference between when measureable runoff starts on the control watershed in compared to the treatment watershed.

Summary statistics are presented in Table 18 and 19. TE control mean total runoff volume and peak Q were 11053.91 ± 4623.44 CF and 2.62 ± 1.11 CFS, respectively. TW treatment mean total runoff volume and peak Q were lower, 8126.18 ± 2412.03 CF and 1.38 ± 0.53 CFS, respectively (Table 18). For each watershed, the maximum values for total runoff volume and peak Q were larger in the TE control watershed than the TW treatment watershed (TE: 42699.07 CF and 9.93 CFS, respectively; TW: 24514.59 CF and 5.13 CFS, respectively). Mean runoff depth and lag time were also lower for the TW treatment watershed. Figure 20-23 graphically display the differences in total runoff volume (CF and percent), total Q (CFS and percent), and peak discharge (CFS and percent) between the watersheds.

*Table 18. Summary Hydrologic Statistics for Study Period; TE Control and TW Treatment Watersheds*

| | Total Runoff Volume (CF) | | Peak Q (CFS) | |
|---|---|---|---|---|
| | TE Control | TW Treatment | TE Control | TW Treatment |
| Mean | 11053.91 | 8126.18 | 2.62 | 1.38 |
| Median | 4217.47 | 4700.11 | 0.80 | 0.67 |
| Std. Dev. | 14620.59 | 7627.50 | 3.51 | 1.67 |
| Maximum | 42699.07 | 24514.59 | 9.93 | 5.13 |
| Minimum | 774.03 | 1482.07 | 0.36 | 0.22 |
| Sample Size | 10 | 10 | 10 | 10 |
| Std. Error | 4623.44 | 2412.03 | 1.11 | 0.53 |

*Table 19. Summary Hydrologic Statistics for Study Period; TE Control and TW Treatment Watersheds*

| | Runoff Depth (in) | | Runoff Ratio | | Lag Time (min) | |
|---|---|---|---|---|---|---|
| | TE Control | TW Treatment | TE Control | TW Treatment | TE Control | TW Treatment |
| Mean | 0.0223 | 0.0160 | 0.0125 | 0.0126 | 54.3333 | 15.1429 |
| Median | 0.0085 | 0.0093 | 0.0102 | 0.0111 | 8.0000 | 6.0000 |
| Std. Dev. | 0.0294 | 0.0152 | 0.0067 | 0.0073 | 71.2570 | 15.4774 |
| Maximum | 0.0860 | 0.0487 | 0.0233 | 0.0301 | 155.0000 | 45.0000 |
| Minimum | 0.0016 | 0.0029 | 0.0046 | 0.0049 | 0.0000 | 1.0000 |
| Sample Size | 10 | 10 | 10 | 10 | 10 | 10 |
| Std. Error | 0.0093 | 0.0048 | 0.0021 | 0.0023 | 22.5334 | 4.8944 |

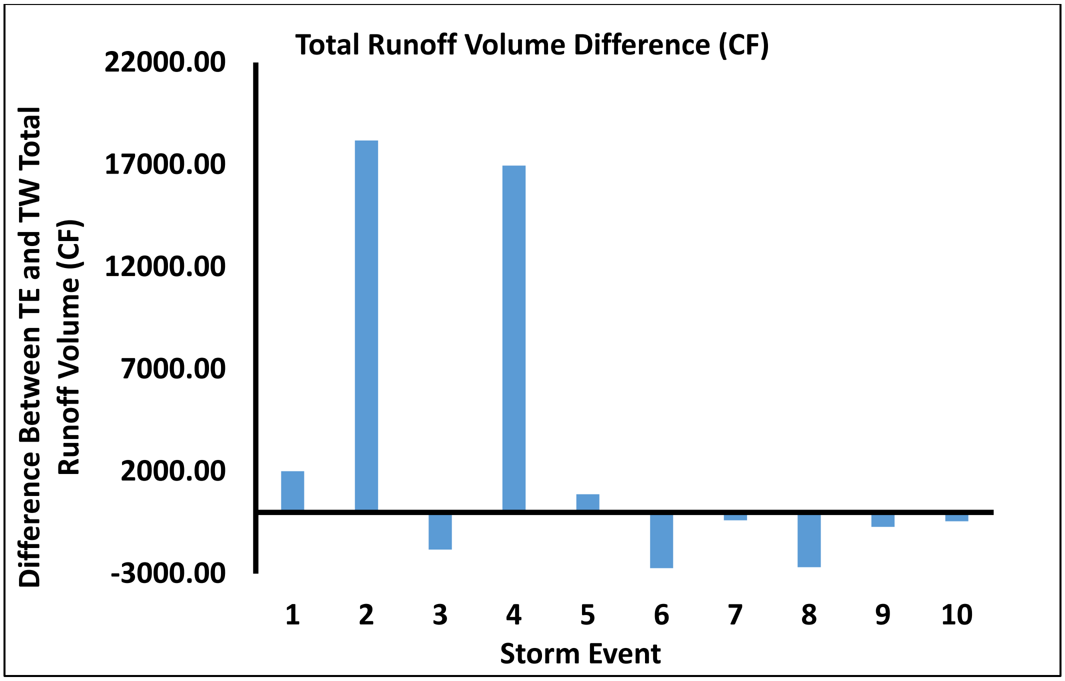


*Figure 20. Plot of the Difference between TE Control and TW Treatment Total Runoff Volume (CF) (n=10)*

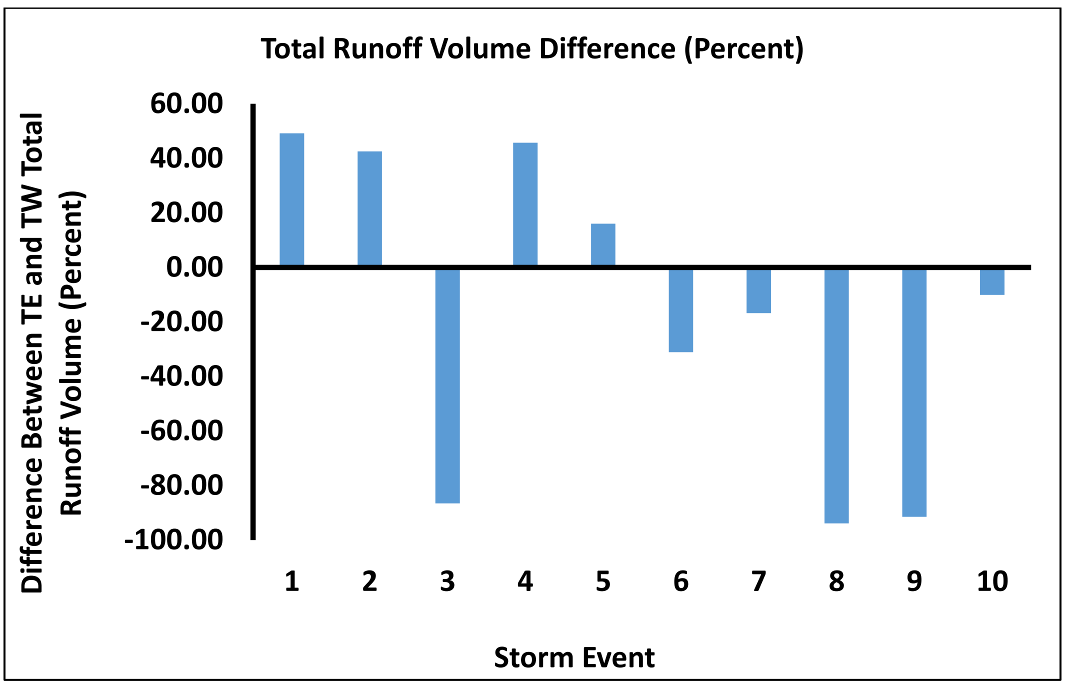


*Figure 21. Plot of the Difference between TE Control and TW Treatment Total Runoff Volume (%) (n=10)*

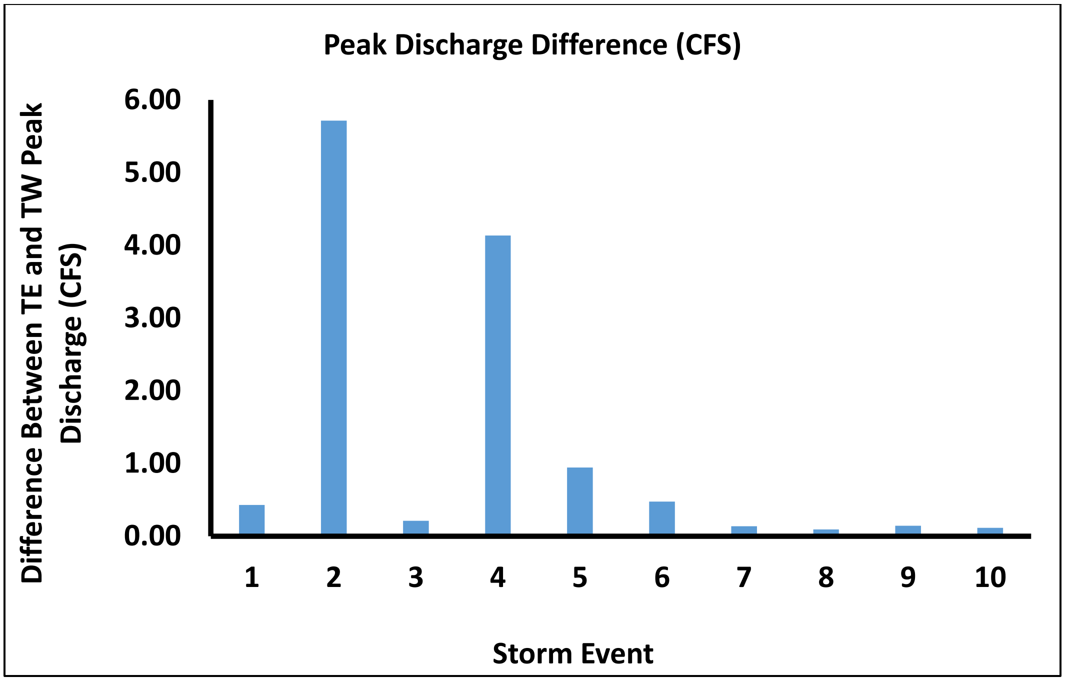


*Figure 22. Plot of the Difference between TE Control and TW Treatment Peak Discharge (CFS) (n=10)*

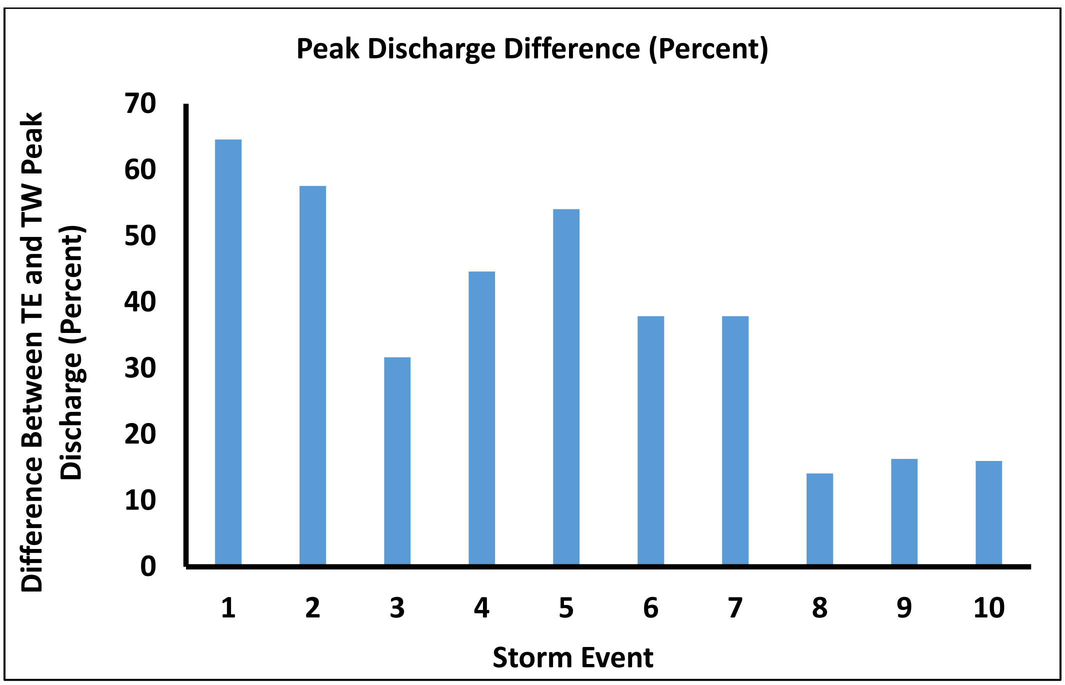


*Figure 23. Plot of the Difference between TE Control and TW Treatment Peak Discharge (%) (n=10)*

Figure 20 displays the difference in total runoff volume between TE control and TW treatment watersheds. There were four occurrences where TE control had a larger total runoff volume and six events where TW treatment had a larger total runoff volume. To quantify those differences, the percent differences were 86, 94, and 92 percent for events 3, 8, and 9, respectively (Figure 21). The 86 percent difference in event 3 represents 1824 CF of stormwater, and the 94 percent difference in event 8 represents 2680 CF, but the 92 percent difference in event 9 only represents 708 CF. For further analyses it was crucial to investigate both the volume and percent differences because less intense precipitation events skewed the percentages when a small difference in volume represented a large percentage of the total runoff volume. Figure 22 and Figure 23 display the differences in peak discharge (CFS) between TE control and TW treatment. Peak discharge was always lower for TW treatment than TE control with a maximum volumetric difference of 5.72 CFS or 57.59 percent (May 24$^{th}$ event) and maximum percent difference of 64.59 percent or 0.43 CFS (May 22$^{nd}$ event). Again, it was important to analyze both volumetric differences and percent differences because small precipitation events can skew the data.

Cheng et al (2005) found that peak flow or runoff volume decreased as the event rainfall runoff depth increased. The reasoning was likely due to ground saturation, as soil pore spaces filled with water there was less capacity for stormwater to be retained in the system. Results of the current study suggest the opposite phenomenon; Figure 24-27 show that as runoff depth increased so did the volumetric and percentile decreases in peak discharge and total runoff volume. The reason for this could be due to the fact that the Trailwoods system had a larger capacity for storage of stormwater and that increases in runoff depth provided more stormwater to be retained in the system. The volumetric comparisons (Figure 24 and Figure 26) have the strongest relationships for total runoff volume and peak Q with $r^2$= 0.7736 and 0.8847, respectively. Percentile comparisons (Figure 25 and Figure 27) have weaker relationships for total runoff volume and peak Q with $r^2$= 0.2711 and 0.1403, respectively. It is also important to note that the regression analysis in Figure 24-27 was skewed by outliers which had much larger decreased peak Q rates and total runoff volumes.

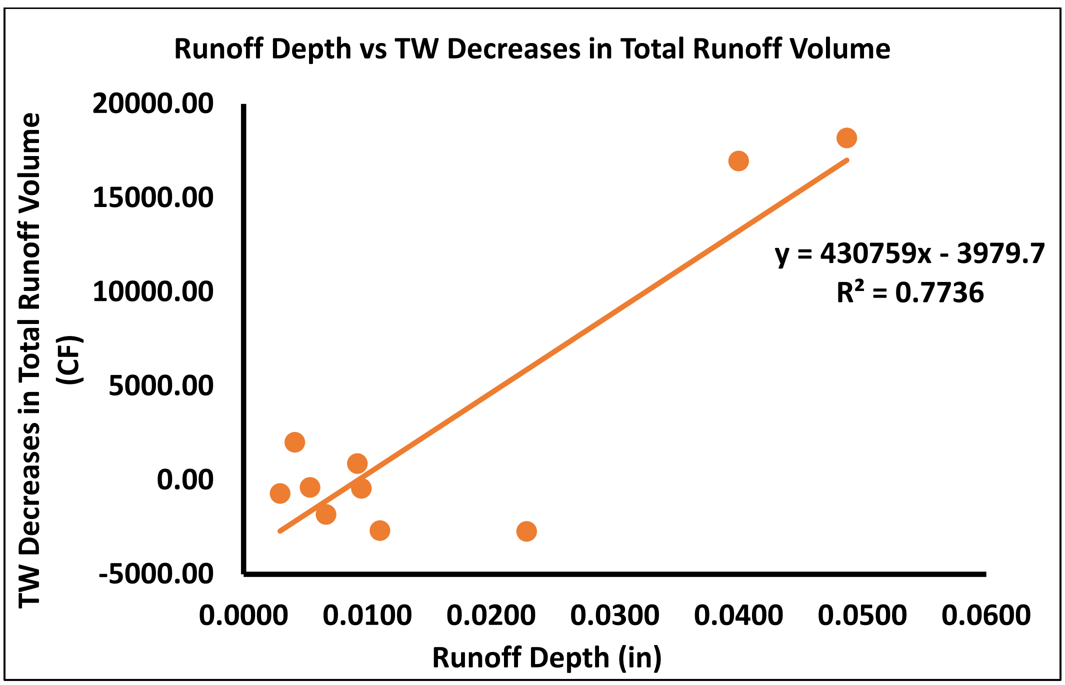


*Figure 24. Runoff Depth (in) versus TW Treatment Decreases in Total Runoff Volume (CF) (n=10)*

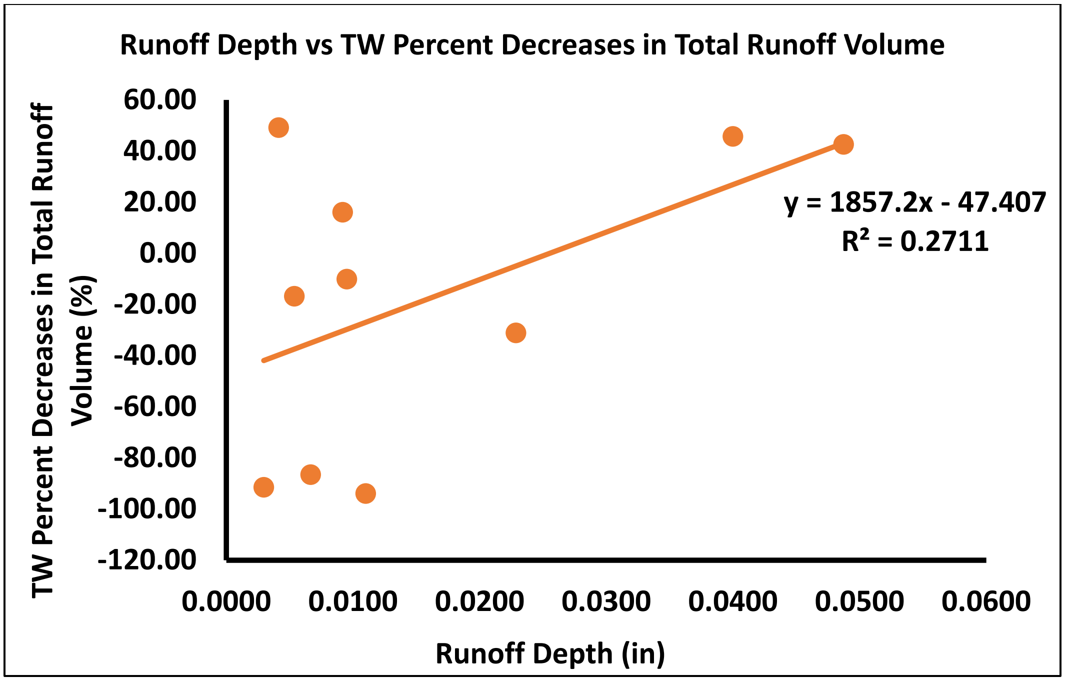


*Figure 25. Runoff Depth (in) versus TW Treatment Decreases in Total Runoff Volume (%) (n=10)*

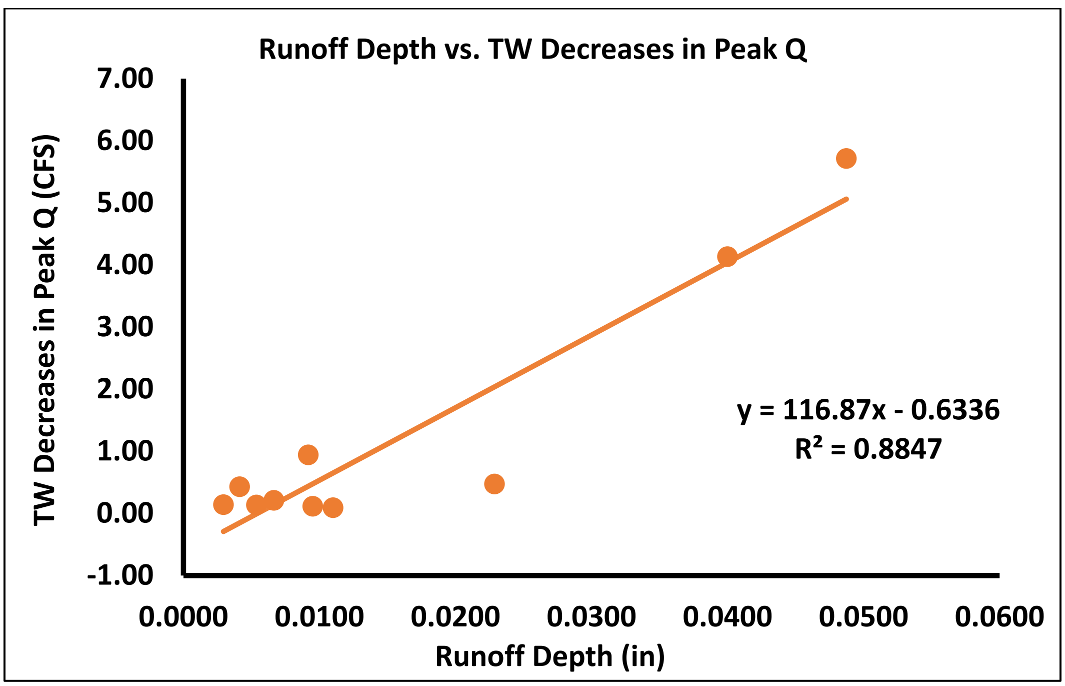


*Figure 26. Runoff Depth (in) versus TW Treatment Decreases in Peak Q (CFS) (n=10)*

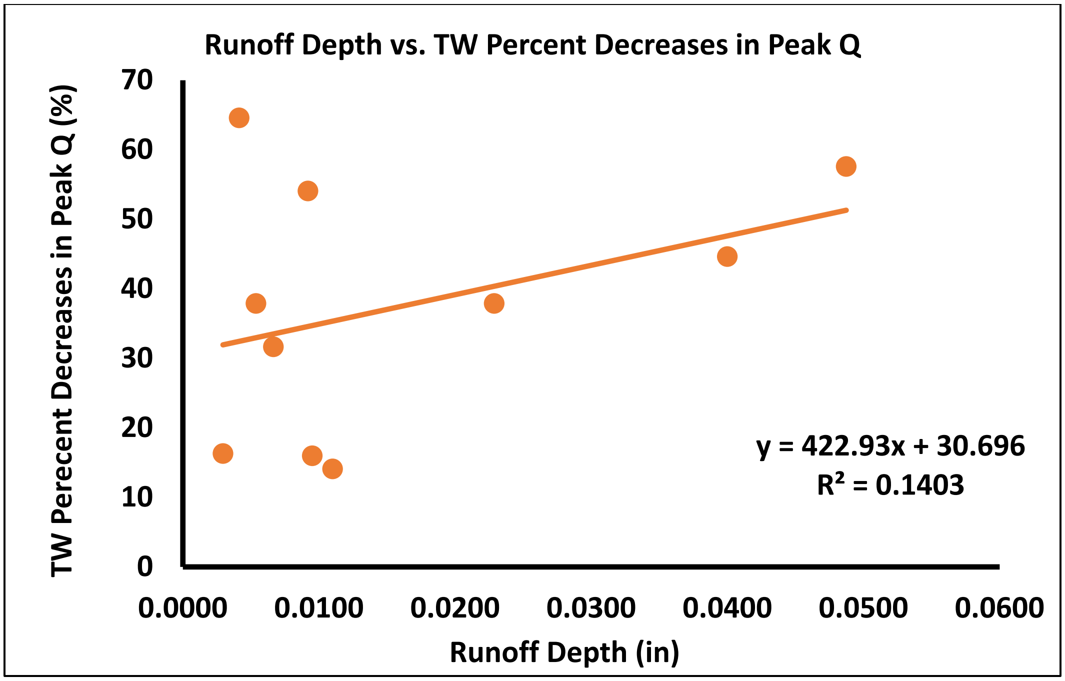


*Figure 27. Runoff Depth (in) versus TW Treatment Decreases in Peak Q (%) (n=10)*

## 3.3 Water Quality Data

Water quality data were collected for 10 precipitation events. Samples were analyzed for TSS, TN, $NO_3$-N, $NH_3$-N, TP, and TDP along with 15 metals (Al, As, Ca, Cd, Co, Cr, Cu, Fe, K, Mg, Mn, Na, Ni, Pb, and Zn). Only metals that had concentrations above detection limits and pose potential toxicity threats (Al, Cr, Cu, Ni, and Zn) will be the focus from here on.

### 3.3.1 Total Suspended Solids Comparison

The mean TSS concentrations for TW treatment watershed were 35.54 ± 9.8 mg/L while TE control had an average concentration of 69.24 ± 24.6 mg/L (n=7). Sample size was only seven because when there was inadequate volumes of stormwater collected priority was given to collection of nutrient sub-samples.

Student's t-tests did not reveal any significant differences for TSS between the watersheds even though there was an average percent difference of 48 percent (33.7 mg/L). Table 20 summarizes the data collected and provides simple statistics outlining the differences between the two basins. Results are supported by Cheng et al (2005) in that TSS concentrations were decreased by 15 percent in their study. Furthermore, a box and whisker plot was created to graphically display the differences in TSS concentrations between the two watersheds (Figure 28), which shows that the mean TSS concentrations for TW treatment were lower, as was the first quartile, while the maximum and third quartile for TE control were larger.

*Table 20. Summary Statistics for TSS Analysis from May to September 2015 (n=7)*

| | TSS Concentration (mg/L) | | |
|---|---|---|---|
| | TE | TW | Difference |
| 5/22/2015 | 165.91 | 34.40 | 131.51 |
| 5/24/2015 | 174.00 | 84.80 | 89.20 |
| 7/3/2015 | 44.40 | 35.60 | 8.80 |
| 7/21/2015 | 1.60 | 1.20 | 0.40 |
| 8/4/2015 | 14.80 | 27.60 | -12.80 |
| 8/19/2015 | 24.40 | 20.00 | 4.40 |
| 9/20/2015 | 46.80 | 58.00 | -11.20 |
| Mean | 69.24 | 35.54 | 33.70 |
| Median | 44.40 | 34.40 | 10.00 |
| Std. Dev. | 65.20 | 26.14 | 39.06 |
| Max | 174.00 | 84.80 | 89.20 |
| Min | 1.60 | 1.20 | 0.40 |
| Std. Error | 24.64 | 9.88 | 14.76 |
| p | 0.076 | --- | --- |

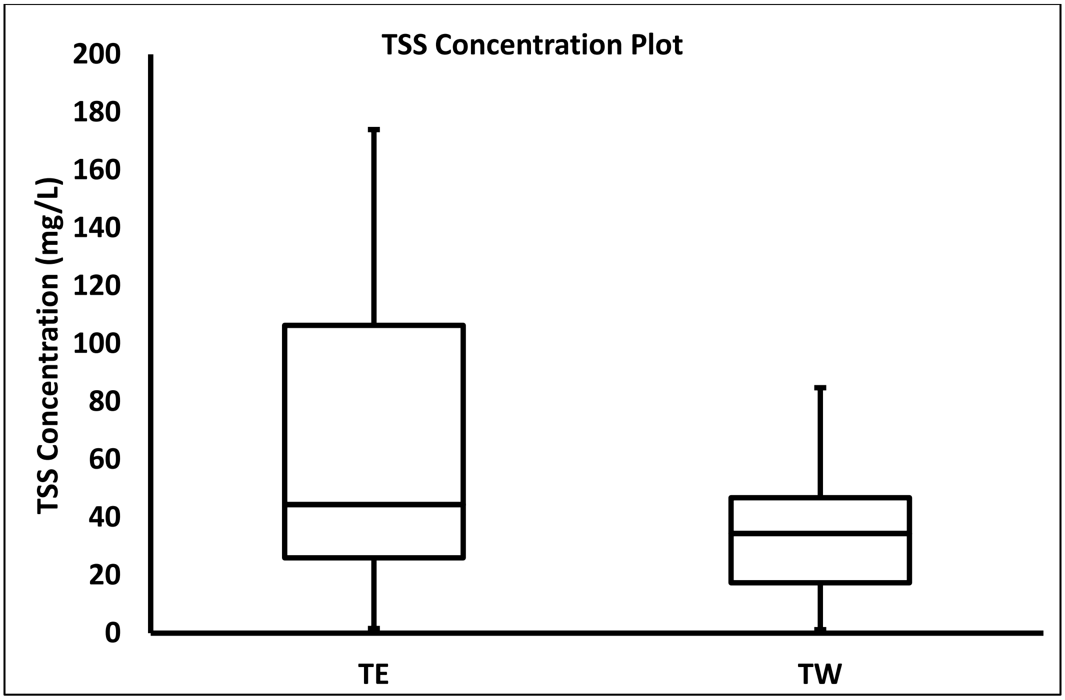


*Figure 28. Box and Whisker Plot for TSS Concentrations Collected from May to September 2015 (n=7)*

### 3.3.2 Nitrogen Compounds Comparison

The mean concentrations for TN, $NO_3$-N, and $NH_3$-N for TW treatment were 4.54 ± 0.87 mg/L, 0.39 ± 0.1 mg/L, and 1.49 ± 1.37 mg/L, respectively. TE control mean concentrations for TN, $NO_3$-N, and $NH_3$-N were 5.83 ± 1.31 mg/L, 1.08 ± 0.22 mg/L, and 1.71 ± 1.04 mg/L, respectively. Student's t-tests revealed that TW treatment $NO_3$-N concentrations were significantly lower than those of TE control (p value = 0.01). The average percent difference in $NO_3$-N concentrations was approximately 63 percent or 0.68 mg/L. The Spearman's rank correlation analysis did not highlight any significant correlation between nitrogen compound concentrations and any hydrologic parameters measured. These results are supported by a review of LID BMP effectiveness completed by EPA (2000), which found an average difference in $NO_3$-N concentrations of 41 percent when LID BMPs were present. Another study completed by Bedan and Clausen (2009) found that $NO_3$-N concentrations were different by more than two and a half times. The wide range of $NO_3$-N removal efficiencies could be attributed to the type of substrate used in rain gardens, the amount and type of LID BMPs implemented at the study site, the role of the biotic community, and contributing sources of $NO_3$-N (e.g., fertilization of lawns) among several other factors. Table 21-23 outline summary data and statistics for all nitrogen compounds analyzed for during the study period.

The significant decrease in $NO_3$-N could be attributed to the biogeochemical cycles occurring in the in the rain gardens. These systems are designed to remove $NH_3$-N in the top organic layer while $NO_3$-N remains mobile in the soil. The unique environment provided by the shallow water table produces an anaerobic oxidizing and reducing environment that promotes microbial communities to remove $NO_3$-N from the stormwater (NRC 2008). Given time, these systems may mature and become more prone to retaining other nitrogen compounds with greater efficiency.

Furthermore, a box and whisker plot was generated to graphically display the differences in concentrations between the various nitrogen compounds in the two watersheds. Figure 29-31 show that mean nitrogen compound concentrations were all less in TW treatment than in TE control. Also, the maximum concentrations for TN and $NO_3$-N were lower in TW treatment than TE control, but the maximum $NH_3$-N concentration was larger. Both TN and $NH_3$-N experienced peak concentrations during the 6/29/2015 event, likely due to a resident of the watershed fertilizing their lawn, because for the next event just four days later, concentrations were down to baseline for $NH_3$-N, but it took about a week for TN concentrations to decrease back to typical values.

*Table 21. Summary Data and Statistics for Total Nitrogen Analyzed for from May to September 2015, (n= 9 and 10) for TE Control and TW Treatment; N.S. = No sample, BDL = Below Detection Limits*

| | TN Concentration (mg/L) | | |
|---|---|---|---|
| | TE | TW | Difference |
| 5/22/2015 | 4.30 | 4.90 | -0.60 |
| 5/24/2015 | 3.70 | 2.70 | 1.00 |
| 6/29/2015 | 14.40 | 9.90 | 4.50 |
| 7/3/2015 | 3.60 | 9.10 | -5.50 |
| 7/7/2015 | N.S. | 3.30 | --- |
| 7/21/2015 | 3.20 | 2.20 | 1.00 |
| 8/4/2015 | 3.50 | 2.80 | 0.70 |
| 8/19/2015 | 3.60 | 2.30 | 1.30 |
| 9/8/2015 | 11.7 | 5.90 | 5.80 |
| 9/20/2015 | 4.50 | 2.30 | 2.20 |
| Mean | 5.83 | 4.54 | 1.29 |
| Median | 3.7 | 3.05 | 0.65 |
| Std. Dev. | 3.93 | 2.74 | 1.19 |
| Max | 14.4 | 9.9 | 4.50 |
| Min | 3.2 | 2.2 | 1.00 |
| Std. Error | 1.31 | 0.87 | 0.44 |
| p | 0.15 | --- | --- |

*Table 22. Summary Data and Statistics for Nitrate-Nitrogen Analyzed for from May to September 2015, (n= 9 and 10) for TE Control and TW Treatment; N.S. = No sample, BDL = Below Detection Limits*

| | $NO_3$-N Concentration (mg/L) | | |
|---|---|---|---|
| | TE | TW | Difference |
| 5/22/2015 | 1.57 | 0.61 | 0.96 |
| 5/24/2015 | 1.77 | 0.22 | 1.55 |
| 6/29/2015 | 2.23 | 0.16 | 2.07 |
| 7/3/2015 | 0.58 | 0.74 | -0.16 |
| 7/7/2015 | N.S. | 0.07 | --- |
| 7/21/2015 | 0.68 | 0.18 | 0.50 |
| 8/4/2015 | 0.79 | 0.49 | 0.30 |
| 8/19/2015 | 0.47 | 0.03 | 0.44 |
| 9/8/2015 | 1.49 | 0.99 | 0.50 |
| 9/20/2015 | 0.08 | BDL | --- |
| Mean | 1.08 | 0.39 | 0.69 |
| Median | 0.79 | 0.23 | 0.56 |
| Std. Dev. | 0.67 | 0.31 | 0.36 |
| Max | 2.2 | 0.99 | 1.21 |
| Min | 0.08 | 0.03 | 0.05 |
| Std. Error | 0.22 | 0.09 | 0.13 |
| p | 0.01 | --- | --- |

*Table 23. Summary Data and Statistics for Ammonia-Nitrogen Analyzed for from May to September 2015, (n= 9 and 10) for TE Control and TW Treatment; N.S. = No sample, BDL = Below Detection Limits*

| | $NH_3$-N Concentration (mg/L) | | |
|---|---|---|---|
| | TE | TW | Difference |
| 5/22/2015 | 0.20 | BDL | --- |
| 5/24/2015 | 0.30 | BDL | --- |
| 6/29/2015 | 10.4 | 14.5 | -4.10 |
| 7/3/2015 | 0.50 | 0.20 | 0.30 |
| 7/7/2015 | N.S. | 0.10 | --- |
| 7/21/2015 | 0.40 | 0.10 | 0.30 |
| 8/4/2015 | 0.30 | BDL | --- |
| 8/19/2015 | 1.5 | BDL | --- |
| 9/8/2015 | 1.8 | BDL | --- |
| 9/20/2015 | BDL | BDL | --- |
| Mean | 1.71 | 1.49 | 0.22 |
| Median | 0.40 | BDL | --- |
| Std. Dev. | 3.13 | 4.34 | -1.21 |
| Max | 10.4 | 14.5 | -4.10 |
| Min | BDL | BDL | --- |
| Std. Error | 1.04 | 1.37 | -0.33 |
| p | 0.45 | --- | --- |

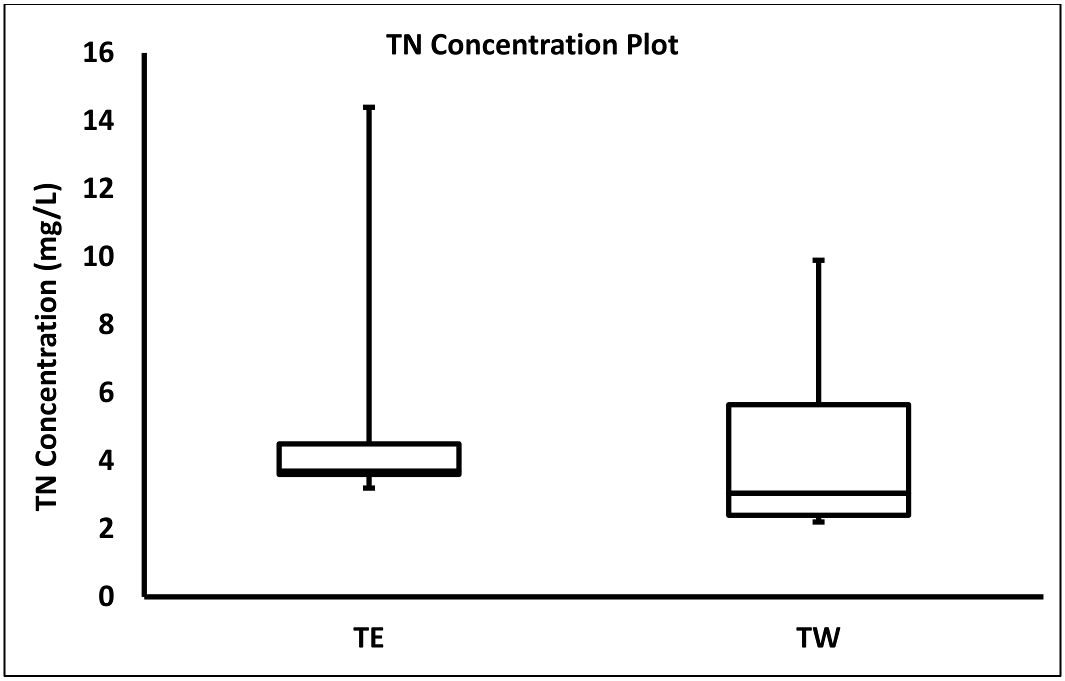


*Figure 29. Box and Whisker Plot for TN Concentrations Collected from May to September 2015 (n=9 and 10) for TE Control and TW Treatment, Respectively*

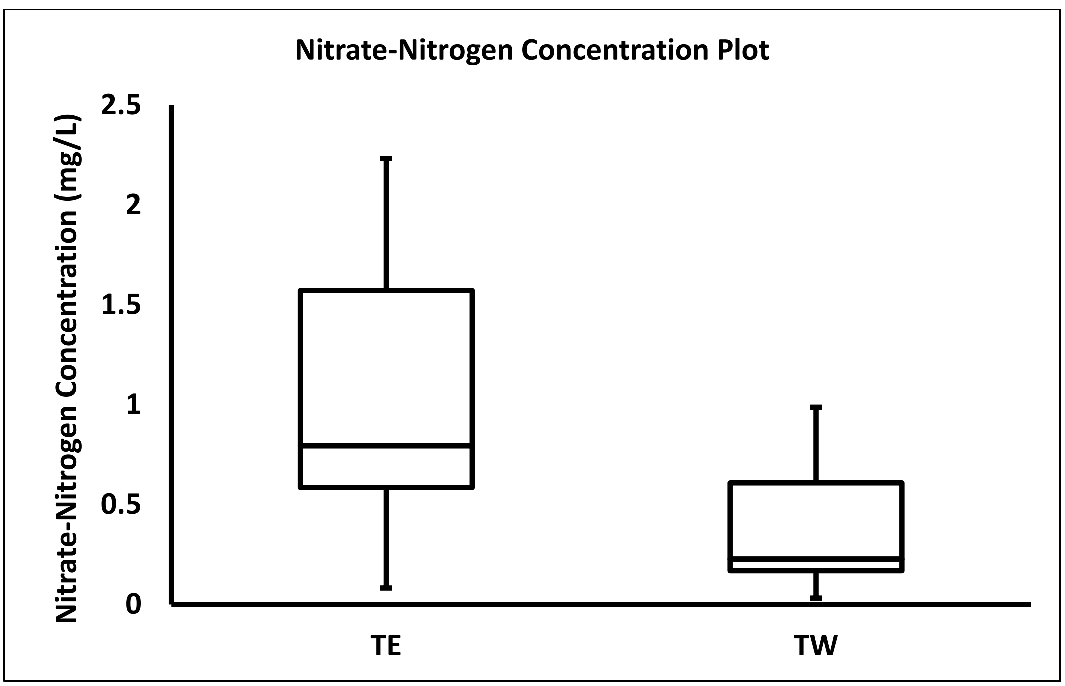


*Figure 30. Box and Whisker Plot for $NO_3$-N Concentrations Collected from May to September 2015 (n=9 and 10) for TE Control and TW Treatment, Respectively*

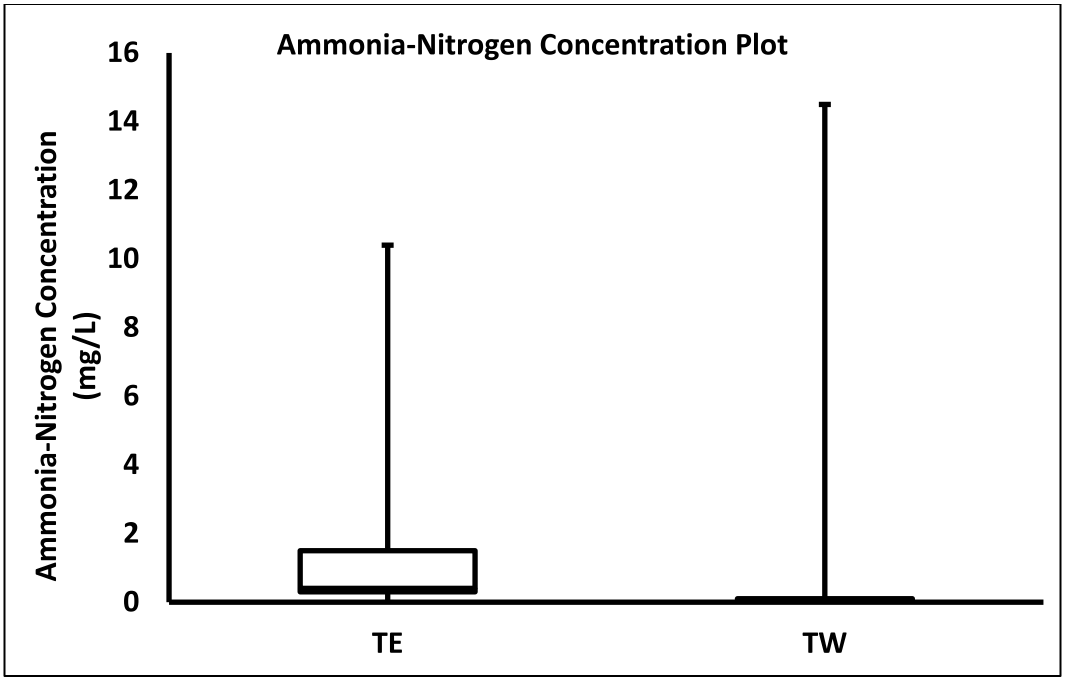


*Figure 31. Box and Whisker Plot for $NH_3$-N Concentrations Collected from May to September 2015 (n=9 and 10) for TE Control and TW Treatment, Respectively*

### 3.3.3 Phosphorus Compounds Comparison

The mean concentrations for TP were 0.19 and 0.36 mg/L for TE control and TW treatment, respectively, which showed increased export of TP concentrations from TW treatment by approximately 47 percent. Mean TDP concentrations were 0.05 and 0.07 mg/L for TE control and TW treatment, respectively, again showing a net difference of approximately 29 percent. Furthermore, the maximum concentrations for both TP and TDP were higher leaving TW treatment than TE control, and overall concentrations for TP were higher leaving TW treatment 70 percent of the time and TDP concentrations were higher leaving TW treatment 60 percent of the time. However, Student's t-tests did not reveal any statistical significance between the two watersheds (p value = 0.06 and 0.08) for TP and TDP, respectively (Table 24 and 25).

*Table 24. Summary Data and Statistics for Total Phosphorus Analyzed for from May to September 2015, (n= 9 and 10) for TE Control and TW Treatment; N.S. = No sample*

| | TP Concentration (mg/L) | | |
|---|---|---|---|
| | TE | TW | Difference |
| 5/22/2015 | 0.23 | 0.31 | -0.08 |
| 5/24/2015 | 0.20 | 0.71 | -0.51 |
| 6/29/2015 | 0.26 | 1.02 | -0.76 |
| 7/3/2015 | 0.11 | 0.50 | -0.39 |
| 7/7/2015 | N.S. | 0.39 | --- |
| 7/21/2015 | BDL | 0.06 | --- |
| 8/4/2015 | 0.06 | 0.15 | -0.09 |
| 8/19/2015 | 0.04 | 0.01 | 0.03 |
| 9/8/2015 | 0.54 | 0.33 | 0.21 |
| 9/20/2015 | 0.04 | 0.09 | -0.05 |
| Mean | 0.19 | 0.36 | -0.17 |
| Median | 0.16 | 0.32 | -0.16 |
| Std. Dev. | 0.16 | 0.30 | -0.14 |
| Max | 0.54 | 1.02 | -0.48 |
| Min | 0.04 | 0.01 | 0.03 |
| Std. Error | 0.05 | 0.10 | -0.05 |
| p | 0.06 | --- | --- |

*Table 25. Summary Data and Statistics for Total Dissolved Phosphorus Analyzed for from May to September 2015, (n= 9 and 10) for TE Control and TW Treatment; N.S. = No sample*

| | TDP Concentration (mg/L) | | |
|---|---|---|---|
| | TE | TW | Difference |
| 5/22/2015 | 0.05 | 0.08 | -0.03 |
| 5/24/2015 | 0.04 | 0.13 | -0.09 |
| 6/29/2015 | 0.10 | 0.16 | -0.06 |
| 7/3/2015 | 0.04 | 0.10 | -0.06 |
| 7/7/2015 | N.S. | 0.09 | --- |
| 7/21/2015 | 0.02 | 0.03 | 0.01 |
| 8/4/2015 | 0.03 | 0.03 | 0.00 |
| 8/19/2015 | 0.03 | 0.03 | 0.00 |
| 9/8/2015 | 0.07 | 0.02 | 0.05 |
| 9/20/2015 | 0.02 | 0.03 | -0.01 |
| Mean | 0.05 | 0.07 | -0.02 |
| Median | 0.04 | 0.05 | -0.01 |
| Std. Dev. | 0.03 | 0.05 | -0.02 |
| Max | 0.1 | 0.16 | -0.06 |
| Min | 0.02 | 0.02 | 0.00 |
| Std. Error | 0.009 | 0.01 | -0.001 |
| p | 0.08 | --- | --- |

These results are supported by a series of other studies including EPA (2000), NRC (2008), Bedan and Clausen (2009), Waldron et al. (2010), and Dhalla and Zimmer (2010). Bedan and Clausen (2009) reported that TP concentrations increased significantly after construction of the LID basin, similar to the report of Waldron et al. (2010), who found the highest phosphorus concentrations were measured after LID implementation. The review of LID BMP case studies by EPA (2000) revealed that when a grassed swale was used in the LID BMP design there were negative removal efficiencies for TP. Similarly, Dhalla and Zimmer (2010) reported an average of a nine percent increase of TP in several LID BMP case studies. These increased phosphorus concentrations were discussed by NRC (2008), who stated phosphorus removal was directly related to the amount of phosphorus in the original rain garden substrate, and was supported by Waldron et al. (2010) who stated relatively high nutrient concentrations can be linked to phosphorus initially present in the rain garden growing media. The primary removal mechanism for phosphorus compounds was most likely sorption to the clay component of the engineered substrate used in the rain gardens. Selection of a clay that has high capacity for phosphorus sorption is important to achieve sustained phosphorus removal and retention (Arias et al. 2001). Clays with high calcium contents promote higher removal of phosphorus compounds due to the formation of insoluble calcium phosphates, which is exaggerated in net alkaline stormwater (Arias et al. 2001).

In an attempt to determine why phosphorus concentrations were higher in TW treatment than TE control effluents, investigation into the engineered growth media used in the Trailwoods West rain gardens site was completed. The engineered growth media used in the rain gardens was 70 percent expanded clay, 20 percent sand, and 10 percent compost by volume. Prior to implementation, an ASTM 3977c (2002) (vertical beam test) analysis for determination of sediment concentrations in water was completed on the rain garden growth media by Soil Control Lab Watsonville, CA. The physical parameters of the media were 75 percent light expanded clay aggregate (LECA), 15 percent sand, and 10 percent organic matter, with a dry bulk density of 5.08 grams per cubic centimeter. Results of the vertical beam test included a 93.9, 9.8, and > 45 percent decreases in TSS, TP, and total metals (Cu, Zn, Fe, and Mn) concentrations, respectively. However, there were percent increases in TN and $NO_3$-N of 51.3 and 1.8 percent, respectively. Unfortunately, specifics about the compost nutrient composition were unavailable, but some assumptions based on Liu et al. (2014) were made. Liu et al. (2014) suggested, that rain garden engineered media should contain, < 10 percent fines, 3-5 percent total organic carbon, a source of Al or Ca for phosphorus sorption, and be placed to a depth of at least two feet. With that said, it seemed the engineered growth media used at the Trailwoods study site was lacking at least one of the parameters suggested by Liu e al. (2014). This could have been either the percent total organic carbon or source of Al or Ca for phosphorus sorption, which resulted in higher phosphorus concentrations measured at TW treatment.

As a part of its nature, compost is commonly used as a fertilizer (e.g., a nitrogen and phosphorus source), and it is thought that leaching from the rain garden media may have caused the excess concentrations of phosphorus compounds to leave the TW treatment watershed compared to the TE control watershed. This issue is especially problematic in the Lake Thunderbird watershed because, according to OCC (2010), the lake already has significant water quality degradation issues stemming from non-point sources resulting in cultural eutrophication. Increased phosphorus exports are the direct opposite function that LID BMPs are designed to perform, so further analysis is required to mitigate this issue so that LID BMPs can successfully be implemented as an alternative to traditional stormwater management. Further analysis could include investigation of various types of rain gardens medias and associated effluent nutrient concentrations or to introduce inorganic additives such as fly that would serve as a sorption surface for phosphorus compounds.

Another box and whisker plot analysis (Figure 32 and Figure 33) was completed for the phosphorus compounds measured during the study period. Notice how the median and maximum values for both TP and TDP are higher for TW treatment watershed. Also the range of the concentrations within the TW treatment watershed for both TP and TDP were larger. These facts suggest that the increased phosphorus compound concentrations were not sourced by precipitation or atmospheric deposition, rather, it was more likely from a source within the TW treatment watershed itself, such as leaching from the engineered substrate.

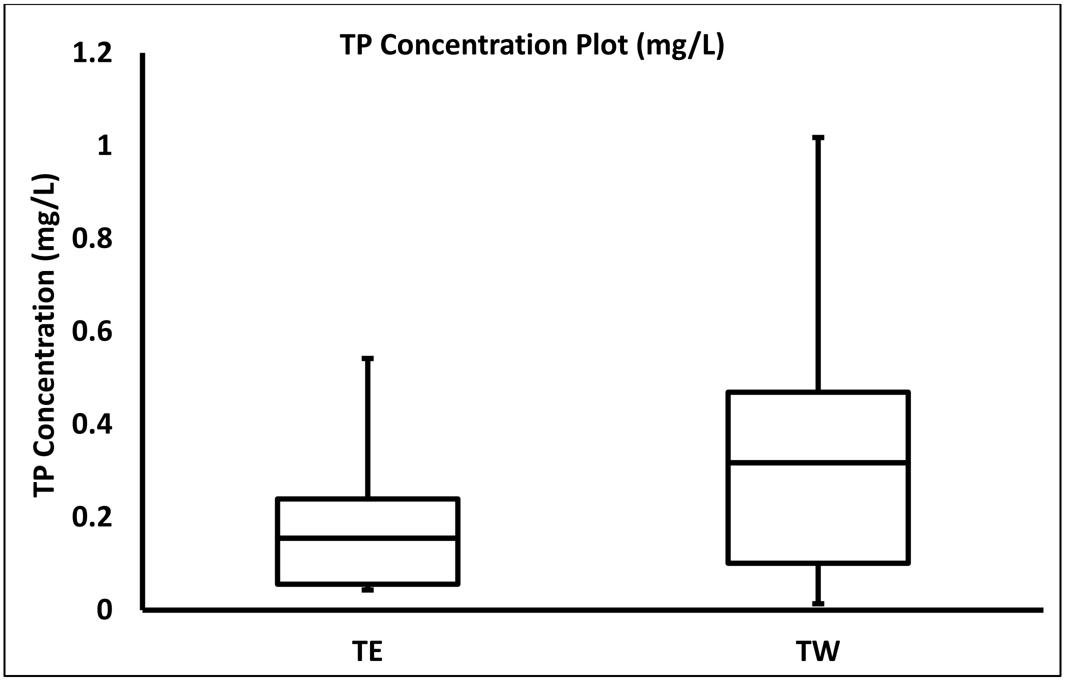


*Figure 32. Box and Whisker Plot for TP Concentrations Collected from May to September 2015 (n=9 and 10) for TE Control and TW Treatment, Respectively*

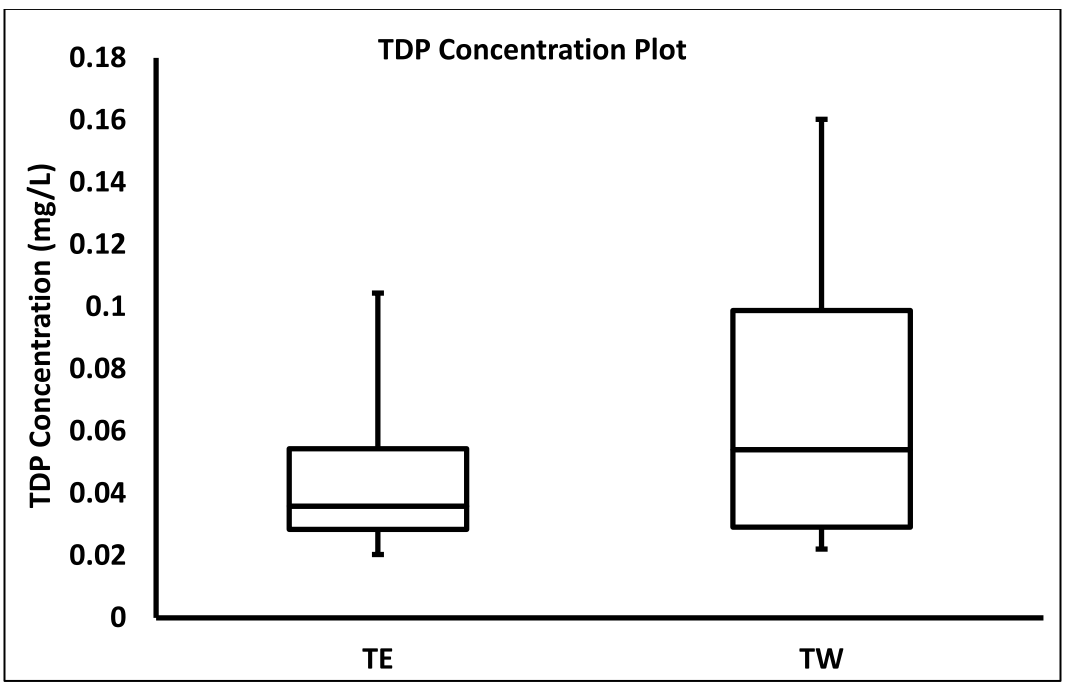


*Figure 33. Box and Whisker Plot for TDP Concentrations Collected from May to September 2015 (n=9 and 10) for TE Control and TW Treatment, Respectively*

### 3.3.4 Total Metals Comparison

Total metal analyses were completed on the collected stormwater. However, samples were generated for only eight events due to inadequate stormwater volumes. These samples were analyzed for a suite of metals (Al, As, Ca, Cd, Co, Cr, Cu, Fe, K, Mg, Mn, Na, Ni, Pb, and Zn) even though metal contamination was not the primary focus in the Lake Thunderbird watershed. Metals are common in stormwater runoff and it was important to quantify all aspects of treatment performed by the LID BMPs (NRC 2008). For the most part, neither TE control nor TW treatment watershed effluents had measureable concentrations of As, Cd, or Co, however there were significant differences in Ca, Cu, and K concentrations (p value = 0.005, 0.004, and 0.017), respectively. Table 26-30, provide summary data and statistics for all events and constituents.

*Table 26. Summary Data and Statistics for Total Metal Concentrations Analyzed for from May to September 2015, (n= 8), BDL = Below Detection Limits, Diff. = Difference; TE Control and TW Treatment*

| | Total Metal Concentrations (mg/L) | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | Al | | | As | | | Ca | | |
| | TE | TW | Diff. | TE | TW | Diff. | TE | TW | Diff. |
| 5/22 | 2.909 | 1.916 | 0.993 | BDL | BDL | --- | 9.441 | 19.734 | -10.293 |
| 5/24 | 6.142 | 1.894 | 4.230 | BDL | BDL | --- | 10.964 | 15.842 | -4.878 |
| 7/3 | 0.782 | 1.756 | -0.974 | BDL | BDL | --- | 8.544 | 9.181 | -0.637 |
| 7/21 | 0.559 | 0.218 | 0.341 | BDL | BDL | --- | 9.608 | 12.821 | -3.213 |
| 8/4 | 0.490 | 0.545 | -0.055 | BDL | BDL | --- | 10.774 | 14.040 | -3.266 |
| 8/19 | 0.724 | 0.382 | 0.342 | BDL | BDL | --- | 9.993 | 11.979 | -1.986 |
| 9/8 | 1.552 | 2.004 | -0.452 | BDL | BDL | --- | 21.985 | 28.316 | -6.331 |
| 9/20 | 0.523 | 0.527 | -0.004 | BDL | BDL | --- | 8.876 | 10.071 | -1.195 |
| Mean | 1.71 | 1.16 | 0.550 | --- | --- | --- | 11.27 | 15.25 | -3.980 |
| Median | 0.75 | 1.15 | -0.400 | --- | --- | --- | 9.80 | 13.43 | -3.630 |
| Std. Dev. | 1.84 | 0.75 | 1.090 | --- | --- | --- | 4.12 | 5.84 | -1.720 |
| Max | 6.14 | 2.00 | 4.140 | --- | --- | --- | 21.98 | 28.32 | -6.340 |
| Min | 0.49 | 0.22 | 0.270 | --- | --- | --- | 8.54 | 9.18 | -0.640 |
| Std. Error | 0.65 | 0.26 | 0.390 | --- | --- | --- | 1.46 | 2.06 | -0.600 |
| p | 0.18 | --- | --- | --- | --- | --- | 0.005 | --- | --- |

*Table 27. Summary Data and Statistics for Total Metal Concentrations Analyzed for from May to September 2015, (n= 8), BDL = Below Detection Limit, Diff. = Difference; TE Control and TW Treatment*

| Total Metal Concentrations (mg/L) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | Cd | | | Co | | | Cr | |
| | TE | TW | Diff. | TE | TW | Diff. | TE | TW | Diff. |
| 5/22 | BDL | BDL | --- | BDL | BDL | --- | 0.003 | 0.003 | 0.000 |
| 5/24 | BDL | BDL | --- | BDL | BDL | --- | 0.006 | 0.003 | 0.003 |
| 7/3 | BDL | BDL | --- | BDL | BDL | --- | 0.003 | 0.003 | 0.000 |
| 7/21 | 0.001 | BDL | --- | BDL | BDL | --- | 0.003 | 0.003 | 0.000 |
| 8/4 | BDL | BDL | --- | 0.001 | 0.001 | 0.000 | 0.003 | 0.003 | 0.000 |
| 8/19 | BDL | BDL | --- | BDL | BDL | --- | 0.003 | 0.002 | 0.001 |
| 9/8 | 0.001 | BDL | --- | BDL | BDL | --- | 0.007 | 0.004 | 0.003 |
| 9/20 | BDL | BDL | --- | BDL | BDL | --- | 0.002 | 0.003 | -0.001 |
| Mean | 0.001 | --- | --- | 0.001 | 0.001 | 0.000 | 0.004 | 0.003 | 0.001 |
| Median | 0.001 | --- | --- | 0.001 | 0.001 | 0.000 | 0.003 | 0.003 | 0.000 |
| Std. Dev. | 0 | --- | --- | 0 | 0 | --- | 0.002 | 0.000 | 0.002 |
| Max | 0.001 | --- | --- | 0.001 | 0.001 | 0.000 | 0.007 | 0.004 | 0.003 |
| Min | 0.001 | --- | --- | 0.001 | 0.001 | 0.000 | 0.002 | 0.002 | 0.000 |
| Std. Error | 0.00 | --- | --- | --- | --- | --- | 0.001 | 0.000 | 0.001 |
| p | 0 | --- | --- | --- | --- | --- | 0.080 | --- | --- |

*Table 28. Summary Data and Statistics for Total Metal Concentrations Analyzed for from May to September 2015, (n= 8), BDL = Below Detection Limits, Diff. = Difference; TE Control and TW Treatment*

| Total Metal Concentrations (mg/L) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | Cu | | | Fe | | | K | |
| | TE | TW | Diff. | TE | TW | Diff. | TE | TW | Diff. |
| 5/22 | 0.007 | 0.006 | 0.001 | 1.659 | 0.800 | 0.859 | 2.318 | 2.431 | -0.113 |
| 5/24 | 0.008 | 0.006 | 0.002 | 4.160 | 1.117 | 3.043 | 3.224 | 2.468 | 0.756 |
| 7/3 | 0.008 | 0.006 | 0.002 | 0.558 | 1.253 | -0.695 | 2.115 | 1.885 | 0.230 |
| 7/21 | 0.008 | 0.003 | 0.005 | 0.457 | 0.222 | 0.235 | 1.912 | 1.534 | 0.378 |
| 8/4 | 0.008 | 0.005 | 0.003 | 0.440 | 0.552 | -0.112 | 1.916 | 1.798 | 0.188 |
| 8/19 | 0.008 | 0.003 | 0.005 | 0.769 | 0.437 | 0.332 | 1.621 | 1.506 | 0.115 |
| 9/8 | 0.032 | 0.021 | 0.011 | 1.479 | 2.581 | -1.102 | 3.796 | 3.684 | 0.112 |
| 9/20 | 0.006 | 0.003 | 0.003 | 0.468 | 0.580 | -0.112 | 2.218 | 1.487 | 0.731 |
| Mean | 0.011 | 0.006 | 0.005 | 1.249 | 0.943 | 0.306 | 2.390 | 2.099 | 0.291 |
| Median | 0.008 | 0.005 | 0.003 | 0.663 | 0.690 | -0.027 | 2.166 | 1.841 | 0.325 |
| Std. Dev. | 0.008 | 0.006 | 0.002 | 1.188 | 0.697 | 0.491 | 0.692 | 0.702 | -0.010 |
| Max | 0.032 | 0.021 | 0.011 | 4.160 | 2.581 | 1.579 | 3.796 | 3.684 | 0.112 |
| Min | 0.006 | 0.003 | 0.003 | 0.440 | 0.222 | 0.218 | 1.621 | 1.487 | 0.134 |
| Std. Error | 0.003 | 0.002 | 0.001 | 0.420 | 0.247 | 0.173 | 0.247 | 0.248 | -0.001 |
| p | 0.004 | --- | --- | 0.257 | --- | --- | 0.017 | --- | --- |

*Table 29. Summary Data and Statistics for Total Metal Concentrations Analyzed for from May to September 2015, (n= 8), BDL = Below Detection Limit, Diff. = Difference; TE Control and TW Treatment*

| | Total Metal Concentrations (mg/L) | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | Mg | | | Mn | | | Na | | |
| | TE | TW | Diff. | TE | TW | Diff. | TE | TW | Diff. |
| 5/22 | 1.903 | 3.864 | -1.961 | 0.030 | 0.018 | 0.012 | 2.582 | 4.058 | -1.476 |
| 5/24 | 3.404 | 3.239 | 0.165 | 0.073 | 0.015 | 0.058 | 3.314 | 2.280 | 1.034 |
| 7/3 | 1.615 | 1.892 | -0.277 | 0.012 | 0.028 | -0.016 | 2.093 | 1.085 | 1.008 |
| 7/21 | 1.442 | 1.559 | -0.117 | 0.013 | 0.011 | 0.002 | 2.665 | 2.071 | 0.594 |
| 8/4 | 1.381 | 1.164 | 0.217 | 0.018 | 0.024 | -0.006 | 2.147 | 1.210 | 0.937 |
| 8/19 | 1.143 | 0.751 | 0.392 | 0.031 | 0.014 | 0.017 | 1.366 | 0.879 | 0.487 |
| 9/8 | 2.709 | 2.156 | 0.553 | 0.054 | 0.115 | -0.061 | 3.401 | 1.211 | 2.190 |
| 9/20 | 1.168 | 0.999 | 0.169 | 0.012 | 0.020 | -0.008 | 1.780 | 1.298 | 0.482 |
| Mean | 1.846 | 1.953 | -0.107 | 0.030 | 0.031 | -0.001 | 2.419 | 1.761 | 0.658 |
| Median | 1.529 | 1.726 | -0.197 | 0.024 | 0.019 | 0.005 | 2.364 | 1.255 | 1.109 |
| Std. Dev. | 0.755 | 1.030 | -0.275 | 0.021 | 0.032 | -0.011 | 0.666 | 0.981 | -0.315 |
| Max | 3.404 | 3.864 | -0.46 | 0.073 | 0.115 | -0.042 | 3.401 | 4.058 | -0.657 |
| Min | 1.143 | 0.751 | 0.392 | 0.012 | 0.011 | 0.001 | 1.366 | 0.879 | 0.487 |
| Std. Error | 0.267 | 0.364 | -0.097 | 0.007 | 0.011 | -0.004 | 0.236 | 0.347 | -0.111 |
| p | 0.357 | | --- | 0.491 | | --- | 0.056 | | --- |

*Table 30. Summary Data and Statistics for Total Metal Concentrations Analyzed for from May to September 2015, (n= 8), BDL = Below Detection Limits, Diff. = Difference; TE Control and TW Treatment*

| | Total Metal Concentrations (mg/L) | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | Ni | | | Pb | | | Zn | | |
| | TE | TW | Diff. | TE | TW | Diff. | TE | TW | Diff. |
| 5/22 | 0.015 | 0.009 | 0.006 | BDL | BDL | --- | 0.012 | 0.011 | 0.001 |
| 5/24 | 0.036 | 0.011 | 0.025 | 0.022 | BDL | --- | 0.019 | 0.006 | 0.013 |
| 7/3 | 0.006 | 0.011 | -0.005 | BDL | BDL | --- | 0.006 | 0.009 | -0.003 |
| 7/21 | 0.006 | BDL | --- | BDL | BDL | --- | 0.006 | 0.006 | 0.000 |
| 8/4 | 0.006 | 0.005 | 0.001 | BDL | BDL | --- | 0.008 | 0.013 | -0.005 |
| 8/19 | 0.006 | 0.005 | 0.001 | BDL | BDL | --- | 0.018 | 0.010 | 0.008 |
| 9/8 | 0.011 | 0.012 | -0.001 | 0.021 | BDL | --- | 0.043 | 0.082 | -0.039 |
| 9/20 | BDL | BDL | --- | BDL | BDL | --- | 0.007 | 0.012 | -0.005 |
| Mean | 0.012 | 0.009 | 0.003 | 0.021 | --- | --- | 0.015 | 0.018 | -0.003 |
| Median | 0.006 | 0.010 | -0.004 | 0.021 | --- | --- | 0.010 | 0.010 | 0.000 |
| Std. Dev. | 0.010 | 0.003 | 0.007 | 0.00 | --- | --- | 0.012 | 0.024 | -0.012 |
| Max | 0.036 | 0.012 | 0.024 | 0.022 | --- | --- | 0.043 | 0.082 | -0.039 |
| Min | 0.006 | 0.005 | 0.001 | 0.021 | --- | --- | 0.006 | 0.006 | 0.000 |
| Std. Error | 0.004 | 0.001 | 0.003 | 0.000 | --- | --- | 0.004 | 0.009 | -0.005 |
| p | 0.180 | | --- | --- | --- | --- | 0.277 | --- | --- |

Figure 34-38, display box and whisker analysis for metals with measureable and recurring concentrations. Aside from a single spike in Zn concentrations in TW treatment, maximum metal concentrations were lower in TW treatment than TE control. This was most likely due to several physical, biogeochemical, and microbial processes occurring throughout the engineered substrate in the rain gardens. Filtration allowed for solids to settle and be removed from the stormwater matrix, perhaps with sorbed metals. After filtration, metal carbonates may have formed which decreased concentrations of Zn and Mn. Further retention of trace metals may have been completed when metal sulfides were formed via bacterial reducing mechanisms (Nairn et al. 2010).

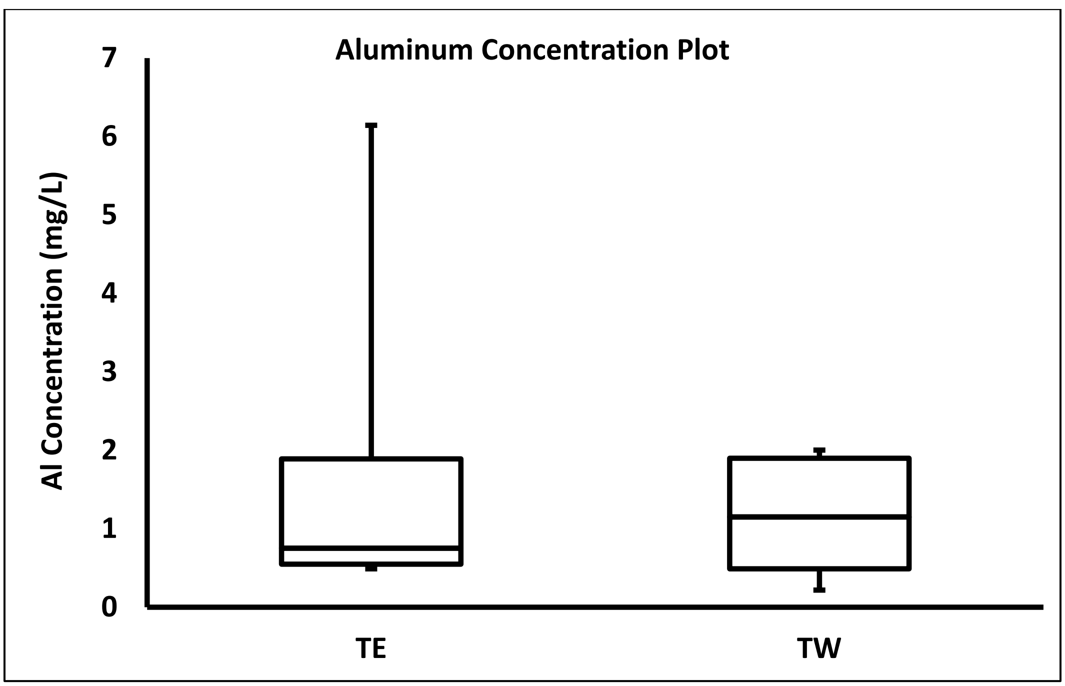


*Figure 34. Box and Whisker Plot for Al Concentrations Collected from May to September 2015 (n=8)*

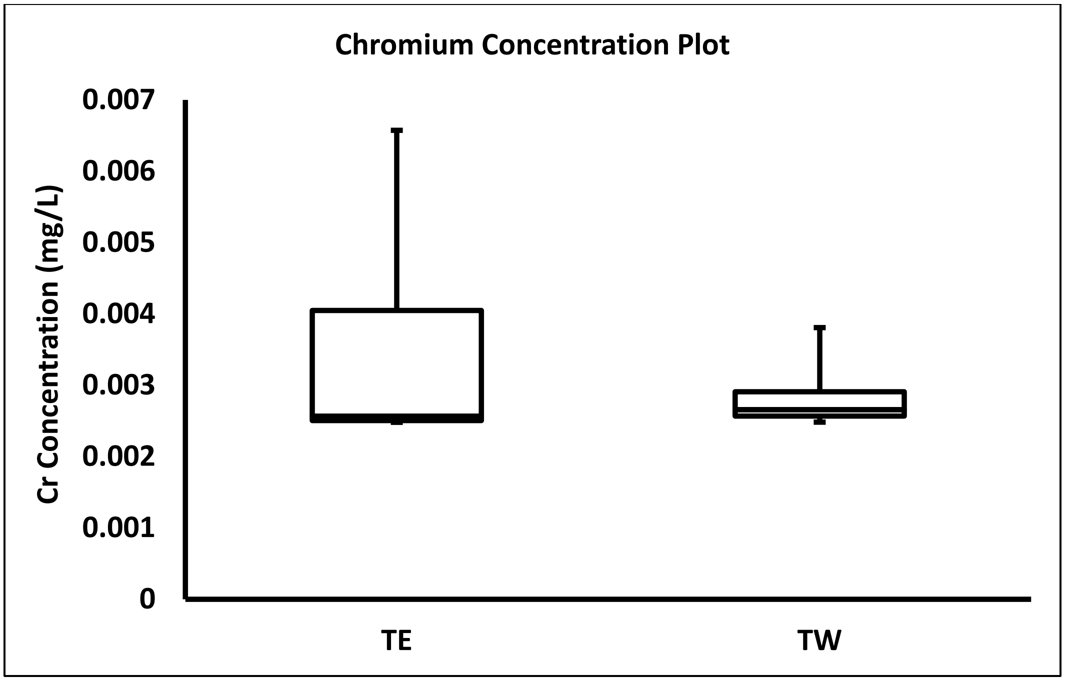


*Figure 35. Box and Whisker Plot for Cr Concentrations Collected from May to September 2015 (n=8)*

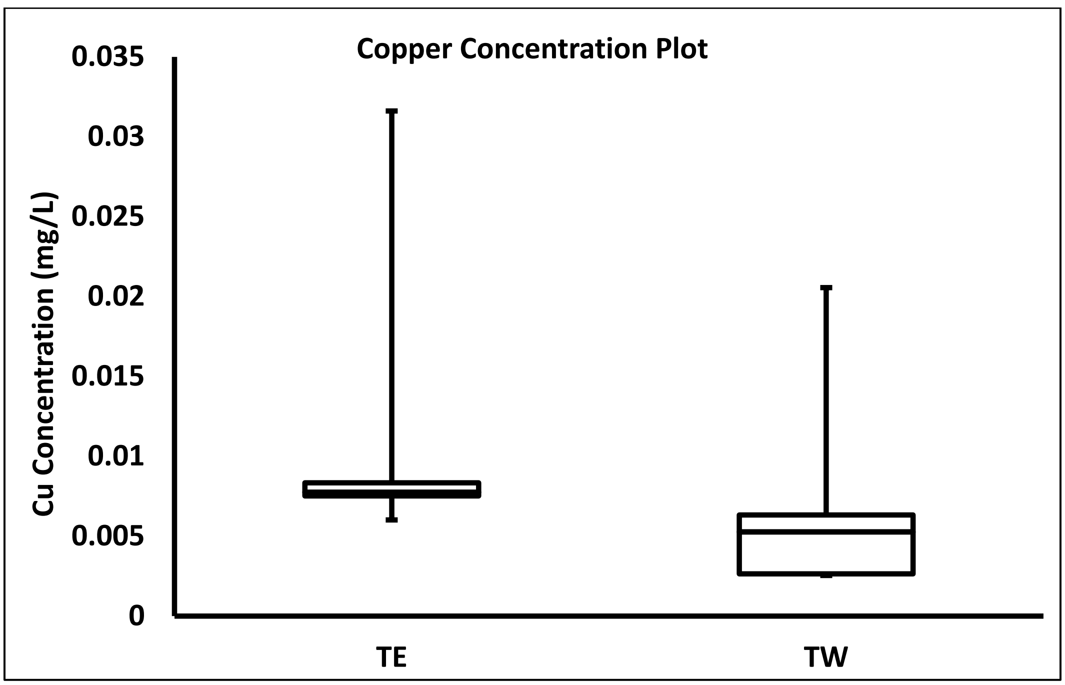


*Figure 36. Box and Whisker Plot for Cu Concentrations Collected from May to September 2015 (n=8)*

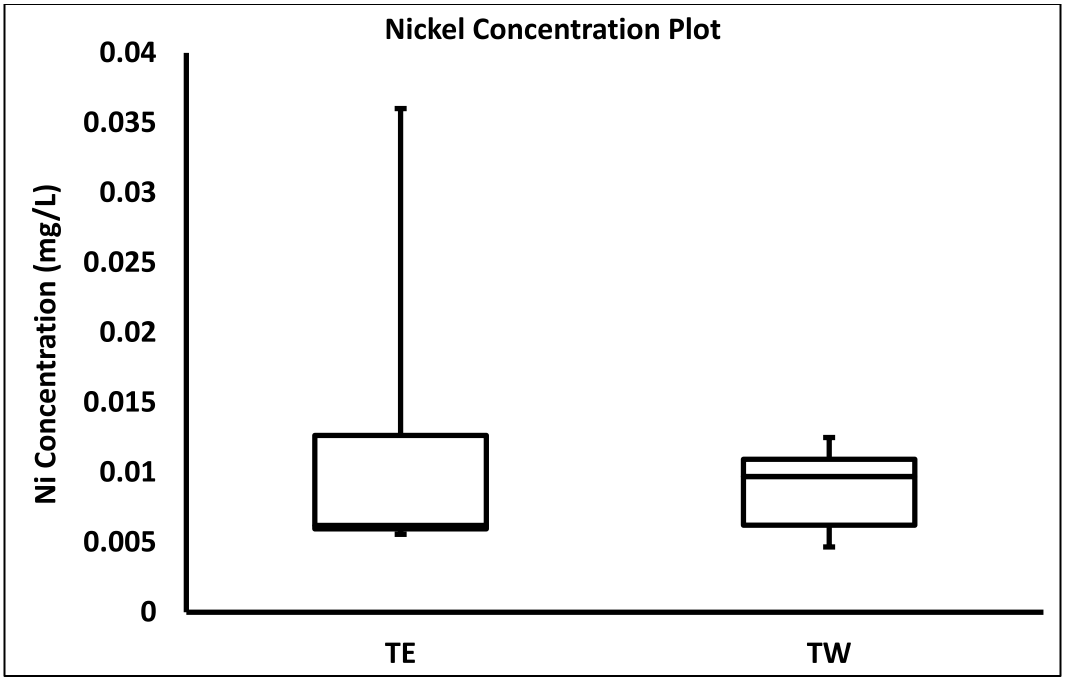


*Figure 37. Box and Whisker Plot for Ni Concentrations Collected from May to September 2015 (n=8)*

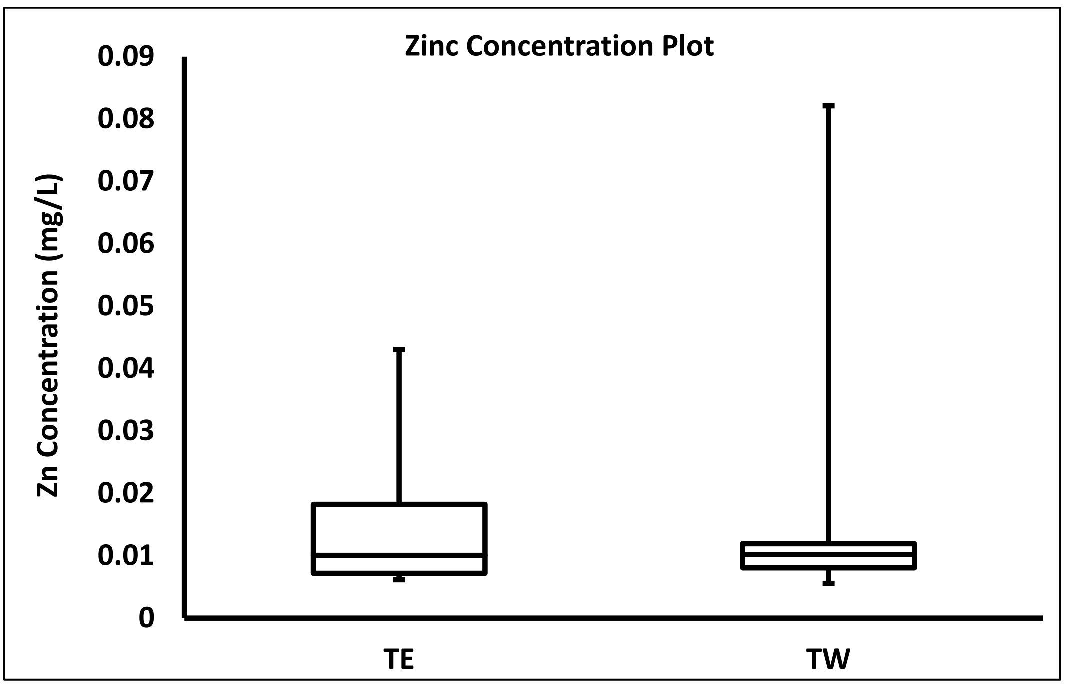


*Figure 38. Box and Whisker Plot for Zn Concentrations Collected from May to September 2015 (n=8)*

### 3.3.5 Mass Loading Data Analysis

Calculation of mass loads was possible due to the composite stormwater sampling method completed for this study. The water quality data paired with the water quantity data allowed for calculation of mass loading rates in g $ha^{-1}$ $day^{-1}$. Mass loading rates were calculated using the total runoff volume and duration of the storm event to acquire a volume per unit time value that was multiplied by the measured concentration of analytes and divided by the surface area of the watershed in hectares to acquire an adjusted mass per unit time per area loading rate.

Overall, TW treatment had smaller mean mass loading rates for TSS, TN, and $NO_3$-N, and larger nutrient mean mass loading rates for $NH_3$-N, TP, and TDP. A Student's t-test revealed that there were no statistical differences between any of the nutrient mass loading rates. These results are supported and refuted by several studies including, EPA (2000), Cheng et al. (2005), Selbig and Bannerman (2008), Bedan and Clausen (2009), and Waldron et al. (2010). It is important to note that the variability seen between these studies is expected since they studied the naturally inconsistent system of precipitation. EPA (2000) and Cheng et al (2005) reported percent decreases in TSS loads of 65 and 14 percent, respectively, compared to the 48 percent reduction for this study. Conversely, other studies monitored basins pre-construction and reported that TSS loads were increased for two years after construction (Selbig and Bannerman 2005) and were increased three-fold for several years after construction, (Bedan and Clausen 2009). The studies that monitored basins prior to construction actually acquired water quality data for pre-development conditions so it makes sense these studies found an increase in TSS loading rates. Cheng et al. (2005) found that TN loads were similar for the duration of the study which coincides with this study's results for which TN mass loading rates were only decreased by 16 percent. On the other hand, Waldron et al. (2010) found an increase in TN mass loading rates. EPA (2000) also reported a similar trend for percent reduction of $NO_3$-N mass loads of 18.5 percent, while the study completed in the Trailwoods neighborhood had a much larger decrease in $NO_3$-N mass loading rates of 75 percent. The difference in percent decreases could be attributed to the amount of LID BMPs in the Trailwoods neighborhood compared to that of the reviewed study that only utilized grassed swales. Bedan and Clausen (2009) reported an 85 percent decrease in the $NH_3$-N mass loading rates, while for this study there was actually a 37 percent increase. This discrepancy could be attributed to the 6/29/2015 event where there was a spike in $NH_3$-N concentrations (most likely from fertilization of lawns). With the 6/29/2015 event excluded there was actually a 73.8 percent decrease in $NH_3$-N mass loading rates when comparing TE control to TW treatment watersheds. Cheng et al. (2005) reported that TP mass loads were 40 percent higher leaving the LID site than the traditional basin; similarly, Selbig and Bannerman (2008) found that for the first four years of their study TP loads were higher leaving the LID basin. Waldron et al. (2010) found that median TP mass loads decreased for the duration of their study. Differences here again could be credited to organic substrates or fertilization.

The TE control watershed and TW treatment watershed mean nutrient mass loading rates are presented in Table 31-33 and Figure 39-44, graphically display the data in the form of box and whisker plots. The results of Spearman rank correlation analyses revealed several significant correlations for mass loading rates. The Spearman rank correlation analysis found that TN mass loading rates were significantly correlated to peak Q and the maximum five-minute precipitation intensity ($p$ value = 0.05 and < 0.05, respectively). This analysis also showed strong correlations for TDP mass loading rates and total runoff volume as well as total event precipitation ($p$ value = < 0.001). These results could potentially explain the export of phosphorus from TW treatment in that there was strong correlations between phosphorus compound mass loading rates and peak Q and total runoff volume. As the latter increased, so did the mass loading rates for phosphorus. As there was more stormwater moving through the basin, increased stormwater volumes were able to leach phosphorus compounds from the engineered substrate in the rain gardens.

*Table 31. Summary Data and Statistics for Nutrient Mass Loading Rates Analyzed for from May to September 2015, N.S. = No Sample, BDL = Below Detection Limits, and Diff. = Difference*

| | Nutrient Mass Loading Rates (g $ha^{-1}$ $day^{-1}$) | | | | | |
|---|---|---|---|---|---|---|
| | | TSS | | | TN | |
| | TE | TW | Diff. | TE | TW | Diff. |
| 5/22/2015 | 75.3 | 7.8 | 67.5 | 2.0 | 1.1 | 0.9 |
| 5/24/2015 | 767.5 | 212.0 | 555.5 | 16.3 | 6.7 | 9.6 |
| 6/29/2015 | N.S. | N.S. | --- | 2.1 | 2.6 | -0.5 |
| 7/3/2015 | 205.7 | 88.3 | 117.4 | 16.7 | 22.6 | -5.9 |
| 7/7/2015 | N.S. | N.S. | --- | N.S. | 2.1 | --- |
| 7/21/2015 | 0.5 | 0.5 | 0.0 | 1.0 | 0.9 | 0.1 |
| 8/4/2015 | 8.2 | 5.0 | 3.2 | 1.0 | 1.0 | 0.0 |
| 8/19/2015 | 7.8 | 12.3 | -4.5 | 1.2 | 1.4 | -0.2 |
| 9/8/2015 | N.S. | N.S. | --- | 1.2 | 1.2 | 0.0 |
| 9/20/2015 | 22.9 | 238.9 | -216.0 | 2.2 | 1.2 | 1.0 |
| Mean | 155.4 | 80.7 | 74.5 | 4.9 | 4.1 | 0.8 |
| Median | 22.9 | 12.3 | 10.6 | 2.0 | 1.3 | 0.7 |
| Std. Dev. | 258.7 | 96.0 | 162.7 | 6.2 | 6.4 | -0.2 |
| Max | 767.5 | 238.9 | 528.6 | 16.7 | 22.6 | -5.9 |
| Min | 0.5 | 0.5 | 0.0 | 1.0 | 0.9 | 0.1 |
| Sample Size | 7 | 7 | --- | 9 | 10 | --- |
| Std. Error | 97.8 | 36.3 | 61.5 | 2.1 | 2.0 | 0.1 |
| p | 0.2 | --- | --- | 0.3 | --- | --- |

*Table 32. Summary Data and Statistics for Nutrient Mass Loading Rates Analyzed for from May to September 2015, N.S. = No Sample, BDL = Below Detection Limits, and Diff. = Difference*

| | Nutrient Mass Loading Rates (g $ha^{-1}$ $day^{-1}$) | | | | | |
|---|---|---|---|---|---|---|
| | | $NO_3$-N | | | $NH_3$-N | |
| | TE | TW | Diff. | TE | TW | Diff. |
| 5/22/2015 | 0.7 | 0.1 | 0.60 | 0.1 | BDL | --- |
| 5/24/2015 | 7.8 | 0.6 | 7.2 | 1.3 | BDL | --- |
| 6/29/2015 | 0.3 | 0.0 | 0.3 | 1.5 | 3.8 | -2.3 |
| 7/3/2015 | 2.7 | 1.8 | 0.9 | 2.3 | 0.5 | 1.8 |
| 7/7/2015 | N.S. | 0.0 | --- | N.S. | 0.1 | --- |
| 7/21/2015 | 0.2 | 0.1 | 0.1 | 0.1 | 0.1 | 0.0 |
| 8/4/2015 | 0.2 | 0.2 | 0.0 | 0.1 | BDL | --- |
| 8/19/2015 | 0.2 | 0.0 | 0.2 | 0.5 | BDL | --- |
| 9/8/2015 | 0.2 | 0.2 | 0.0 | 0.2 | BDL | --- |
| 9/20/2015 | 0.0 | BDL | --- | BDL | BDL | --- |
| Mean | 1.4 | 0.3 | 1.1 | 0.8 | 1.1 | -0.3 |
| Median | 0.2 | 0.1 | 0.1 | 0.3 | 0.3 | 0.0 |
| Std. Dev. | 2.4 | 0.5 | 1.9 | 0.8 | 1.6 | -0.8 |
| Max | 7.8 | 1.8 | 6.0 | 2.3 | 3.8 | -1.5 |
| Min | 0.0 | 0.0 | 0.0 | 0.1 | 0.0 | 0.1 |
| Sample Size | 9 | 10 | --- | 9 | 10 | --- |
| Std. Error | 0.8 | 0.2 | 0.6 | 0.3 | 0.8 | -0.5 |
| p | 0.1 | --- | --- | 0.5 | --- | --- |

*Table 33. Summary Data and Statistics for Nutrient Mass Loading Rates Analyzed for from May to September 2015, N.S. = No Sample, BDL = Below Detection Limits, and Diff. = Difference*

| | Nutrient Mass Loading Rates (g ha$^{-1}$ day$^{-1}$) | | | | | |
|---|---|---|---|---|---|---|
| | TP | | | TDP | | |
| | TE | TW | Diff. | TE | TW | Diff. |
| 5/22/2015 | 0.106 | 0.070 | 0.036 | 0.025 | 0.018 | 0.007 |
| 5/24/2015 | 0.872 | 1.782 | -0.91 | 0.158 | 0.331 | -0.173 |
| 6/29/2015 | 0.037 | 0.270 | -0.233 | 0.015 | 0.042 | -0.027 |
| 7/3/2015 | 0.521 | 1.231 | -0.71 | 0.172 | 0.256 | -0.084 |
| 7/7/2015 | N.S. | 0.247 | --- | N.S. | 0.054 | --- |
| 7/21/2015 | BDL | 0.026 | --- | 0.006 | 0.012 | -0.006 |
| 8/4/2015 | 0.018 | 0.050 | -0.032 | 0.008 | 0.010 | -0.002 |
| 8/19/2015 | 0.014 | 0.009 | 0.005 | 0.009 | 0.017 | -0.008 |
| 9/8/2015 | 0.057 | 0.065 | -0.008 | 0.008 | 0.004 | 0.004 |
| 9/20/2015 | 0.021 | 0.046 | -0.025 | 0.012 | 0.015 | -0.003 |
| Mean | 0.206 | 0.379 | -0.173 | 0.046 | 0.076 | -0.03 |
| Median | 0.047 | 0.068 | -0.021 | 0.012 | 0.017 | -0.005 |
| Std. Dev. | 0.298 | 0.583 | -0.285 | 0.064 | 0.111 | -0.047 |
| Max | 0.872 | 1.782 | -0.91 | 0.172 | 0.331 | -0.159 |
| Min | 0.014 | 0.009 | 0.005 | 0.006 | 0.004 | 0.002 |
| Sample Size | 9 | 10 | -2 | 9.000 | 10.000 | --- |
| Std. Error | 0.105 | 0.184 | -0.079 | 0.021 | 0.035 | -0.014 |
| p | 0.057 | --- | --- | 0.070 | --- | --- |

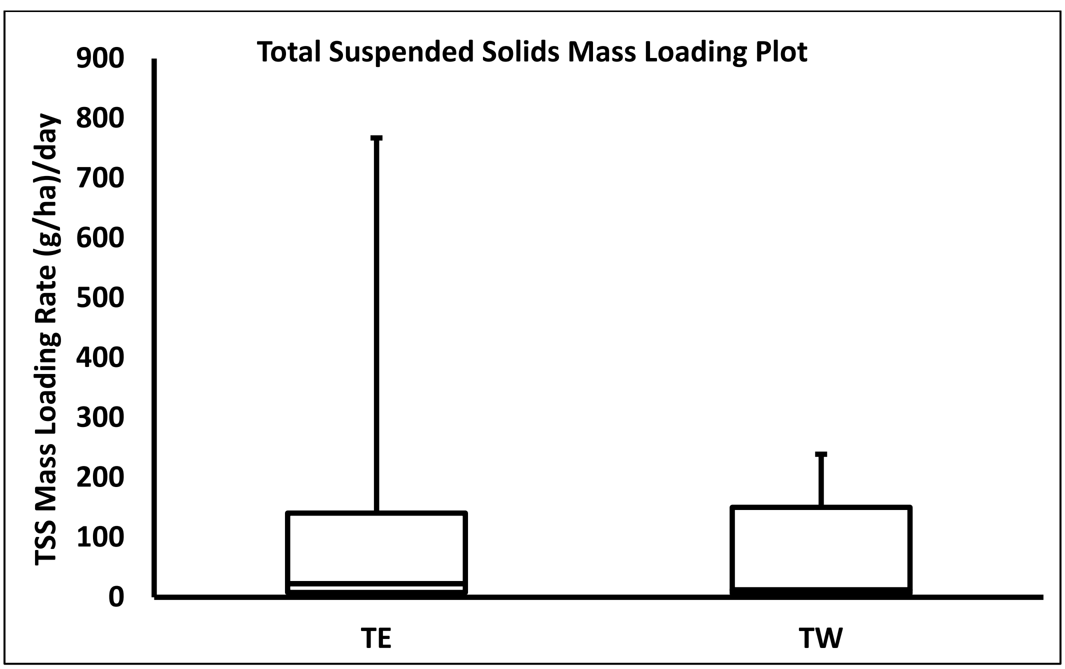


*Figure 39. Box and Whisker Plot for TSS Mass Loading Rates Collected from May to September 2015 (n=7) for TE Control and TW Treatment, Respectively*

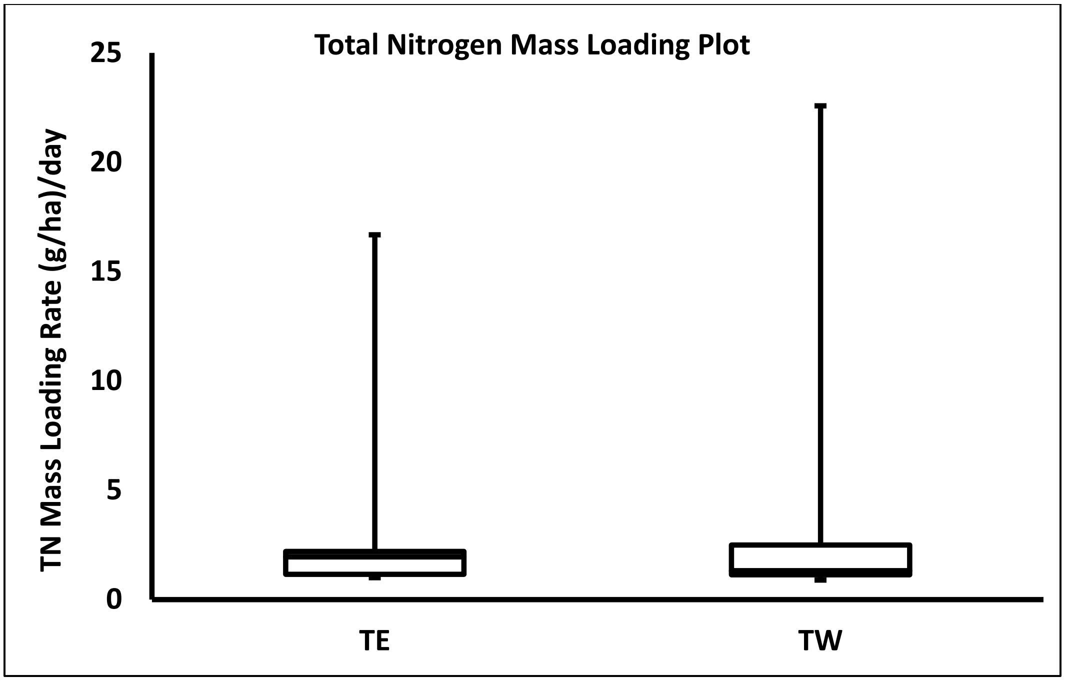


*Figure 40. Box and Whisker Plot for TN Mass Loading Rates Collected from May to September 2015 (n=9 and 10) for TE Control and TW Treatment, Respectively*

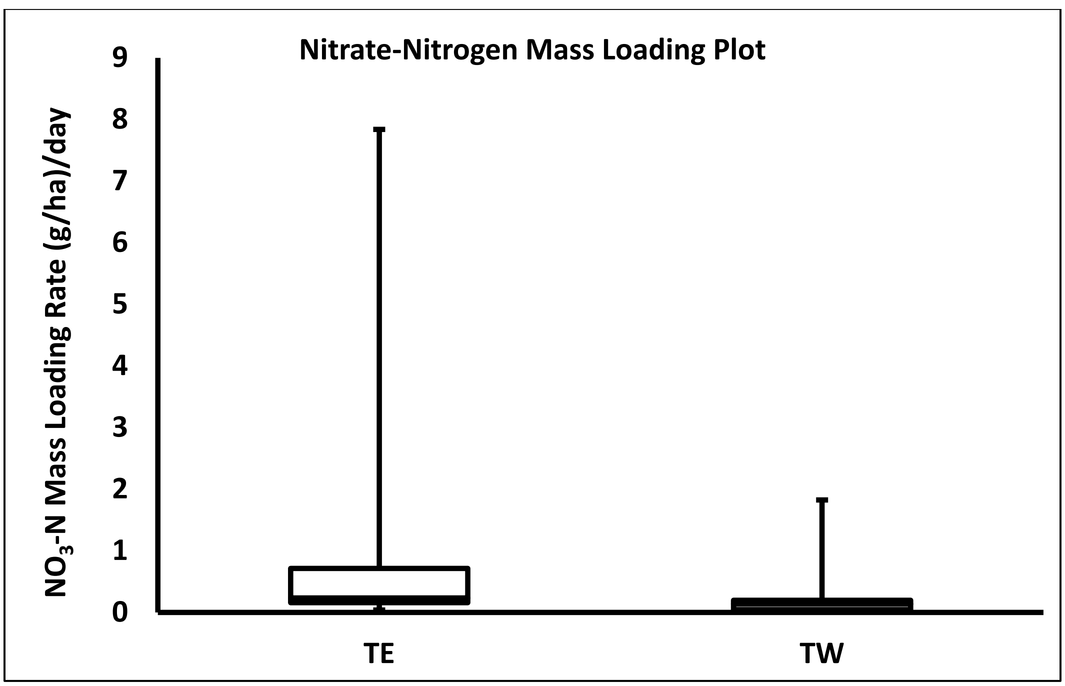


*Figure 41. Box and Whisker Plot for $NO_3$-N Mass Loading Rates Collected from May to September 2015 (n=9 and 10) for TE Control and TW Treatment, Respectively*

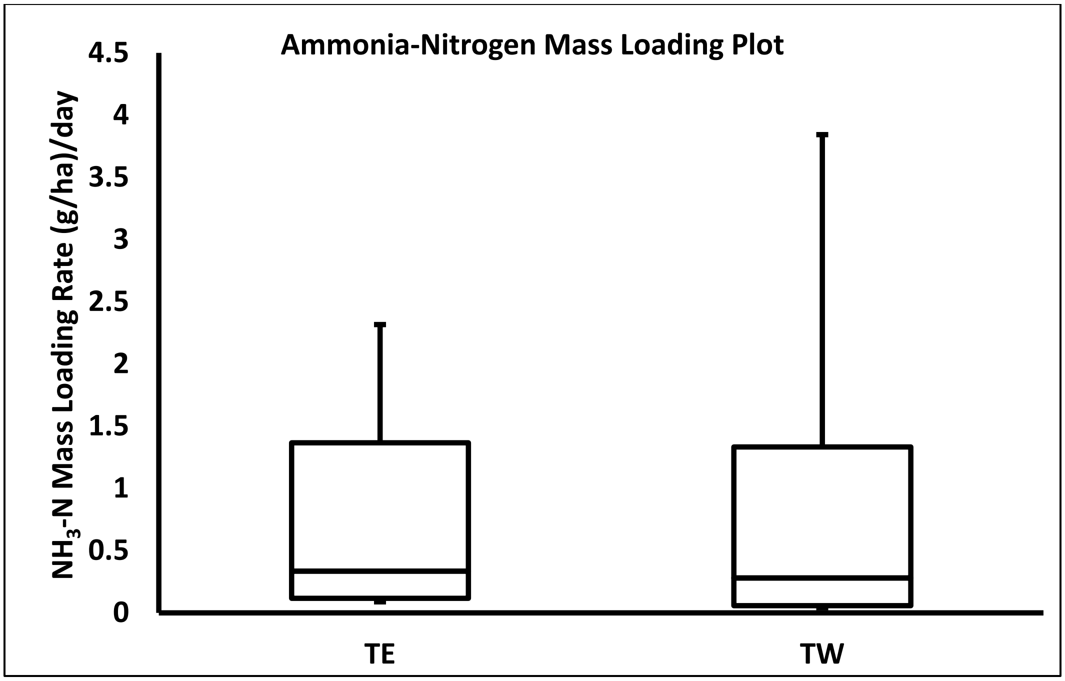


*Figure 42. Box and Whisker Plot for $NH_3$-N Mass Loading Rates Collected from May to September 2015 (n=9 and 10) for TE Control and TW Treatment, Respectively*

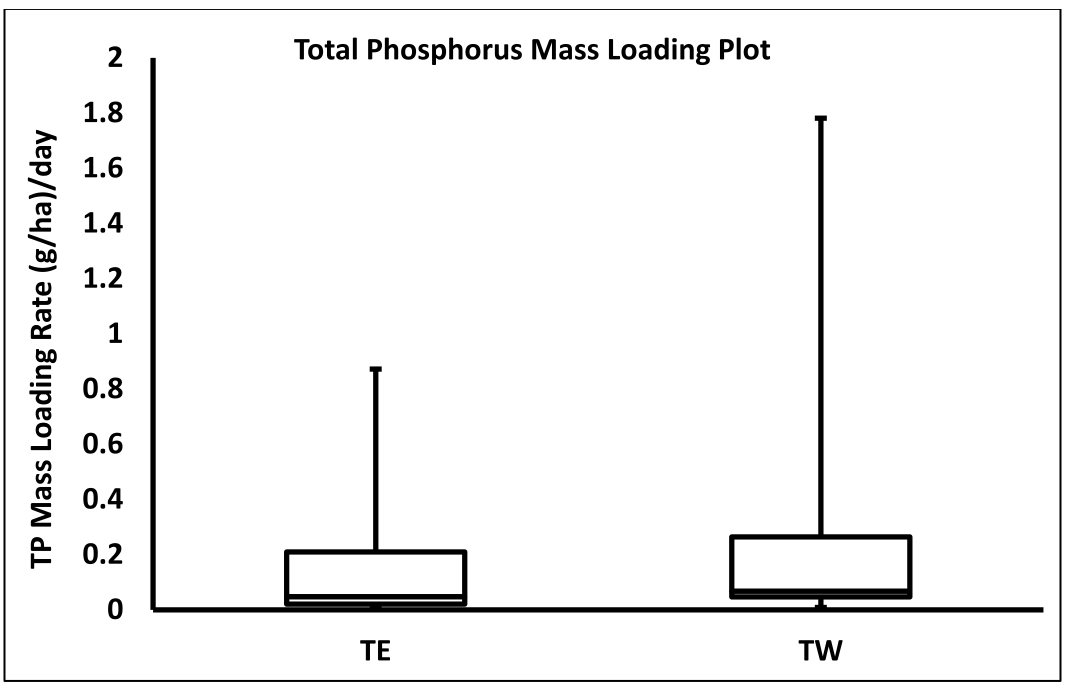


*Figure 43. Box and Whisker Plot for TP Mass Loading Rates Collected from May to September 2015 (n=9 and 10) for TE Control and TW Treatment, Respectively*

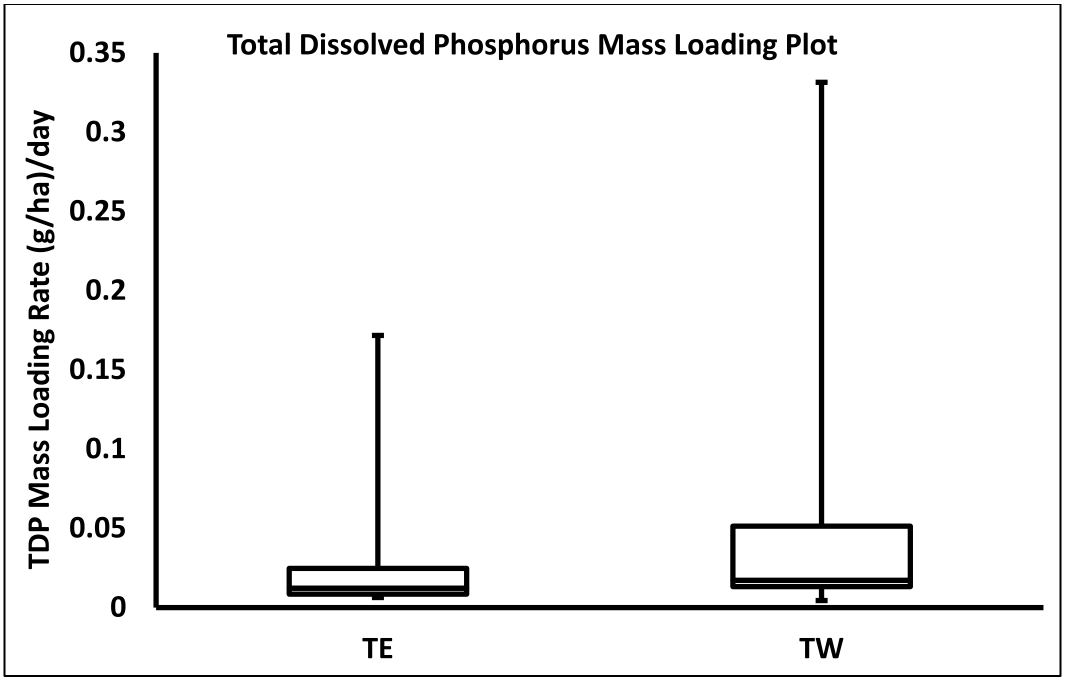


*Figure 44. Box and Whisker Plot for TDP Mass Loading Rates Collected from May to September 2015 (n=9 and 10) for TE Control and TW Treatment, Respectively*

For the total metal mass loading rates discussed in this study, all means were smaller for the TW treatment watershed (**Error! Reference source not found.**), most likely due to the anaerobic conditions in the rain gardens allowing for metal retention through formation of metal carbonates and sulfides via various biogeochemical and microbial processes. However, calcium mass loading rates were consistently larger in TW treatment than in TE control. Typically composts have a lime addition which provided more calcium ions as binding sites. This result indirectly supported the idea that excess phosphorus loads came from leaching of the engineered substrate. Overall, total metal mass loads were nominal in size and well below levels of environmental concern. Figure 45-49, provide a graphical overview of the data presented above in the form of box and whisker plots.

*Table 34. Summary Data and Statistics for Total Metal Mass Loading Rates Analyzed for from May to September 2015, N.S. = No Sample, BDL = Below Detection Limits, and Diff. = Difference*

| Total Metal Mass Loading Rates (g ha$^{-1}$ day$^{-1}$) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | Al | | | Cr | | | Cu | |
| | TE | TW | Diff. | TE | TW | Diff. | TE | TW | Diff. |
| 5/22 | 1.3207 | 0.4360 | 0.884 | 0.0016 | 0.0007 | 0.001 | 0.0034 | 0.0015 | 0.0019 |
| 5/24 | 27.0899 | 4.7335 | 22.356 | 0.0259 | 0.0072 | 0.018 | 0.0372 | 0.0142 | 0.023 |
| 6/29 | N.S. | N.S. | --- | N.S. | N.S. | --- | N.S. | N.S. | --- |
| 7/3 | 3.6214 | 4.3583 | -0.736 | 0.0116 | 0.0068 | 0.005 | 0.0364 | 0.0156 | 0.0208 |
| 7/7 | N.S. | N.S. | --- | N.S. | N.S. | --- | N.S. | N.S. | --- |
| 7/21 | 0.1746 | 0.0881 | 0.086 | 0.0008 | 0.0010 | -0.0002 | 0.0024 | 0.0011 | 0.0013 |
| 8/4 | 0.1451 | 0.1858 | -0.040 | 0.0007 | 0.0009 | -0.0002 | 0.0023 | 0.0017 | 0.0006 |
| 8/19 | 0.2321 | 0.2347 | -0.002 | 0.0008 | 0.0015 | -0.0007 | 0.0027 | 0.0016 | 0.0011 |
| 9/8 | 0.1645 | 0.0766 | 0.087 | 0.0007 | 0.0008 | -0.0001 | 0.0034 | 0.0041 | -0.001 |
| 9/20 | 0.2553 | 0.2797 | -0.024 | 0.0012 | 0.0014 | -0.0002 | 0.0029 | 0.0014 | 0.0015 |
| Mean | 4.1254 | 1.2991 | 2.826 | 0.0054 | 0.0025 | 0.0029 | 0.0113 | 0.0051 | 0.0062 |
| Median | 0.2437 | 0.2572 | -0.013 | 0.0010 | 0.0012 | -0.0002 | 0.0031 | 0.0016 | 0.0015 |
| Std. Dev. | 8.7519 | 1.8799 | 6.872 | 0.0085 | 0.0026 | 0.0059 | 0.0147 | 0.0057 | 0.009 |
| Max | 27.0899 | 4.7335 | 22.356 | 0.0259 | 0.0072 | 0.0187 | 0.0372 | 0.0156 | 0.0216 |
| Min | 0.1451 | 0.0766 | 0.068 | 0.0007 | 0.0007 | 0 | 0.0023 | 0.0011 | 0.0012 |
| Sample Size | 8 | 8 | --- | 8 | 8 | --- | 8 | 8 | --- |
| Std. Error | 3.0943 | 0.6646 | 2.429 | 0.0030 | 0.0009 | 0.0021 | 0.0052 | 0.0020 | 0.0032 |
| p | 0.1727 | --- | --- | 0.1284 | --- | --- | 0.0581 | --- | --- |

*Table 35. Summary Data and Statistics for Total Metal Mass Loading Rates Analyzed for from May to September 2015, N.S. = No Sample, BDL = Below Detection Limits, and Diff. = Difference*

| Total Metal Mass Loading Rates (g ha$^{-1}$ day$^{-1}$) | | | | | | |
|---|---|---|---|---|---|---|
| | | Ni | | | Zn | |
| | TE | TW | Diff. | TE | TW | Diff. |
| 5/22 | 0.0067 | 0.0020 | 0.0047 | 0.0056 | 0.0024 | 0.0032 |
| 5/24 | 0.1589 | 0.0277 | 0.1312 | 0.0840 | 0.0159 | 0.0681 |
| 6/29 | N.S. | N.S. | --- | N.S. | N.S. | --- |
| 7/3 | 0.0287 | 0.0261 | 0.0026 | 0.0290 | 0.0214 | 0.0076 |
| 7/7 | N.S. | N.S. | --- | N.S. | N.S. | --- |
| 7/21 | 0.0018 | BDL | --- | 0.0019 | 0.0022 | -0.0003 |
| 8/4 | 0.0016 | 0.0018 | -0.0002 | 0.0023 | 0.0044 | -0.0021 |
| 8/19 | 0.0020 | 0.0029 | -0.0009 | 0.0058 | 0.0060 | -0.0002 |
| 9/8 | 0.0011 | 0.0025 | -0.0014 | 0.0046 | 0.0165 | -0.0119 |
| 9/20 | BDL | BDL | --- | 0.0037 | 0.0061 | -0.0024 |
| Mean | 0.0287 | 0.0105 | 0.0182 | 0.0171 | 0.0094 | 0.0077 |
| Median | 0.0020 | 0.0027 | -0.0007 | 0.0051 | 0.0060 | -0.0009 |
| Std. Dev. | 0.0539 | 0.0116 | 0.0423 | 0.0266 | 0.0069 | 0.0197 |
| Max | 0.1589 | 0.0277 | 0.1312 | 0.0840 | 0.0214 | 0.0626 |
| Min | 0.0011 | 0.0018 | -0.0007 | 0.0019 | 0.0022 | -0.0003 |
| Sample Size | 8 | 8 | --- | 8 | 8 | --- |
| Std. Error | 0.0204 | 0.0047 | 0.0157 | 0.0094 | 0.0024 | 0.007 |
| p | 0.1723 | --- | --- | 0.2054 | --- | --- |

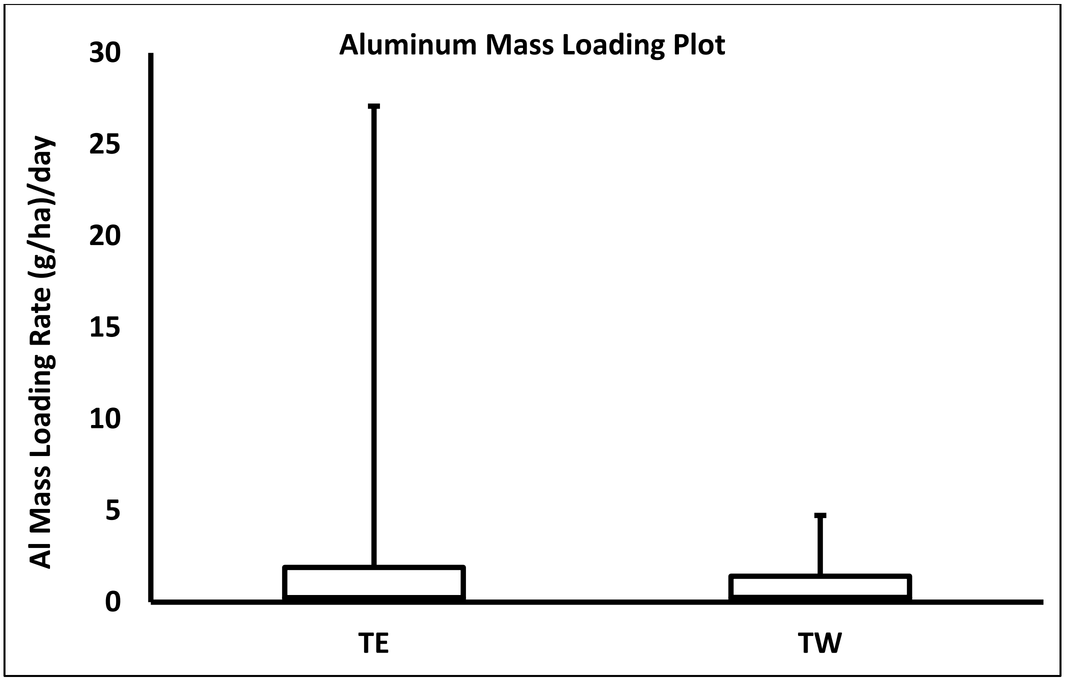


*Figure 45. Box and Whisker Plot for Al Mass Loading Rates Collected from May to September 2015 (n=8) for TE Control and TW Treatment, Respectively*

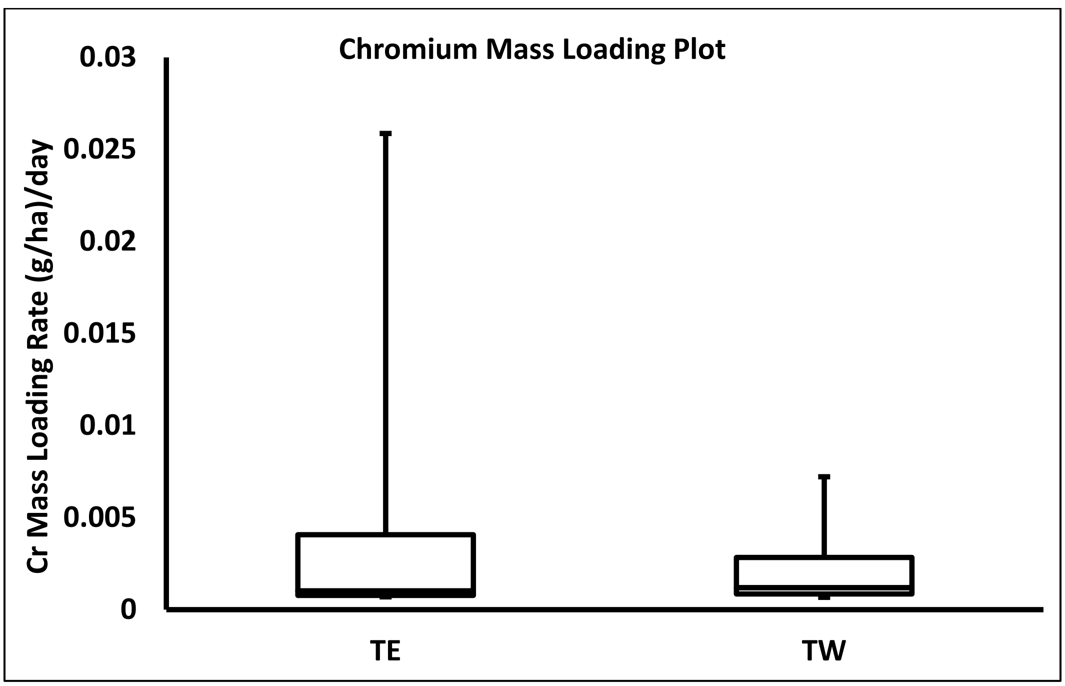


*Figure 46. Box and Whisker Plot for Cr Mass Loading Rates Collected from May to September 2015 (n=8) for TE Control and TW Treatment, Respectively*

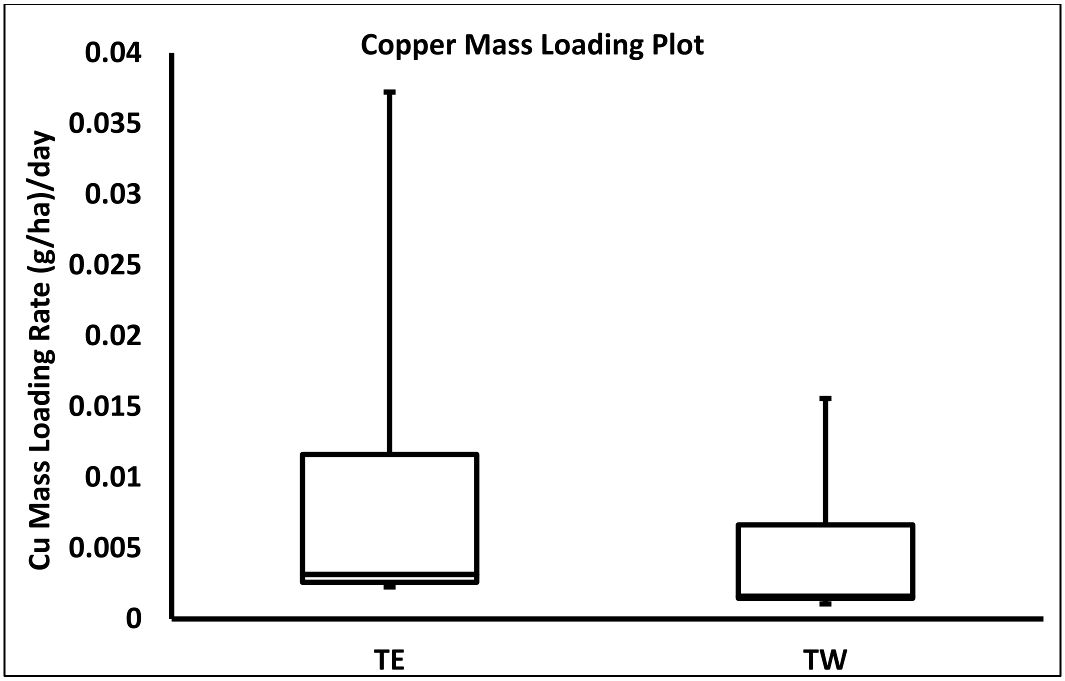


*Figure 47. Box and Whisker Plot for Cu Mass Loading Rates Collected from May to September 2015 (n=8) for TE Control and TW Treatment, Respectively*

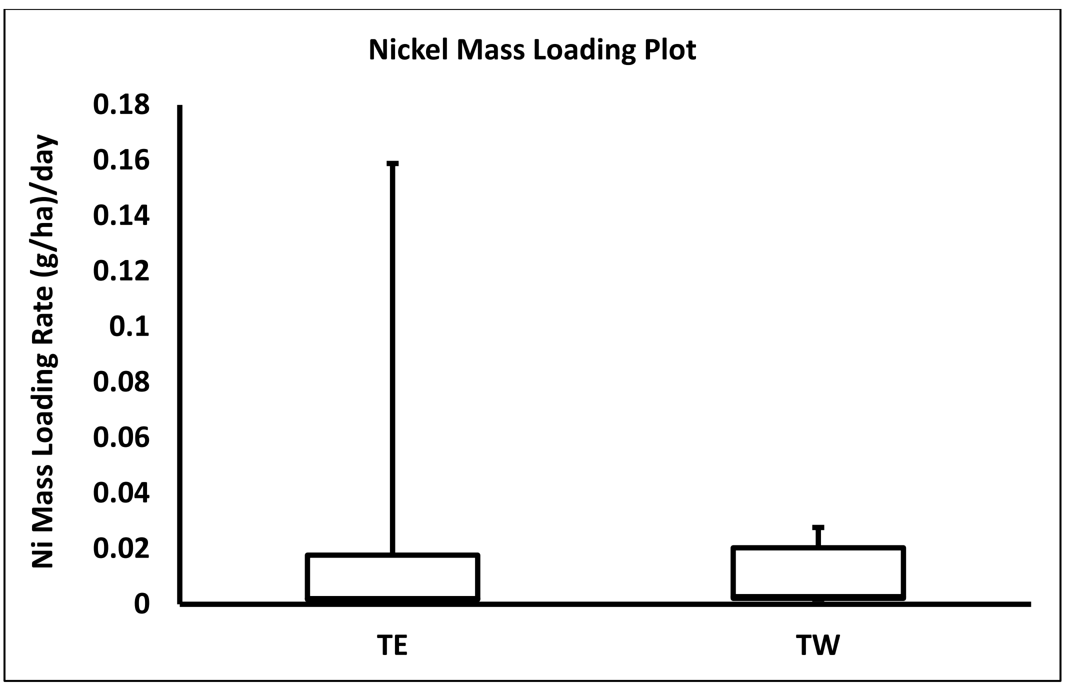


*Figure 48. Box and Whisker Plot for Ni Mass Loading Rates Collected from May to September 2015 (n=8) for TE Control and TW Treatment, Respectively*

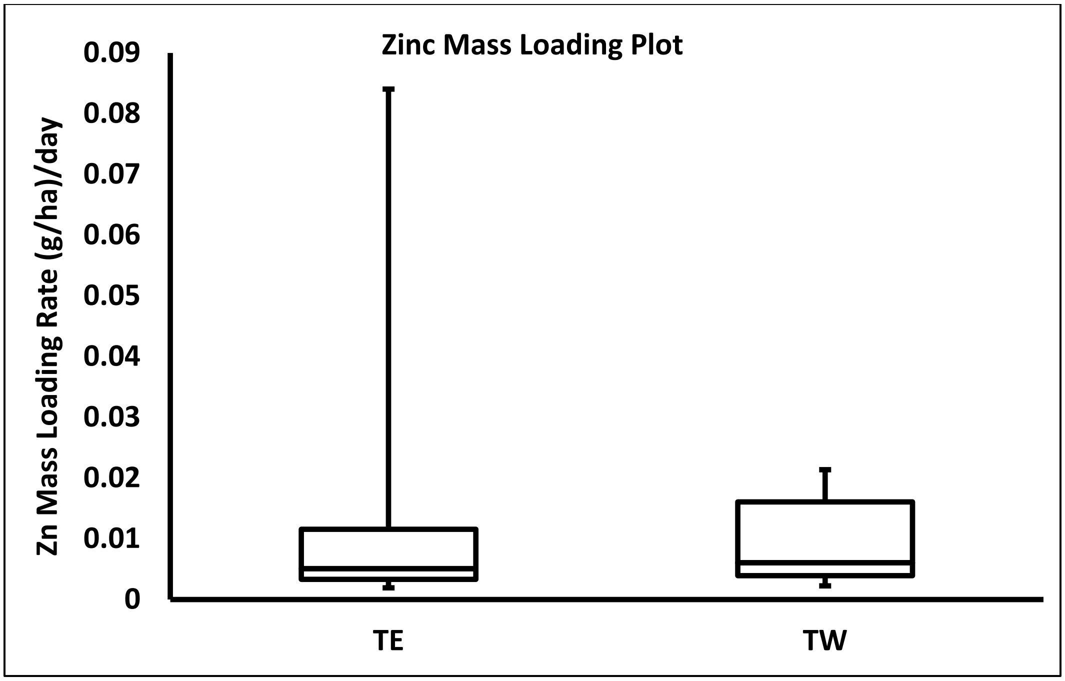


*Figure 49. Box and Whisker Plot for Zn Mass Loading Rates Collected from May to September 2015 (n=8) for TE Control and TW Treatment, Respectively*

## 3.4 Ecosystem Service Valuation

### 3.4.1 Provisioning – Rain Barrels

Value provided by rain barrels in the TW treatment watershed was calculated in an attempt to determine if the provisioning ecosystem service provided could offset water consumption costs. Using data from the City of Norman (2016), it was determined that water rates are established in brackets. For example, if a residence uses between 5,001 and 15,000 gallons of water they will pay $4.10 for every 1,000 gallons consumed plus a base fee of $6.00, whereas a residence which consumes between 15,001 and 20,000 will pay $5.20 per 1,000 gallons consumed with a $6.00 base fee. This knowledge, along with rain barrel size and assumptions made about the percentage of rooftop that drains into the barrels, allowed for valuation. Figure 50 shows the bracketed water usage along with potential dollars saved per year if the rain barrel was properly utilized. Assumptions had to be made in order to perform this valuation, some of which included that the water in the rain barrels was completely used after each event > 0.24 inches (magnitude of the design storm that would fill the rain barrels) and was emptied for storms < 0.24 inches that did not fill the rain barrel. Rain barrel value was extrapolated to the basin scale to allow for calculation of other ecosystem services provided by other implemented LID BMPs. It is likely that similar ecosystem service valuation studies would perform the same generalization because valuation of ecosystem services on a basin scale versus a lot-level approach would provide more realistic data. Overall, household and basin wide values ranged from $15.38 and $261.46 per year, respectively, for the lower water usage bracket to $25.04 and $425.68 per year, respectively, for the upper bracket.

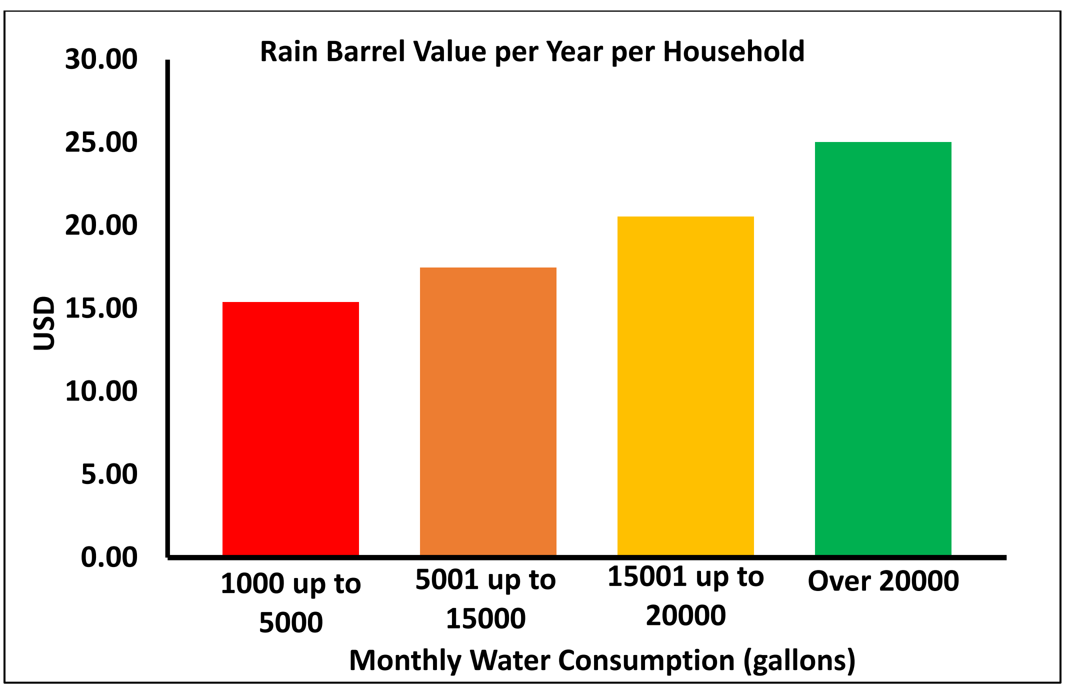


*Figure 50. Calculated Rain Barrel Value per Household versus Monthly Household Drinking Water Consumption*

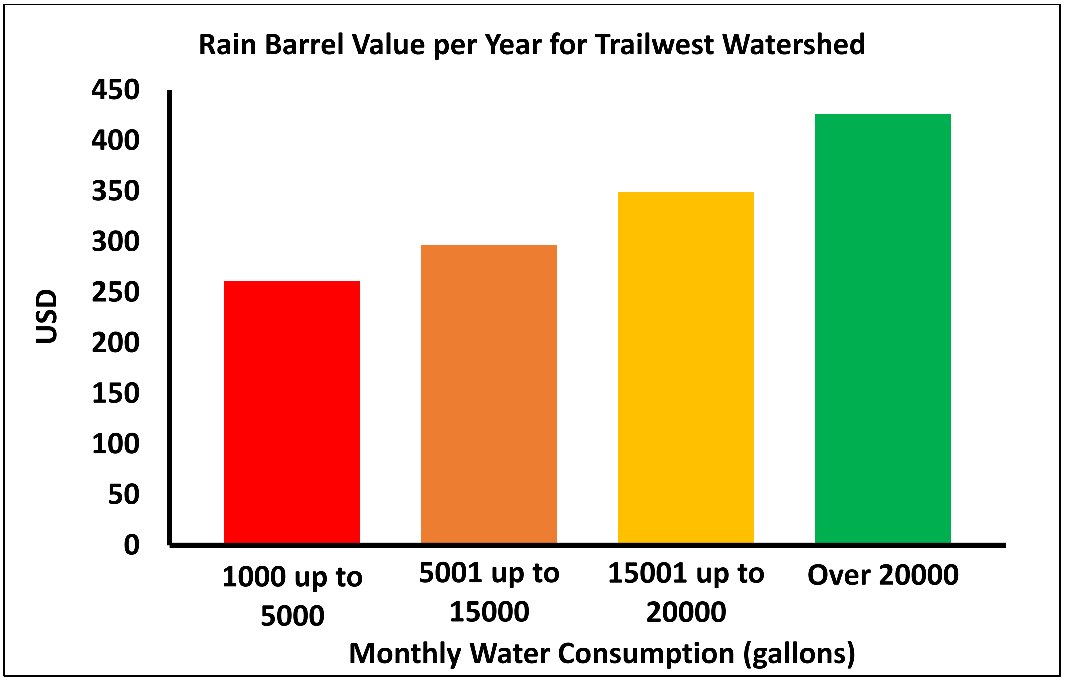


*Figure 51. Calculated Rain Barrel Value for Trail West Watershed versus Monthly Household Drinking Water Consumption*

### 3.4.2 Regulating – Flood Attenuation

The regulating value provided in the form of flood attenuation was calculated to determine the hydrologic effectiveness of the LID BMPs at the Trailwoods site. First, a comparison to traditional stormwater management was conducted to determine the size of stormwater ponds based on engineering design criteria. The City of Norman (2006) outlines design criteria for these types of installments, in which it was stated that the ponds must be able to handle excess runoff from a one year frequency precipitation event ($Q_{100}$) under fully urbanized conditions. Armed with that knowledge, it was possible to determine the worst case scenario in which there would be zero percent reduction in total runoff volume caused by the complete absence of LID BMPs. Figure 52 shows this value as 0 percent decrease with a resulting cost of $47656. From there it was decided it would be advantageous to provide a wide range of percent reductions and associated costs to allow developers and city planners to determine the best course of action for their region. Overall, the largest cost difference was between the 50 and 75 percent reduction in total runoff volume. Designing for a 100 percent reduction in total runoff volume by LID BMPs was not possible because stormwater pond sizing and resultant costs were needed for comparison. So, calculation of a 95 percent reduction in total runoff volume was completed. Selbig and Bannerman (2008) measured a 96 percent decrease in potential runoff volume through storage, retention, and infiltration by forested hillslopes, lawns and grassed swales.

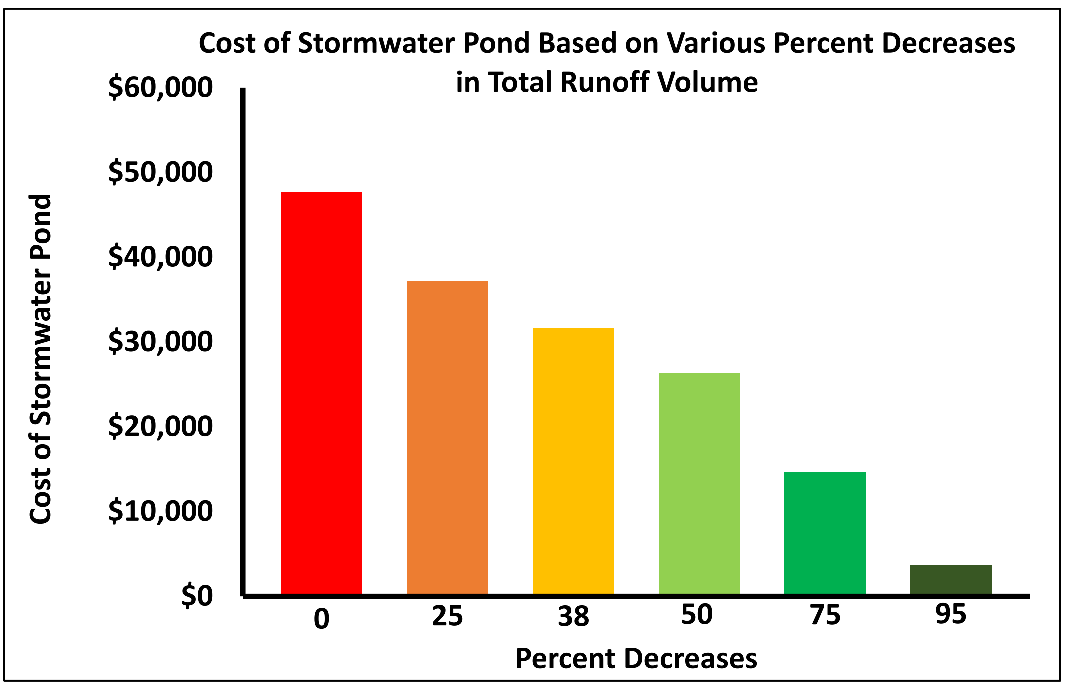


*Figure 52. Stormwater Pond Costs Saved Based on Percent Decreases in Total Runoff Volume*

### 3.4.3 Regulating – Nutrient Retention

The regulating value provided by the LID BMPs at Trailwoods in the form of nutrient and total metal retention was calculated two different ways. The first is highlighted in Figure 53 and Figure 55 and has units of U.S Dollars Year$^{-1}$ one percent change$^{-1}$. For example, each percent of TSS removed in TW treatment via LID BMPs provides a cost effectiveness of $359.18; conversely for substances of which there was export (like TP) it would cost $489.89 for each percent increase when compared to TE control. Over time it was evident that value of these LID BMPs decreases for several reasons, including, less efficient removal of pollutants, compounding operation and maintenance costs, and a lack of increased sources of revenue. After the first twenty years of implementation, it seems the most value was lost, perhaps due to the required operation and maintenance of the rain gardens and permeable pavement that must occur every twenty years to ensure functionality.

The second method of valuation is emphasized in Figure 54 and Figure 56 and has units of a given percent change Year$^{-1}$ U.S. dollar$^{-1}$. For example, for each 0.13 percent of TSS removed per year a dollar was gained and for every 0.25 percent of TP exported per year a dollar was lost. Again notice how as time progresses the LID BMP systems become more efficient and it costs less to remove a percent of a pollutant on year sixty than it would a day after implementation. Wossink and Hunt (2003), performed a similar analysis where they related the costs of implementation to the ecological benefit of LID BMPs in order to assess which practices are the most cost effective.

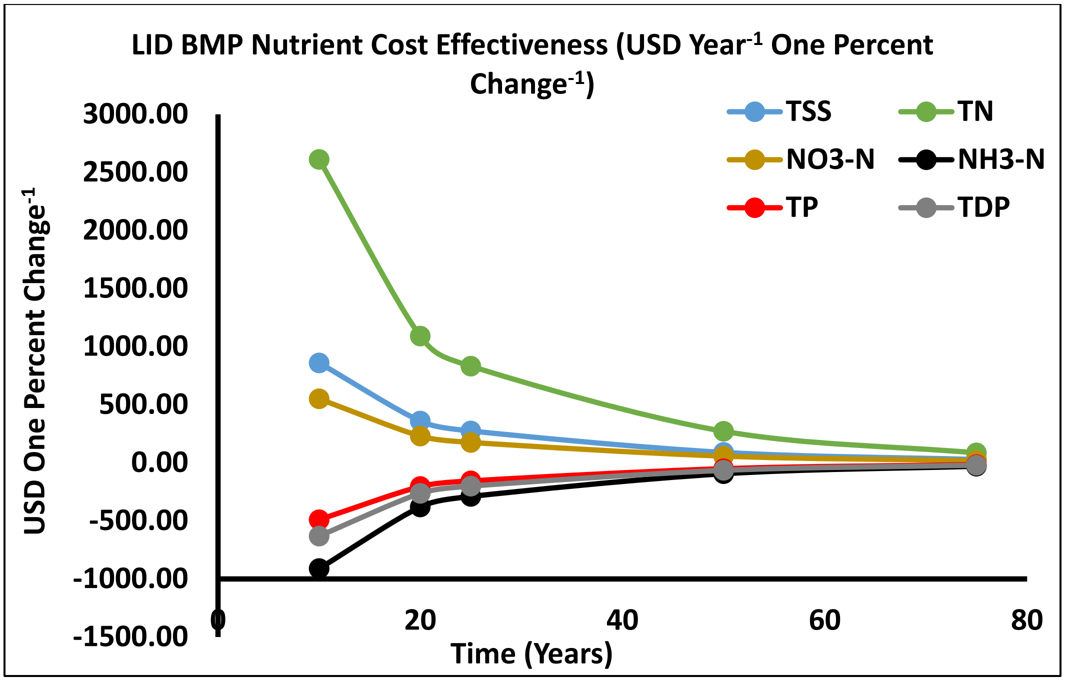


*Figure 53. USD Value per Year for a 1 Percent Change in Nutrient Mass Loading Rates*

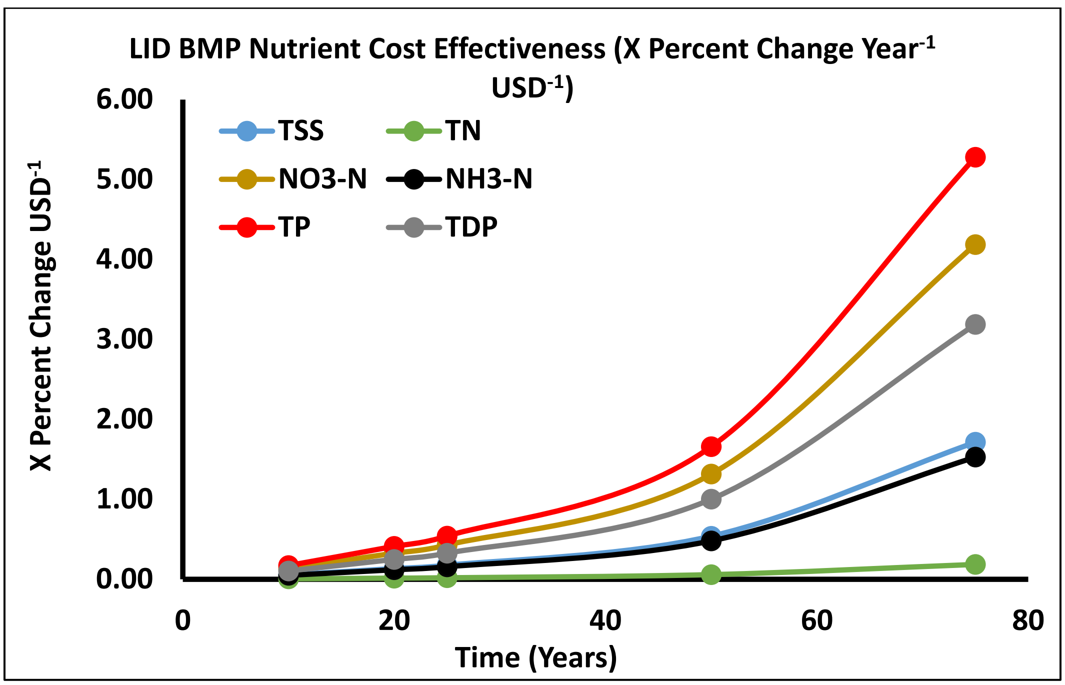


*Figure 54. Percent Change in Nutrient Mass Loading Rates Equal to 1 USD versus Time*

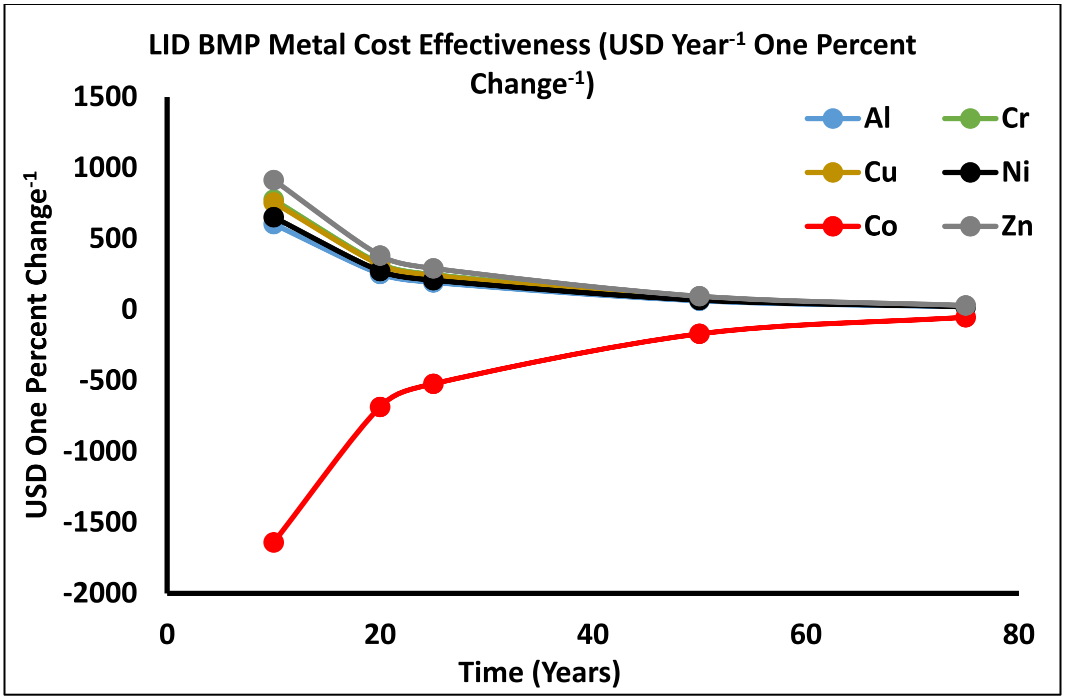


*Figure 55. USD Value per Year for a 1 Percent Change in Total Metal Mass Loading Rates*

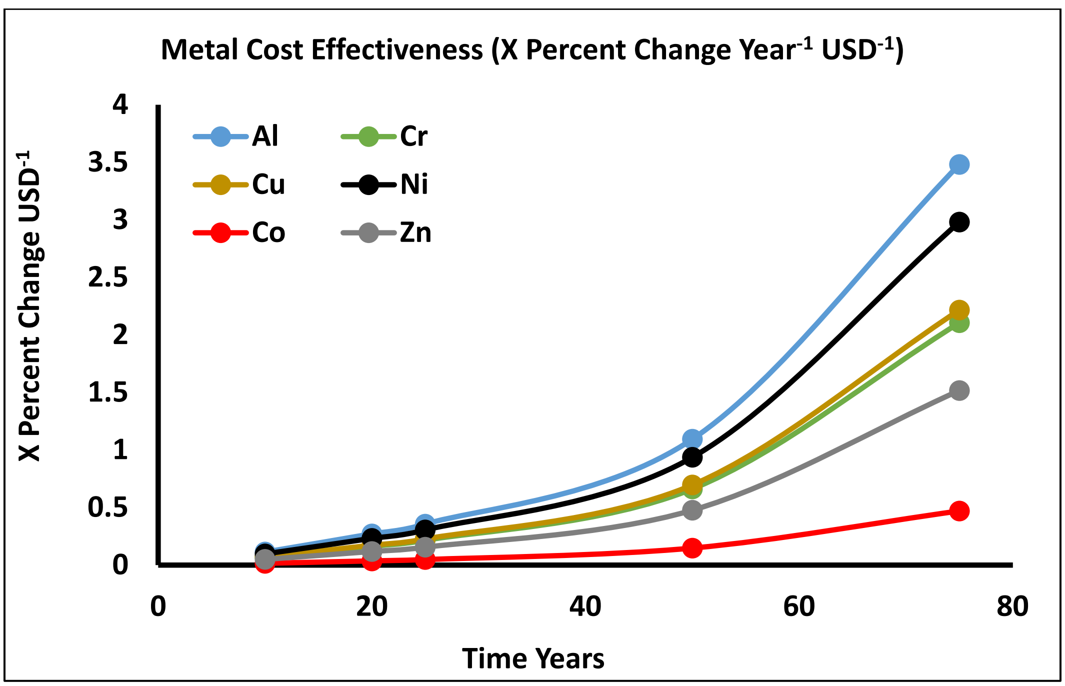


*Figure 56. Percent Change in Total Metal Mass Loading Rates Equal to 1 USD versus Time*

In order to calculate this cost effectiveness, numerous assumptions had to be made, including the, number of construction workers, their salary, hours to complete the project, cost of implementation of LID BMPs, future discount rates, life cycle costs, and revenue generated by other LID BMPs present at the site. Given these assumptions, a methodology outlined by EPA (2007) was followed carefully and adequate research was completed to ensure that assumptions were not without warrant. Typically this type of analysis is completed prior to construction to ensure that revenue generated will exceed capital costs, but since pre-development data were not collected, this analysis provided an outlook into the future of the LID BMPs at the Trailwoods residential neighborhood.

## 4.0 Conclusions

Collection of stormwater runoff data for ten precipitation events at two watersheds in a portion of the Trailwoods residential neighborhood allowed for direct analysis of LID BMP effectiveness. The impact of these LID BMPs was evaluated through water quality and water quantity analyses, along with quantification of ecosystem services provided. The ten precipitation events had magnitudes ranging from 0.34 to 3.99 inches with maximum five-minute intensities ranging from 0.48 to 2.64 inches per hour. These precipitation events produced total runoff volumes ranging from 744 to 42700 CF for TE control and 14820 to 24514 CF for TW treatment, with an overall total runoff volume of 110539 CF for TE control and 81261 CF for TW treatment, a 26.5 percent difference. A Spearman's rank correlation analysis also showed a strong correlation between total runoff volume and event total precipitation (p value = <0.005). Peak Q rates in TW treatment were significantly lower for all events (p value = 0.040) with an average difference of 47.3 percent. A Spearman's rank correlation analysis also revealed strong correlations between peak Q and event total precipitation (p value = 0.01) and peak Q and maximum five-minute intensity (p value = <0.05). Runoff depths were significantly different (p value = 0.01) between the two watersheds, while runoff ratios and lag times did not display any significant differences.

TSS concentrations ranged from 1.6 to 174.0 mg/L for TE control and 1.2 to 84.8 mg/L for TW treatment, with an average percent difference of 48 percent. TN concentrations ranged from 3.2 to 14.4 mg/L for TE control and 2.2 to 9.9 mg/L for TW treatment, with an average percent difference of 22 percent. $NO_3$-N concentrations ranged from 0.08 to 2.2 mg/L for TE control and 0.03 to 0.99 mg/L for TW treatment, with an average percent difference of 63.8 percent. $NH_3$-N concentrations ranged from BDL to 10.4 mg/L for TE control and BDL to 14.5 mg/L for TW treatment, with an average percent difference of 13 percent. TP concentrations ranged from 0.04 to 0.54 mg/L for TE control and 0.01 to 1.02 mg/L for TW treatment, with an average percent difference of -89 percent. Finally, TDP concentrations ranged from 0.02 to 0.1 mg/L for TE control and 0.02 to 0.16 mg/L for TW treatment, with an average percent difference of -40 percent. Overall, significant difference existed between TE control and TW treatment for $NO_3$-N concentrations (p value = 0.01). Although there are substantial increases in both TP and TDP concentrations they were not found to be significant. In general metal concentrations were lower leaving TW treatment than TE control most likely due to metal retention in the engineered substrate in the rain gardens.

Moreover, mass loading rates also followed similar trends but significance was found between several parameters. TN mass loading rates were significantly correlated with peak Q and the maximum five-minute precipitation intensity (p value = 0.05). TDP mass loading rates were significantly correlated with total runoff volume and total event precipitation (p value = < 0.05).

Ecosystem service values were calculated to determine if LID BMPs provided long-term economic benefits when compared to traditional stormwater management. Overall, the three ecosystem services evaluated in this study showed that indeed the LID BMPs at the study site could provide an economic benefit over time. The provisioning value provided by rain barrels showed that there is value in stormwater capture that increases with water consumption. For example a household that consumes between 5000 and 15000 gallons of water a month will find a value of $17.50 per year, while a household that consumes more than 20,000 gallons a month will find a value of $25.04 a year. The regulating value of flood attenuation provided by the LID BMPs at the study site showed that as the amount of total runoff volume captured in the LID BMPs increased, so did the flood attenuation value. For example a system that had a zero percent decrease in total runoff volume would require a stormwater pond that would cost approximately $47,656. If LID BMPs were implemented that resulted in a 50 percent decrease in total runoff volume, the stormwater pond would be much smaller and only cost $26,289. This metric allows for developers to determine if implementation of LID BMPs would provide for more efficient use of dollars versus simply building a stormwater pond. The LID BMPs also focus on water quality whereas a stormwater pond only focuses on water quantity, so yet another benefit was quantified by the final ecosystem service. However, if stormwater was sampled after it was released from the existing stormwater pond north east of the study site, this study could have compared true traditional urban stormwater practices to that of LID BMPs. The regulating ecosystem service in the form of nutrient retention showed that over time the cost effectiveness of LID BMPs increases. This is due to the fact that one USD was associated with less than a one percent change in nitrate-nitrogen mass loading rates at year ten but almost a two percent change in nitrate-nitrogen mass loading rates at year 60. This value was calculated two ways, the first provides a USD year$^{-1}$ one percent change$^{-1}$ value, while the second is the value of a given percent change year$^{-1}$ one USD$^{-1}$.

Overall, presence of LID BMPs at the study site had a positive influence on water quantity and quality, while providing an economic benefit in the form of ecosystem services. Mean concentrations and loading rates were lower for the TW treatment watershed all constituents aside from $NH_3$-N and phosphorus compounds, which again was thought to be a result of the engineered substrate in the rain gardens. Peak discharge rates were lower for all storm events, and total discharge rates were lower for 50 percent of the precipitation events. An ecosystem services analysis did show that the TW treatment watershed was provided with long term economic benefits, which over time could outweigh the capital costs of construction. In conclusion, the data collected represent a highly variable manmade system in which LID BMPs do provide beneficial water quantity and quality functions, and economic alternatives to traditional stormwater management. Ecosystem services should be focused on in future studies to better quantify the overall benefits provided by these types of systems.

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