ABSTRACT

THOMAS, DUWANE. Stormwater Manufactured Treatment Devices: Evaluating the Performance of a Subsurface Chamber System and Establishing a Framework to Assess Pathogen Treatment. (Under the direction of Dr. William F. Hunt).

Stormwater runoff from the urban environment is characterized as having higher volumes, peak flow rates, and pollutant loads, compared to those of a natural watershed. Researchers have linked urban runoff to numerous environmental impacts including accelerated streambank erosion, increased frequency of flooding, and disrupted aquatic habitats. Urban runoff is also a major contributor of pathogens to surface water, which are derived mainly from the fecal matter of humans and warm-blooded animals. The presence of pathogens in stormwater runoff not only carries health implications for recreational water users, but also economic ramifications associated with the loss of tourism and commercial fishing as a result of beach and other water body closures.

Nature-based stormwater control measures (SCMs) are employed to mitigate the impact of urban runoff. However, siting these practices in heavily developed areas can be difficult due to the cost and scarcity of available land. Proprietary SCMs, which are also referred to as Manufactured Treatment Devices (MTDs), are designed to treat runoff while using little or no usable land area. This, coupled with their potential for effective hydrologic and water quality management, makes them viable options for urban and ultra-urban stormwater treatment.

In this study, StormTech's® subsurface chamber system (SCS), installed in Raleigh, NC, was monitored to assess its hydrologic and water quality treatment performance. The SCS is an underground detention system that treats runoff via filtration, infiltration, and sedimentation. The SCS was evaluated using the requirements specified by the North Carolina Department of Environmental Quality (NCDEQ) for NEw Stormwater Technologies (NEST). During the 14-

month study period, the SCS provided reliable water quality improvements, lowering total suspended solids (TSS), total nitrogen (TN), and total phosphorus (TP) concentrations by mean values of 86%, 34%, and 64%, respectively. TSS outflow concentrations met performance thresholds for primary SCMs set by NCDEQ and effluent loads of TN and TP met targets for the Tar Pamlico, Jordan, and Falls Lake watersheds. The SCS also effectively lowered runoff volume and peak flow rates. Mean and median values for volume reduction were 65% and 75%, respectively, while mean and median values for peak flow reduction were 89% and 97%, respectively.

A meta-analysis was conducted to obtain data on the pathogen-mitigating performance of SCMs. The changes in geometric mean concentrations for three fecal indicator bacteria (FIB), namely fecal coliform, *Escherichia Coli (E. Coli)*, and *enterococci*, were compiled for a total of 32 SCMs [bioretention cells (7), constructed stormwater wetlands (CSWs) (7), wet ponds (8), dry detention basins (3), infiltration basins (3), bioswale (1), and MTDs (3)]. SCMs were ranked into three tiers based on their ability to treat FIB. SCMs in Tier 1 provided the most effective treatment of FIB and generally employed filtration through media (bioretention cells and infiltration basins), or for 2 of 3 FIB, sedimentation concomitant with phytoremediation (CSWs). SCMs in Tier 2 were less effective than those of Tier 1, but still provided some removal of FIB. These SCMs employed sedimentation (wet ponds). Tier 3 included SCMs that did not provide effective removal of FIB and generally increased outflow concentrations to receiving waters. SCMs in Tier 3 (dry ponds and the MTDs studied herein) lacked filtration and only temporarily employed sedimentation. The bioswale included in the analysis was not placed into a tier due to limited data. The performance data were also used to construct a framework that can be used to assess whether an SCM’s mitigation of FIB is comparable to that of Tier 1 SCMs.

Stormwater Manufactured Treatment Devices: Evaluating the Performance of a Subsurface Chamber System and Establishing a Framework to Assess Pathogen Treatment.

By
Duwane Thomas

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APPROVED BY

Dr. William F. Hunt Committee Chair	Dr. Emily Berglund
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DEDICATION

I dedicate this thesis to my family. Thank you for your sacrifice, support, and for always believing in me.

BIOGRAPHY

Duwane Jeremy Fitz-Allan Thomas was born on October 10th, 1990 on a small island in the southern Caribbean called Saint Vincent and the Grenadines. He is the son of Reginald and Cheryl Thomas and brother to Jon-Anthony and Esmond Thomas. He grew up a lover of the outdoors and physical activity. As a youngster, he frequented many of the country's beaches, nature trails, and cultural sites.

Tennis is a big part of Duwane's life. At the age of 11, he discovered his passion for the sport and spent most of his teenage years competing at national tournaments. He was ranked among the top players in the country and was one of two tennis players to represent his country in the 2008 Commonwealth Youth Games held in Pune, India. He likely passed his passion for the sport to his younger brother Jon-Anthony, who became the youngest tennis champion in the country's history.

Duwane left his home country to attend Morgan State University on a tennis scholarship. He graduated in 2014 with a bachelor's degree in civil engineering. After working in North Carolina for a few years, he began pursuing his Master's degree at North Carolina State University in spring of 2018. Duwane transferred to the Biological and Agricultural Engineering department in the Summer of 2019 to pursue his newly found interest in stormwater management. Duwane credits his excitement for the field to the enthusiastic teaching style of Dr. William Hunt, who he says not only brought the field to life during a stormwater course, but also gave it a deeper sense of purpose. Upon graduation, Duwane plans to pursue a career in stormwater/ water resource management.

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1.1: Introduction

One of the most overt effects of urban land development is the change in the hydrology of a watershed. The removal of vegetation for the construction of roadways, buildings, and other impervious urban infrastructure lessens the infiltration and evapotranspiration of stormwater, which causes considerable increases in runoff generation during storm events (Herald, 2003; Chen et al., 2017). The resulting hydrologic changes throughout a watershed can be depicted by comparing pre- and post-development stormwater hydrographs (Figure 1-1). The post-development hydrograph is noticeably taller and narrower than that of pre-development, indicating greater runoff volumes, higher peak flow rates, and shorter time durations to the peak flow rate. The impact on ambient water bodies receiving urban runoff can be detrimental. The effects include: (1) accelerated channel erosion, (2) changes to channel morphology and hydraulics, and (3) damage to aquatic habitat (Walsh et al., 2005).

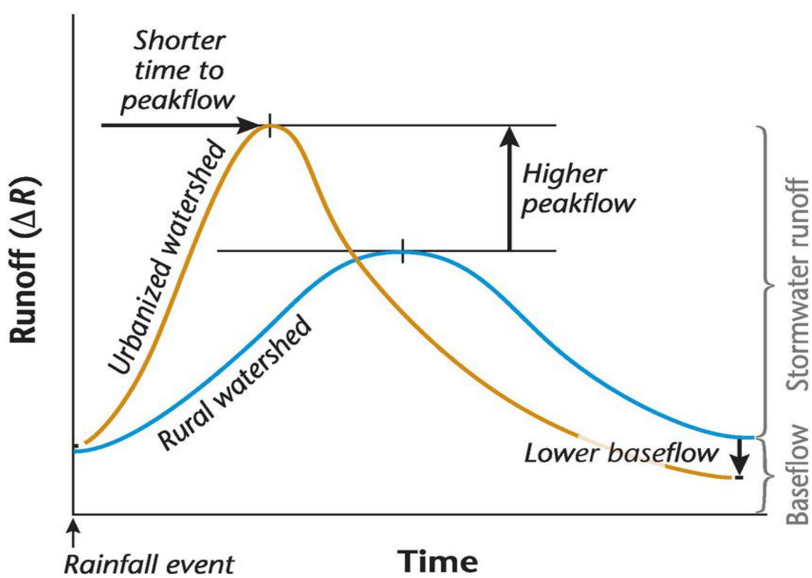

Figure 1-1. Hydrograph depicting the differences in pre-development versus post-development stormwater discharge. Source: Cambridge University Press.

Furthermore, as runoff drains to ambient water bodies, it transports pollutants that accumulate on urban infrastructure such as sediment, heavy metals, nutrients, oils, and greases (Herald, 2003; Hatt et al., 2004; Valtanen et al., 2014; Vogel and Moore, 2016). These pollutants are largely amassed from anthropogenic activities including construction (e.g., debris and soil displacement), agriculture (e.g., fertilizers and insecticides), industrial processes (e.g., coal burning emissions and waste effluent), and transportation (e.g., vehicular exhaust, tire and brake wear, and road degradation) (Makepeace et al., 1995; Wik et al., 2008; Barbosa et al., 2012). Polluted runoff degrades the quality of many water bodies throughout the United States (USEPA, 2020a). The National Summary of State Information, which summarizes the conditions of surface waters as reported by each state, lists urban runoff as the second largest source of impairment of 'bays & estuaries', 'coastal shoreline', and among the top ten contributors to the impairment of 'rivers & streams', 'lakes, reservoirs, and ponds', 'oceans and near coastal', and 'Great Lakes shoreline'. Receiving waters are susceptible to a range of adverse effects including accelerated rates of eutrophication, disruption of aquatic habitats and productivity, discoloration, odor, among other effects (Figure 1-2) (Aryal et al., 2010; Walsh et al., 2016).

Figure 1-2. Effects of polluted ambient waters: Fish kills in the Neuse River (left) (Courtesy newbernsj.com) and algal blooms in the Chowman River (right) (Courtesy coastalreview.org).

Stormwater runoff is also a significant source of pathogens (i.e., disease-causing microorganisms) to receiving waters (Yau et al., 2014; Fongaro et al., 2015). Pathogens are primarily introduced to stormwater via direct contact with fecal matter from humans and warm-blooded animals (both domesticated and wild) or by mixing with pathogen-laden wastewater in combined sewer systems (USEPA, 2001; Arnone and Walling, 2007). Researchers have recognized that increases in the impervious area of a watershed also coincides with greater influxes of pathogenic microbes to surface waters (McCarthy et al. 2007; Krometis et al., 2009; Chelsea Nagy et al., 2011). Hence, urban runoff flowing to recreational waters can pose a significant threat to the health and safety of water users. Studies have highlighted that recreational water use in close proximity to urban stormwater outfalls has resulted in higher rates of waterborne illnesses (Haile et al., 1999; Gaffield et al., 2003). These findings underscore the importance of urban stormwater management, particularly near recreational waters.

In the United States, federal regulation of stormwater is administered through the National Pollutant Discharge Elimination System (NPDES) stormwater program. The NPDES provides stipulations for stormwater discharge from three sources: (1) Municipal Separate Storm Sewer Systems (MS4), (2) construction sites, and (3) industrial activities. Most states, tribes, and territorial governments are delegated the responsibility of permitting, administrating, and enforcing aspects of the regulation (USEPA, 2020b). Permit holders are mandated to execute stormwater management plans that adhere to the program's standards to avoid monetary, legal, or other forms of penalties (USEPA, 2020c).

Compliance with stormwater management regulations often involves the use of approved stormwater control measures (SCMs) to curb the hydrologic and water quality impact of urban runoff. SCMs, which include bioretention cells, wet ponds, permeable pavements, among other systems, treat runoff through physical, chemical, and biological mechanisms such as sorption, filtration, detention, sedimentation, and biological uptake (NCDEQ, 2017). Researchers have demonstrated that SCMs can effectively manage urban runoff by reducing runoff volume, peak flow rate, and pollutant loads prior to release into ambient waters (e.g., Hathaway et al., 2009; Davis et al., 2010; Volgel and Moore, 2016). SCMs can also be used to shift the hydrology of an urban watershed closer to pre-development conditions; a concept referred to as Low Impact Development (Echart et al., 2017).

Proprietary SCMs, which are also referred to as manufactured treatment devices (MTDs), are being increasingly utilized for stormwater management in urban environments. These systems maintain several advantages over non-proprietary SCMs, as they: (1) are more cost effective to implement, (2) have smaller footprints, (3) are more customizable to target specific pollutants, (4) are more conveniently incorporated in tandem, and (5) provide dual land use if located underground (NCDENR, 2012). Having a small footprint makes MTDs particularly desirable in urban and ultra-urban environments where siting larger non-proprietary SCMs can be difficult due to the cost and limited availability of land. Notable drawbacks, however, include more frequent inspection and maintenance, as well as difficulty in identifying and diagnosing failure in underground systems (NCDENR, 2012).

Although MTDs are expected to provide effective stormwater treatment, few peer-reviewed publications exist regarding their performance. Most performance evaluations are limited to manufacturer testing, which lack independent, third-party scrutiny. Fewer still publications document MTDs' ability to remove pathogens. Given MTDs are increasingly utilized to manage urban runoff, exploring their potential to mitigate pathogens may help to inform the decision making of land developers and regulatory agencies. This review explores pathogenic microorganisms in urban runoff and their potential removal by MTDs.

1.2: Pathogenic Microorganisms

1.2.1: The Impact of Pathogens in Recreational Waters

Various point and nonpoint sources introduce pathogens to lakes, rivers, beaches, and other recreational surface waters, often to levels that can be harmful to water users. In addition to urban runoff, pathogens can be introduced to recreational waters from (1) treated or untreated wastewater outfalls, (2) dry and wet atmospheric deposition of airborne pathogens (i.e., pathogens adhered to dust particulates sourced from plant surfaces and soils), (3) direct fecal deposition of wildlife (e.g., waterfowl), (4) agricultural runoff, and (5) septic tank leakage (Geldreich, 1996; Pandey et al., 2014; Weerasundara et al., 2017). Regardless of the source, the diversity and abundance of pathogenic microbes are largely dependent on the microbial community and physicochemical conditions (e.g., pH, temperature, turbidity) of the receiving water, which can either encourage or hinder their survival (Gupta et al., 2017).

Many epidemiological studies have documented outbreaks of illnesses stemming from recreational water use (Marion et al. 2010; Wade et al. 2012; Papastergiou et al. 2012; Arnold et al. 2013; Graciaa et al., 2018). The most common etiologic agents responsible for outbreaks are bacteria (*Escherichia Coli* serotypes, *Legionella* spp., *Campylobacter* spp., *Salmonella* spp., *Shigella* spp., and *Pseudomonas* spp.), viruses (norovirus, adenovirus), and protozoa (*cryptosporidium* spp. and *Giardia* spp.) (Korajkic et al., 2018). Pathogens infect humans during recreational activities via skin contact or accidental ingestion of water. Consumption of contaminated seafood can also cause infection (Arnone and Walling, 2007). Once infected, common health effects include gastrointestinal illness (i.e., diarrhea & vomiting), infection of the ears, eyes, skin, or respiratory system (Geldreich, 1996; Yamahara et al., 2012).

There is also an economic burden associated with waterborne illnesses associated with pathogens and recreational water bodies. DeFlorio-Baker et al. (2018) estimated that in the United States, costs related to purchasing medication, visiting health care providers and hospitals, and indirect costs of time away from work, can range between \$2.2-\$3.7 billion annually. Coastal tourism, which makes up 85% of all tourism-related revenue in the United States (Dorfman et al., 2007), is also negatively impacted by pathogen-related beach closures. Nearly 40% of coastal and Great Lake “swimming” beaches had at least one advisory or closure in 2018 due to elevated concentrations of pathogen indicators (USEPA, 2019a).

1.2.2: Monitoring Pathogens: Fecal Indicator Bacteria

Determining the concentration of pathogenic microbes in recreational waters is necessary for assessing potential health risks and determining adherence to water quality regulations.

Enumerating individual pathogens, however, is operationally complicated, costly, and ultimately impractical. Instead, regulatory agencies use representative fecal indicator bacteria (FIB) to signify the presence of pathogens in fresh and marine waters (USEPA, 2012). While not themselves pathogenic to humans, these indicators are abundant in the intestines of warm-blooded animals, and their presence indicates recent fecal contamination and potential accompanying pathogens. The use of FIB is justified by studies that observe significant correlations between their presence and illnesses at sampling sites (Marion et al., 2010; USEPA, 2012). Furthermore, these indicator bacteria have characteristics that are desirable for monitoring, namely: high survival rates, easy detection, present in large quantities, and economical sustainability (USEPA, 2012). However, some researchers have shared concerns with the use of the current regulatory FIB (i.e., *Escherichia Coli* and *enterococci*) to indicate pathogen presence. Researchers highlight that these indicators are not representative of all pathogens, noting waterborne disease outbreaks in the waters where FIB are absent (Papastergiou et al. 2012; Arnold et al., 2013). Researchers also suggest that relying exclusively on FIB ignores pathogens that are nonhuman and non-enteric in origin and thus cannot represent all pathogen groups (Makepeace et al., 1995).

There are two main methods of quantifying FIB in water samples: a membrane filtration method and liquid broth test. In the membrane filtration method, samples are filtered and incubated at a specific temperature, after which the emerging bacterial colonies are counted [reported as colony forming units (CFU)]. In the liquid broth test, samples are placed in tubes containing a nutrient broth and incubated, after which an index is used to determine the most probable number (MPN) of bacteria based on the release of gas in the tubes (Meyers et al., 2007; USEPA, 2012).

1.2.3: Water Quality Regulations for Pathogens

Water quality regulations for pathogens in recreational and bathing waters were first developed on a federal level by the National Technical Advisory Committee (NTAC) in 1968. The fecal coliform (FC) concentrations for primary contact (i.e., swimming, surfing, etc.) recreational waters were not to exceed a log mean of 200 CFU/100 mL for a minimum of five samples taken within a 30-day period. Additionally, ten percent of all samples should not exceed 400 CFU/100 mL FC during any 30-day period (USEPA, 1986). In 1986, the regulations were updated. For freshwater, the geometric mean (GM) concentration of *Escherichia Coli* (*E. Coli*) and *enterococcus* were not to exceed 126 CFU/100 mL and 33 CFU/100 mL, respectively. For marine waters, the GM concentration of *enterococcus* was not to exceed 35 CFU/100 mL (USEPA, 1986). In 2012, the water quality criteria were again updated to include both the GM (50th percentile of expected water quality distribution) and a statistical threshold value (STV), (90th percentile of expected water quality distribution) (Table 1-1). The STV was incorporated to account for variability in indicator bacteria concentrations, which in conjunction with GM, is thought to be more protective of water quality. The new criteria also provide two sets of recommendations, which allows states to decide which set of values is more appropriate based on their illness rates (Table 1-1) (USEPA, 2012).

Table 1-1. Current water quality criteria for full-body contact in recreational waters ‡

Indicator Species	Recommendation 1		Recommendation 2	
	GM[a]	STV[a]	GM[a]	STV[a]
E. Coli (fresh waters)	126	410	100	320
Enterococci (fresh & marine waters)	35	130	30	110

‡: "The waterbody geometric mean should not be greater than the selected geometric mean magnitude in any 30-day interval. There should not be greater than a ten percent excursion frequency of the selected STV magnitude in the same 30-day interval."
[a] - Units: CFU/ 100 mL

Furthermore, under the NPDES permit program, point source discharges leading to impaired water bodies, which include stormwater sewers, are subjected to pollutant loading limits referred to as Total Maximum Daily Loads (TMDLs) (USEPA, 2020b; 2020c). States develop TMDLs to specify pollutant waste loads that a water body can receive to achieve and maintain water quality standards (USEPA, 2019b). FIBs are among the pollutants subjected to TMDLs.

1.3: Mechanism of Pathogen Treatment in SCMs

Managing the concentration of pathogens in urban runoff is imperative to minimize the risk of water borne illnesses for recreational water users, safeguard economic value of a water body, and to protect aquatic ecosystems. Makepeace et al. (1995) reported that typical concentrations for *E. Coli* and *enterococci* in urban runoff range from 12 - 4700 CFU/100 mL and 120 – 340,000 CFU/100 mL, respectively, which evinces that pathogenic concentrations can be significantly higher than water quality criteria. Researchers have investigated the use of SCMs to mitigate FIB concentrations in stormwater and have concluded that many of the treatment mechanisms employed by SCMs can remove and/or inactivate pathogens before releasing runoff into ambient water bodies. These mechanisms, as described in peer-reviewed journal articles, are presented in subsequent sections.

1.3.1: Filtration

Filtration is a physical process that involves the passing of water through a filter medium such as engineered media or a geotextile membrane. The efficiency of a filter medium for pathogen removal is a function of the rate of collision, the attractive forces of the medium's surface, and adhesiveness of the microbe's membranes (Bozorg et al., 2015). Filtration efficiency is also impacted by media texture, media content, water velocity, proportion of microbes already adsorbed to particles, and the presence of antimicrobial agents (Meinders et al., 1995; Stevik et al., 2004; Li et al. 2014). If the pore size of the medium is small enough, pathogens can also be retained in media via straining (Stevik et al., 2004). The presence of biofilm reduces porosity, which increases the rate of collision and straining (Bozorg et al., 2015), but can, however, assist in pathogen survival and resistance to predators (Fraser et al., 2018). Removal efficiency can also be significantly impacted by operating conditions such as stormwater overloading, clogging, short circuiting, and proliferation of captured microorganisms (Peng et al., 2016).

Once captured, the survival of the microbes depends on several conditions including availability of nutrients, media moisture content (desiccation may contribute to increased die-off rate), and microbial community (i.e., competition and predation) (Zhang et al., 2011; Chandresena et al., 2012; Haig et al., 2015). It is possible for persisting pathogens to be desorbed or resuspended into the water column during subsequent storm events, which in some instances can lead to net increases in outflow concentrations of pathogens (Chandresena et al., 2012).

1.3.2: Sedimentation

The removal of solids from a water column via settling is referred to as sedimentation. Some microorganisms, such as fecal coliform, helminths, and fecal streptococci, are more efficiently removed owing to their higher settling velocities compared to other microorganisms (Alexandros & Akratos, 2016). Although pathogens settle on their own naturally, their partitioning behavior plays a critical role in the sedimentation process (Characklis et al., 2005). As they adsorb to algae, particulate organic matter, and other particulate matter in the water column (typically ranging 0.45 μm to 30 μm), their overall density and settling velocity can be substantially increased (Burkhardt III et al., 2000; Jeng et al., 2005). Once settled, pathogen survival can vary based on the environmental stressors (e.g., temperature and availability of nutrients) (Pachepsky & Shelton, 2011). The resuspension of settled microbes is also possible during turbulent conditions (Brookes et al., 2004).

1.3.3: Adsorption

Adsorption of microbes can be explained by the double layer theory (DLVO). Microbes have a pH-dependent surface charge in water, which influences their mobility and colloidal behavior. Electrostatic interactions between the microbes and the media or other charged surfaces can form weak bonds (Stevik et al., 2004; Zhang et al., 2010). The isoelectric point of the media, which refers to the pH that coincides with no net electrical charge, plays a crucial role in adsorption. Media with higher isoelectric points, such as clay, would see better adsorption than media with lower isoelectric points, such as sand (Ferguson et al., 2003). Adsorption is also dependent on physical, chemical, and microbiological factors of media such as the media's content, cation exchange capacity, pH, presence of biofilm, temperature, abundance of organic matter, and water velocity (Ferguson et al., 2003; Stevik et al., 2004). Adsorption can increase a pathogen’s susceptibility to the removal processes of filtration and sedimentation but can also increase resilience to inactivation because of increased protection from extracellular enzymes, toxins, and photoinactivation (Ferguson et al., 2003).

1.3.4: Phytoremediation

Phytoremediation can be described as the use of plants to degrade, extract, or immobilize pollutants in soil and water (Ruby & Appleton, 2010). Plant roots secrete antimicrobial compounds, such as coumaric, rosmarinic, and ferulic acid, to protect themselves from diseases. Antimicrobial compounds are also stored in leaves, seeds, and flowers, and can be released through tissue as they fall and decompose on media surfaces (Shirdashtzadeh et al., 2017). Researchers note that these compounds can increase the die-off rates of pathogenic indicators (Walker et al., 2003; Strehmel et al., 2014; Shirdashtzadeh et al., 2017). Plant root exudates also contain sugar, amino acids, organic acids, and oxygen, which promotes a highly diverse microbial community in the rhizosphere and forces pathogens to compete for nutrients (Rippy, 2015; Chandresena et al., 2017). Roots also provide additional surfaces for pathogen adsorption particularly if biofilm is present.

Some studies suggest that the effectiveness of phytoremediation varies with vegetation type. Rippy (2015) noted that some species of shrubs have exhibited higher removal of FIB than sedges. Chandresena et al. (2012) proposed that differences in removal efficiencies may be due to changes to the physical structure of soil (e.g., creation of macropores). Conversely, other researchers such as Alexandros et al., (2016), submit that plant species has no significant impact on pathogen removal. These contrasting findings imply that more work should be done to identify the impact of vegetation type on pathogen removal.

1.3.5: Photoinactivation

Exposure to sunlight is widely accepted by researchers as a very effective method of mitigating pathogens. Sunlight stimulates the production of Photo-Productive Reactive Intermediates (PPRI), which impairs cell function. The absorption of photons also damages the chemical structure of vital macromolecules such as DNA, proteins, and enzymes (Reed, 2004; Nelson et al., 2018). Photoexcitation of certain molecules also stimulates the production of oxidizing species, which have been observed to hamper pathogen viability (Ferreira et al., 2020). The wavelengths attributed to photoinactivation of pathogens ranges from 200 to 700 nm (Nelson et al., 2018). Pathogen inactivation efficiency is predicated on several factors including duration of exposure, the light intensity (corresponding to the time of day), as well as the susceptibility of the pathogen to ultraviolet light (varies between organisms) (Burkhardt III et al., 2000; Boehm et al., 2012). Notably, the physicochemical characteristics of runoff, including temperature, pH, and turbidity, can also impact effectiveness of photoinactivation (Reed, 2004; Nelson et al., 2018).

1.3.6: Microbial Community: Competition and Predation

Indigenous microorganisms in the soil or media of SCMs can also control the pathogenic population in stormwater via predation and competition (Wanjugi & Harwood, 2013). Nematodes and protozoa are two of the premier predators of pathogenic bacteria. Both ingest pathogens by phagocytosis, which involves the engulfment, digestion, and excretion of microbes (Rippy, 2015). Although grazing rates vary, these microorganisms can consume substantial quantities of microorganisms. A single protozoan, for example, can consume between 5 and 73 bacteria per hour in porous sediment (Rippy, 2015). The efficiency of predation depends on several factors including microbial diversity, community size (predation slows at lower population sizes), and proportion of particulate bound pathogens (some microbes are more adept at consuming free-phase pathogens) (Stevik et al., 2004; Zhang et al., 2010; Rippy, 2015). Researchers also recognize that some microorganisms are less vulnerable to predation than others. For instance, e*nterococci* have thick cell walls that prevent digestion by protists (Rippy, 2015).

Moreover, some microbes in a community are more persistent owing to their ability to compete for available nutrients. These microbes can: (1) adapt for survival by changing their metabolic activity, (2) excrete substances that repress other microbes, (3) produce spores (i.e., robust structures that support resilience in unfavorable conditions) in response to adverse conditions (Stevik et al., 2004). Nutrient availability, demand, consumption rates, ability to prevail over hostile interaction, and ability to efficiently consume nutrients, all contribute to establishing the predominance of microbial taxa (Rippy, 2015).

1.4: Manufactured Treatment Devices

<u>1.4.1: Design and Stormwater Treatment</u>

There are a variety of MTDs on the market. Vaulted filtration systems are among the most common types of MTDs (Figure 1-3). These are typically compact, rapid-flow-through devices that filter runoff using an engineered media or filter membrane. Engineered media are commonly housed is internal chambers, cylindrical cartridges, or other internal compartment within the system. Some vaulted filters incorporate vegetation, which typically consists of select trees, shrubs, or other plants that are: (1) known to provide effective sequestration of pollutants and (2) resilient enough to survive drastic and sudden changes in hydrologic conditions. Underground detention systems are also common (Figure 1-4). They utilize chambers, pipes, or other designated storage areas to facilitate sedimentation and peak flow reduction. These systems can also be configured to infiltrate runoff.

Stormwater runoff treatment in MTDs is design-dependent, but typically employs filtration and/or sedimentation. Filtration-based MTDs treat runoff as it passes through filter membranes or engineered media. Engineered media are typically composed of sand, perlite, zeolite, granular activated carbon, or a patented combination of multiple grains. Media can provide surfaces for the adsorption of pollutants and support vegetation to promote biological uptake of pollutants. Detention-based MTDs temporarily store runoff to provide: (1) hydrologic benefits such as peak flow and volume mitigation, and (2) water quality benefits by facilitating sedimentation. Like non-proprietary SCMs, MTDs can be configured to reduce runoff volume by exfiltration into native soil and/or by evapotranspiration, while peak flows can be mitigated by stormwater detention and controlled release via outlet structures.

Figure 1-3. Schematic of two vaulted MTDs: Contech's Filterra ® (left) is a vaulted biofiltration system and Advanced Drainage System's (ADS) BayFilter ® (right) is a vault cartridge filtration system. Source: NCDEQ, (2017).

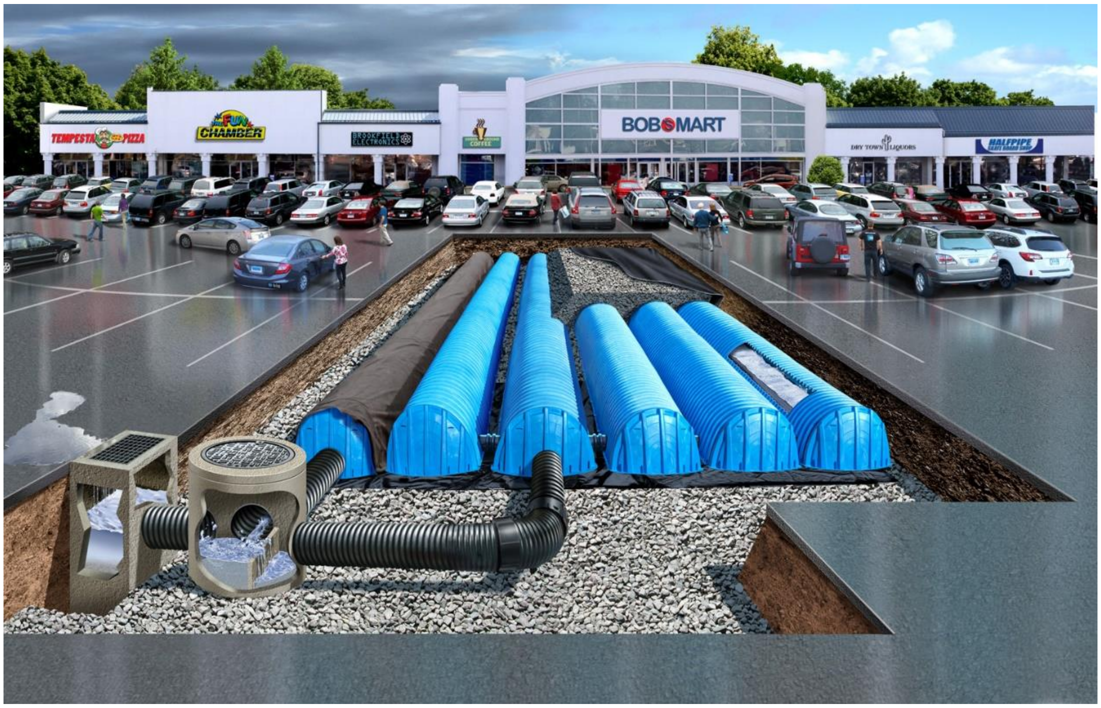

Figure 1-4. Schematic of a subsurface detention MTD: Cultec's Contactor® and Recharger®. (Image courtesy Cultec).

1.4.2: Evaluation of MTD Performance

There is currently no federal guidance on evaluating the treatment performance of MTDs. Several states have opted to develop protocols that can be used to certify the performance of emerging MTDs. The Technology Acceptance Reciprocity Partnership (TARP) and the Technology Assessment Protocol – Ecology (TAPE) are currently two of the most prominent evaluation programs (NJDEP, 2009; Sample et al., 2012; WSDE, 2018). North Carolina has developed the NEw Stormwater Technology (NEST) program to certify MTDs. All three programs detail standardized methods for data collection and testing, which are typically carried out by independent third-party entities.

Notably, these programs vary in their priorities; TARP prioritizes evaluating an MTD's ability to remove sediment and sediment-bound pollutants, TAPE focuses on sediment, dissolved metals, phosphorus and oils, while NEST focuses on sediment and nutrients (NJDEP, 2009;

NCDEQ, 2017; WSDE, 2018). Following approval, MTDs can be installed by site developers to meet water quality goals of the state (Table 1-2).

Table 1-2. Descriptions of MTDs approved for use in the state of North Carolina.

Practice	Company	Description	Treatment Mechanisms
BayFilter™	ADS	A cartridge system that features a spiral media configuration to treat stormwater in subsurface precast vaults, manholes or cast-in-place housing.	Filtration, Adsorption
Filterra™	Contech	A precast vaulted biofiltration system that provides high flow subsurface filtration of runoff through patented media.	Filtration, Adsorption, Biological uptake, Evapotranspiration
Stormfilter™	Contech	An underground cartridge filtration vault that treats stormwater using custom media held in various cartridge sizes and configurations.	Filtration, Adsorption
Silva Cell™	Deep Root	A modular soil containment system that facilitates an area of lightly compacted soil or media to primarily enhance/support tree growth, while also managing stormwater.	Filtration, Adsorption, Biological uptake, Evapotranspiration

Source: NCDEQ, 2017

1.5: Potential for Pollutant Mitigation in MTDs

1.5.1: Total Suspended Solids (TSS)

TSS is primarily removed from stormwater via filtration and sedimentation (Shammaa & Zhu, 2001). Detention-based MTDs are capable of effective sedimentation if designed with appropriate volume storage and hydraulic retention time (HRT). MTDs can remove TSS via filtration using engineered media and filter fabrics. Although peer-refereed publications on TSS removal in MTDs are limited, Smolek et al. (2018) observed effective TSS mitigation in a

Filterra® biofiltration system. They reported mean and median reductions in TSS of 92% and 94%, respectively. Page et al. (2015) evaluated the performance of Deeproot's Silva Cell® and observed mean reduction in TSS of 86%. However, it is worth noting that because of the compact footprint that is common among most MTDs, filtration efficiency can quickly deteriorate due to clogging (Smolek et al., 2018). Consequently, routine maintenance is needed to maintain effective treatment.

1.5.2: Nutrients

Nitrogen and phosphorus are the main nutrients of concern in stormwater runoff (USEPA, 1999). Like with TSS, particulate forms of nitrogen and phosphorus are primarily removed from stormwater via filtration and sedimentation. Sample et al. (2012) notes that MTDs that remove particles of sizes less than 50 µm will likely observe reliable removal of particulate nutrients. However, most MTDs lack the mechanisms necessary for the removal of dissolved nutrients (i.e., biological uptake, denitrification, etc.). Payne et al. (2014) acknowledges that vegetated systems are able to sequester dissolved nutrients, but since there is generally a limited amount of vegetation employed by MTDs, it is likely that MTD vegetation would have minimal impact on nutrient removal.

Smolek et al. (2018) observed total nitrogen (TN) and total phosphorus (TP) reductions of 40% and 55%, respectively, when evaluating a Filterra® biofiltration system. The authors attributed reductions in TP to the removal of particulate forms of TP, and reductions in TN to the removal of organic nitrogen, a mostly sediment-bound form of nitrogen.

1.5.3: Pathogens

Davis et al. (2010) likened the removal of pathogens in SCMs to the removal of particulate matter, suggesting that filtration and sedimentation are primary treatment mechanisms. Although MTDs generally employ filtration and sedimentation, it is worth noting that several key design elements hinder pathogen sequestration: (1) subsurface designs prohibit photoinactivation, (2) rapid-flow through devices have minimal HRT, which hinders FIB capture in media (Kim et al., 2012), (3) compact vegetated MTDs generally have few plants and employ a fraction of the media volume used in non-proprietary SCMs, which limits the amount of phytoremediation (i.e., rhizospheres) and the cumulative surface area available for adsorption. Ultimately, these factors suggest that typically designed MTDs are not optimized for pathogen removal and likely rely on their proficiency for filtration and sedimentation.

Minimal peer-reviewed research evaluating FIB treatment in MTDs exists. Schifman et al. (2016) reported significant removal of *E. Coli* (ranging from 71% to 89%) in two proprietary box tree filters. Hathaway et al. (2009) evaluated *E. Coli* and FC removal from three sedimentation-based MTDs and found that only one system demonstrated effective removal of FC, while the others had net increases in concentrations in both FIB. A synthesis of this literature suggests that filtration-based MTDs would likely remove more FIB’s than sedimentation-based MTDs. One explanation could be that most sedimentation-based MTDs lack the necessary HRT for effective FIB removal. More peer-reviewed studies in this realm are needed.

1.6: Conclusion

MTDs are emerging as capable SCMs for urban stormwater management. Based on the treatment mechanisms they employ, MTDs should generally mitigate TSS and particulate-bound pollutants, which include some fraction of nutrients and pathogens. However, limited peer-refereed publications evaluate how MTDs treat water quality. More field evaluations of the hydrologic and water quality performance are needed to better understand MTD capabilities. Future work should add to the body of research on how MTDs perform in the field, particularly for pathogen removal.

This review also highlights some of the complexities and importance of pathogen mitigation in stormwater. While several pathogen removal mechanisms have been identified in research, there still needs to be an understanding of how these mechanisms can be employed to make SCMs more effective. Future research should not only expand on the body of knowledge regarding pathogen removal in SCMs, but also identify which SCMs may be most effective for stormwater treatment.

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CHAPTER 2: Evaluating the Hydrological and Water Quality Treatment Performance of StormTech's Subsurface Chamber System in Raleigh, NC

2.1: Abstract

Stormwater runoff from urban infrastructure is a leading cause of natural waterbody degradation in the United States. Generally, stormwater control measures (SCMs) are used to curb the impact of urban runoff by reducing runoff volume, peak flow rate, and pollutant load. Proprietary SCMs are a broad category of manufactured SCMs that treat runoff using many of the mechanisms utilized by nature-based SCMs. However, unlike most SCMs, proprietary stormwater control devices are designed to treat runoff while occupying little or no usable land area. This makes them particularly suited for implementation in urban and ultra-urban environments where space is limited and costly. StormTech® has designed and patented the subsurface chamber system (SCS) - an underground detention system that treats runoff by employing filtration, infiltration, and sedimentation. While seemingly promising, the performance of this system in North Carolina has yet to be detailed in peer-refereed publications. Using monitoring standards established for NC's NEw Stormwater Technology (NEST) program, this study evaluated the hydrologic and water quality treatment ability of a SCS installed in Raleigh, NC. The SCS was subjected to 73 precipitation events ranging 4.32 mm - 92.71 mm (0.17 in - 3.65 in) over a 14-month study period. The system was able to effectively reduce runoff volume and peak flows attaining median event reductions of 74% and 96%, respectively, and was able to infiltrate approximately 43% of the runoff entering the system. Of the monitored events, 16 were sampled for water quality analysis, which demonstrated that the system provided effective pollutant removal. Total suspended solids (TSS) concentrations were significantly reduced by a median of 91% - generating effluent concentrations that bested NC standards (median effluent EMC of 3.22 mg/L). Total nitrogen (TN) and total phosphorus (TP)

concentrations were significantly lowered by median values of 35% and 68%, respectively. Nutrient loads were also significantly reduced, meeting the thresholds for nutrient sensitive watersheds in Piedmont NC. TP loads were reduced by 81% to 0.1 kg/ha/yr and TN loads were lowered by 65% to 2.1 kg/ha/yr. These results suggest that the SCS can be a viable option for stormwater management in North Carolina.

2.2: Introduction

Urbanization alters the natural hydrology of a watershed by replacing portions of permeable, vegetative cover with impervious surfaces (Klein, 1979; Booth et al., 1997; Hsieh & Davis, 2005; Volgel & Moore, 2016). The resulting increase in runoff generation often overwhelms stormwater drainage systems, causing floods, damaging infrastructure, and threatening public safety (Burian & Finlay, 2002). Furthermore, as drainage systems convey runoff to waterways, they increase the water's velocity and erosive power, resulting in aquatic ecosystem disruption due to accelerated streambank erosion, altered streambed composition, and other geomorphic modifications (Barbosa et al., 2012). Impervious surfaces also limit the degradation potential of pollutants (Aryal et al., 2010; Passeport and Hunt, 2009), resulting in more pollutants being mobilized and draining to natural waterways during a storm event. This stimulates algal production, reduces the abundance and diversity of benthic macroinvertebrates, and suppresses biomass and aquatic diversity, among other aquatic ecosystem impacts (Fletcher et al., 2014).

Strategies for addressing stormwater management are informed by federal and state regulations. In North Carolina, the North Carolina Department of Environmental Quality (NCDEQ) is responsible for enforcing stormwater management strategies. To meet stormwater management goals, NCDEQ promulgates the use of stormwater control measures (SCMs). SCMs are systems engineered to treat stormwater through various pollutant removal mechanisms including filtration, adsorption, and sedimentation (Bratieres et al., 2008). Some SCMs are also designed to reduce peak flow discharges, attenuate stormwater volume, as well as promote evaporation, infiltration, and groundwater recharge with the goal of mimicking pre-development conditions (Bell et al., 2016). Commonly used SCMs include constructed stormwater wetlands, wet ponds, bioretention cells, biofilters, and sand filters.

Private companies also engineer and patent manufactured SCMs, which are commonly referred to as Manufactured Treatment Devices (MTDs). While MTDs vary widely in size, configuration, and application, they incorporate many of the same fundamental treatment mechanisms that are used in nature-based SCMs (Smolek et al., 2018). MTDs are typically housed in precast chambers, embedded in underground cavities, or installed as attachments to existing manholes, catch-basins, or curb inlets. MTD configurations and treatment mechanisms are tailored to the expected hydrologic and water quality characteristics of a specific watershed and their intended application (i.e., residential, commercial, industrial, or municipal).

Commonly prioritized among most practices is the removal of sediment, particulate matter, trash and debris through filtration. Filtration is generally implemented via membranes or engineered media. Filter membranes are commonly made of geotextile fabrics, while engineered media are composed of sand, perlite, zeolite, granular activated carbon (GAC), or some, likely patented, combination of a few materials. Some MTDs are also designed to mitigate peak flow and runoff volume. Like nature-based SCMs, runoff volume is reduced by exfiltration into native soil and/or by evapotranspiration. Peak flows are mitigated by stormwater detention in media void space or other designated storage space and subsequent slow release through flow-controlled outlet structures. Most manufactured practices also incorporate provisions for internal or external stormwater bypass utilized during large storm events.

MTDs are gaining popularity as viable stormwater treatment options owing to several advantages they maintain over traditional SCMs. These products can be less expensive to implement (when considering land costs), smaller in size, more customizable to target specific pollutants, more conveniently incorporated in tandem, and can allow dual use of the land if the system is located underground (NCDENR, 2012). Having a smaller footprint improves feasibility in urban and ultra-urban areas where land is costly and limited (Smolek et al., 2018). Notable drawbacks, however, include more frequent inspection and maintenance for smaller systems and difficulty in identifying and diagnosing failure in underground systems (NCDENR, 2012).

StormTech's® patented subsurface chamber system (SCS) was examined herein (Figure 2-1). The SCS is an underground detention system that is typically manufactured for municipal, industrial, recreational, residential, or commercial use under parking lots and roadways (StormTech, 2014). It detains runoff in rows of open bottom chambers and in the void space of the surrounding layers of stone. The SCS can be configured to infiltrate water into underlying soil to increase volume mitigation and peak flow reduction. The SCS benefits from having a geotextile wrapped chamber referred to as an isolator row®, which acts as a pre-treatment filtration chamber by capturing and treating much of the first flush volume. This in turn reduces clogging and preserves void space in the other chambers to prolong system infiltration (StormTech, 2014).

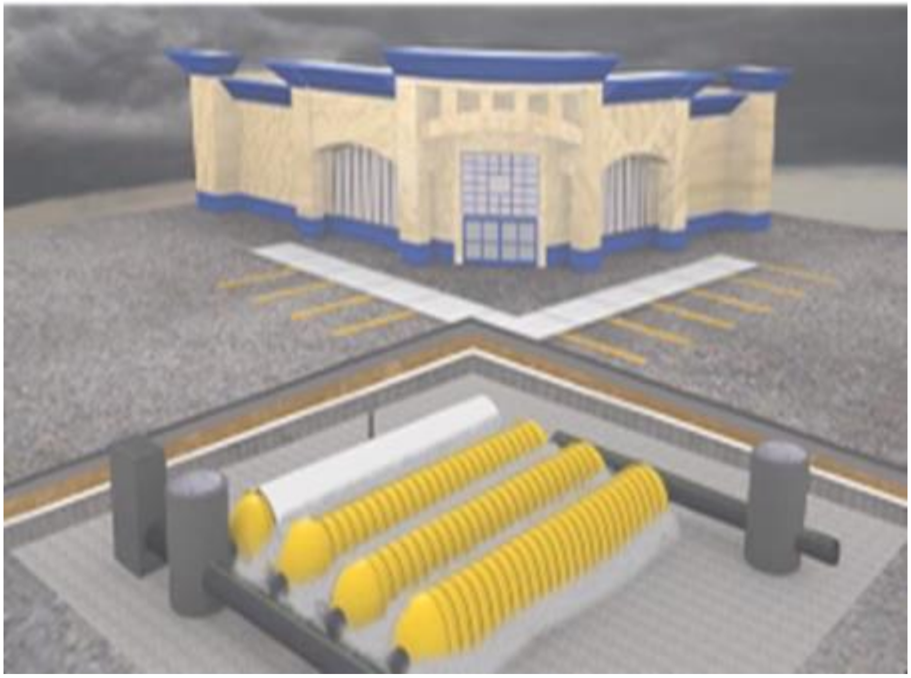

Figure 2-1. Schematic of StormTech's ® SCS and isolator row (in white) (Image courtesy of Advanced Drainage Systems).

To be approved for use in North Carolina, MTDs must adhere to NCDEQ's NEw Stormwater Technology (NEST) program requirements. Currently approved MTDs include Deeproot's Silva Cell™, Contech's Stormfilter™, Contech's Filterra™, and Advanced Drainage Systems' Bayfilter™ (NCDEQ, 2018). The performance of StormTech's® SCS has yet to be independently evaluated and reported on in peer-refereed articles. It is hypothesized that based on the treatment mechanisms it employs (i.e., detention, filtration, and sedimentation), the SCS should provide effective water quality and hydrologic treatment of urban stormwater runoff. The research goal herein is to assess an SCS's hydrologic and pollutant mitigation capabilities in accordance with the NEST program.

2.3: Methodology

2.3.1: Site Description

StormTech's subsurface chamber system was installed in the summer of 2014 beneath a parking lot located at Capital Oaks senior-living housing complex in Raleigh, NC (Figure 2-2, Table 2-1). Raleigh, NC, receives an average annual rainfall depth of 1092 mm (43.1 in), which is relatively well distributed seasonally. Raleigh, NC, has an average annual temperature of 16 ^{o}C (60.8 ^{o}F) (NOAA, 2020).

The watershed area draining to the system totals 1.46 ha (3.61 ac) and consists of approximately 0.29 hectares (0.74 acres) of paved surface, 0.49 ha (1.23 ac) of roof area, and 0.68 ha (1.63 ac) of unpaved/grassed area. The watershed was approximately 55% impervious. According to USDA's Web Soil Survey (2020), the underlying soil on site was hydrologic soil group (HSG) B, signifying soils that have "moderately fine to moderately coarse texture" with "moderate infiltration when moderately wet". Treated stormwater is piped to a small creek that flows along the northern boundary of the property.

Table 2-1. Site Description of SCS in Raleigh, NC.

Characteristic	SCS
Year Installed	2014
Location	35° 52' 5.664" N 78° 41' 46.32" W
Drainage Area (hectares)	1.46
Watershed Composition	Multi-family Residential
Imperviousness (%)	55
SCS Chamber Size	MC-4500
SCS Area (m^2)	469
SCS Storage Volume (m^3)	593
Underlying Soil Type	Sandy Loam (HSG-B)

- Per StormTech®

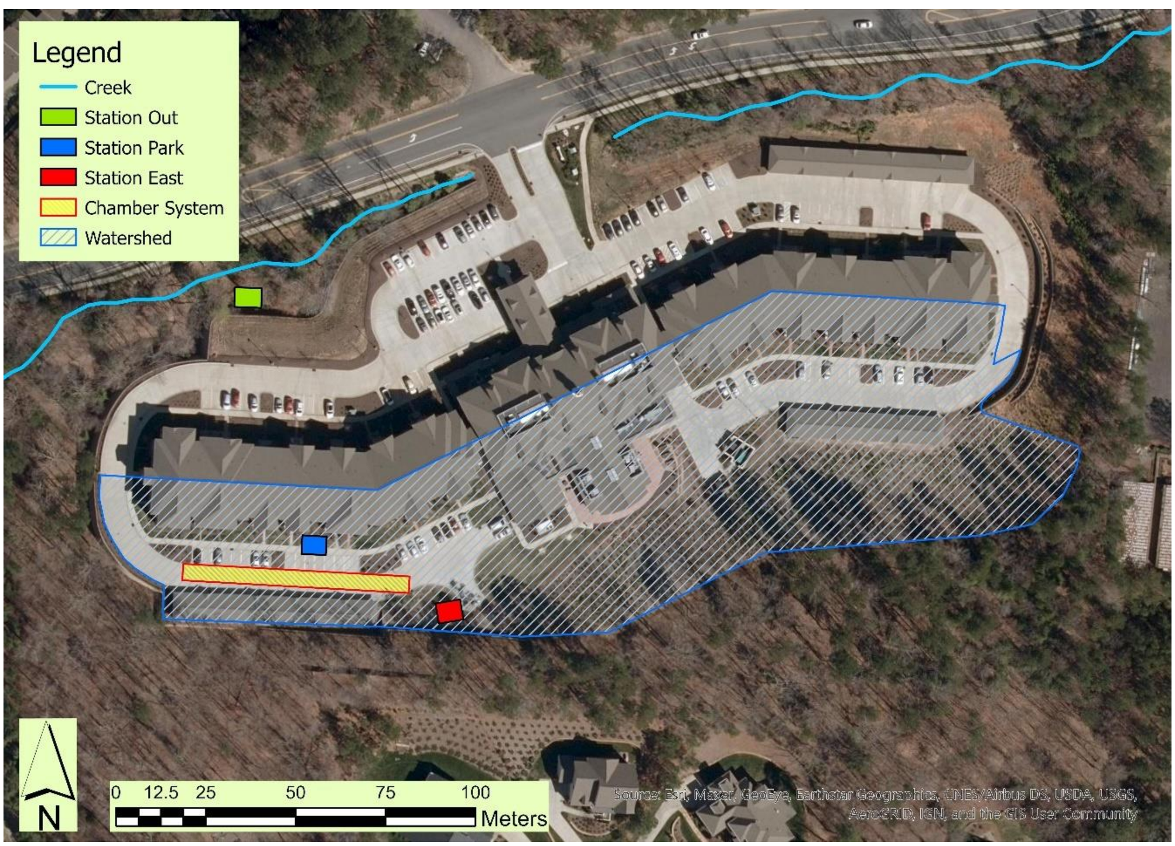

Figure 2-2. Location of StormTech's SCS and monitoring stations at Capital Oaks retirement community.

2.3.2: SCS Configuration

The base of the system comprised 230 mm (9 in) of clean, crushed, compacted angular stone (AASHTO M43: #3, #4) that sat atop one layer of nonwoven geotextile (ADS geosynthetic 601T) lining the excavated area (Figure 2-3). The foundation stone that aligns with the immediate inlet of each chamber was covered with one layer of woven geotextile (ADS geosynthetic 315WTM) to provide scour protection. Four open-bottom, polypropylene chambers and one isolator row lie on the stone-base and are covered with 305 mm (12 in) of clean, crushed, angular cover stone (AASHTO M43: #3, #4). The cover stone was naturally consolidated to maintain an estimated 40% void space for runoff storage. Nonwoven geotextile fabric covered the top of the cover stone to mark the upper limit of the storage area. As installed, the footprint of the chambers and isolator row cover 53 m x 9 m (173 ft x 29 ft, LxW) for a total area of 469 m^2 (5,047 ft^2). The SCS's storage capacity (including the perimeter stone volume) is listed as 593 m^3 (20,900 CF).

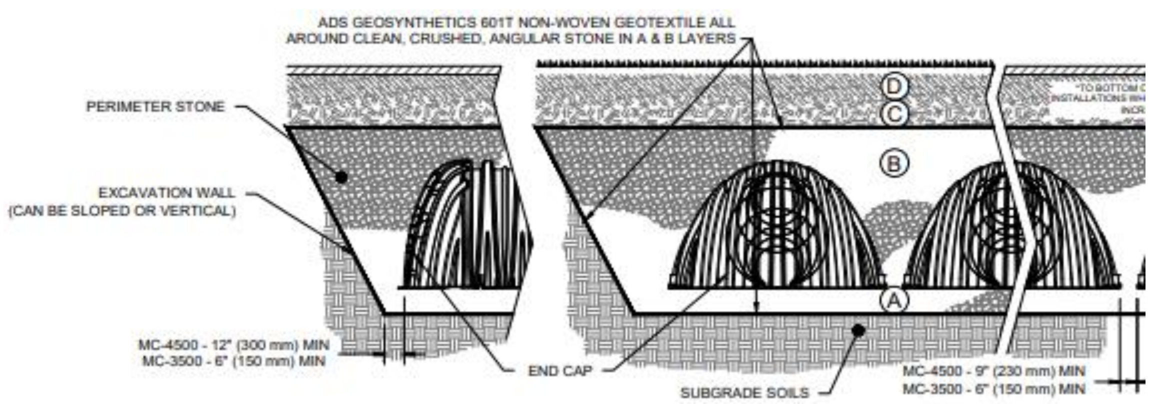

Figure 2-3. Schematic of structural cross-section of the SCS (Image courtesy StormTech®).

Runoff was piped into the SCS via a 380-mm (15-in) High Density Polyethylene (HDPE) pipe and a 610-mm (24-in) HDPE pipe referred to as inlet ‘Park’ and inlet ‘East’, respectively. Inlet ‘Park’ received mainly roof and parking lot runoff generated from the western portion of the watershed, while inlet ‘East’ receives runoff generated from the roof, parking lot, and unpaved/grassed area on the eastern portion of the watershed (Figure 2-4). The system is also equipped with a 150-mm (6-in) HDPE perforated underdrain located at the invert of the foundation bed, which drains to a flow-controlling outlet structure located on the western end of the system. Stormwater exits the outlet structure via a 760-mm (30-in) HDPE pipe that is referred to as ‘Out’ in later monitoring discussions.

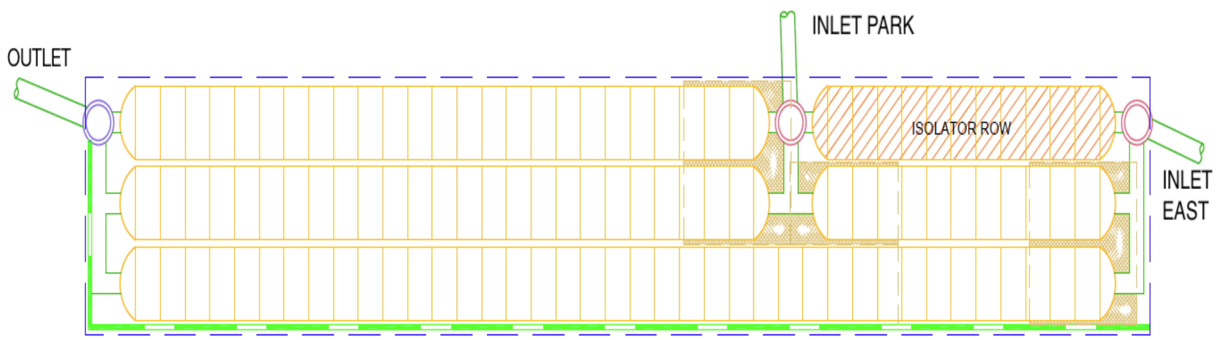

Figure 2-4. Schematic of Isolator Row and chamber configuration (Image courtesy StormTech®).

2.3.3: System Operation

During a storm event, the first flush volume, which is the runoff generated during initial stages of a storm event (generally consisting of a disproportionately high pollutant concentration, Sansalone and Cristina, 2004), flows directly into the isolator row from the catchment basin. The geotextile fabric that lines the isolator row filters runoff before it percolates to the underdrains. Once ponding in the isolator row exceeds 0.53-m (1.73-ft), stormwater flows to the remaining chambers via 610-mm (24-in) bypass manifolds. Inflowing stormwater is stored in the chambers’ cavity and in the surrounding void space of foundation and cover stone. Stored runoff is drawn down by the underdrain, infiltrating into underlying soil, or via 610-mm (24-in) outlet manifolds located at the end of each chamber. The underdrain and outlet manifolds flow directly to the outlet structure (Figure 2-5). A 1.77-m (5.8-ft) weir wall equipped with 6 orifices is located within the outlet structure to facilitate the controlled release of stormwater from the SCM. During large storm events, the SCS provides internal bypass as excess runoff volumes overflow the weir wall and flow directly to the outlet pipe.

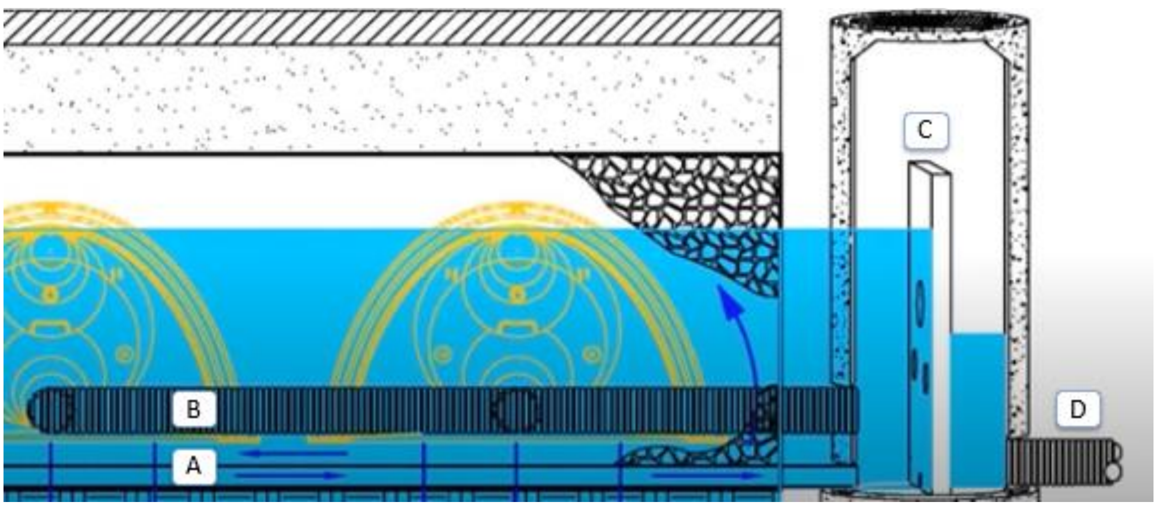

Figure 2-5. Schematic demonstrating flow-controlled release from the SCS's outlet structure. As the water level rises in the chambers and surrounding gravel layer, it flows through the underdrain (A) and outlet manifolds (B) to the outlet structure. The weir wall (C), slows the release of water to the outlet pipe (D) (Image courtesy Advanced Drainage System®).

<u>2.3.4: Data Collection</u>

2.3.4.1: Monitoring Station Setup

Stations were oriented to monitor runoff entering both inlets and leaving the outlet structure. Each monitoring station was outfitted with v-notch weirs, Teledyne ISCO 730 bubbler modules, and Teledyne ISCO 6712 portable samplers (Teledyne-Isco^{TM}, Lincoln, NE) to facilitate hydrologic data collection and water quality sampling. Station ‘East’ was also equipped with both a Teledyne ISCO 674 automated tipping bucket rain gauge (Teledyne-Isco^{TM}, Lincoln, NE) and a manual rain gauge. All monitoring equipment, aside from the rain gauges, were stored in metal enclosures for protection and were only accessible to field personnel (Figure 2-6).

Figure 2-6. Monitoring Stations at Capital Oaks: (A) Station “Park” monitors roof and parking lot runoff from the western portion of the watershed (B) Station “East” monitors roof, parking lot, and unpaved/grassed runoff from the easter portion of the watershed parking lot (C) Station “Out” monitors effluent leaving the outlet structure.

2.3.4.2: Precipitation Data

System monitoring was conducted from November 2018 to January 2020. Storm events occurring during this time were measured by an ISCO 674 automated tipping bucket rain gauge, which quantified precipitation by counting alternating tips of buckets that each correspond to 0.25 mm (0.01 in) of rainfall depth. Discrete storm events were identified by a gap of at least six hours in precipitation. After each event, rainfall data were retrieved from the ISCO using a Rapid Transfer Device (RTD) and later uploaded to Flowlink® (Version 5.10.206). The depths recorded by the manual rain gauge were used to augment the accuracy of the tipping bucket gauge measurements. After each storm event, both rain gauges were inspected for blockages and the manual rain gauge was emptied.

Owing to unforeseen circumstances, some precipitation data - namely all of the rainfall durations, rainfall intensities, and antecedent dry periods - were supplemented by the State Climate Office of North Carolina (NCSCO, 2016) database. Data were retrieved from the REED-Reedy Creek Field Laboratory weather station located 8.1 km (5.05 mi) from the research site.

2.3.4.3: Hydrologic Data

The ISCO bubbler module was used to measure and record stage data (i.e., the hydraulic head flowing over the crest of the v-notch weirs) in 2-minute intervals. After each event, hydrology data were retrieved from the ISCO using a Rapid Transfer Device (RTD) and later uploaded to Flowlink®. Flowlink® was then used to convert the stage data into flow rates using known stage-discharge relationships (Table 2-2). Peak flow rates and runoff volumes were determined by utilizing runoff hydrographs generated in Flowlink® for each storm. Bypass effluent was unable to be separated from the total effluent with the employed monitoring setup.

Table 2-2. Flow measuring devices and conversion equations for each monitoring station.

Monitoring Station	Flow Measuring Devices	Conversion Equations
Inlet Park	60^{o} v-notch weir ISCO 730 bubbler module	$Q = 796.7\ H^{2.5}$
Inlet East	90^{o} v-notch weir ISCO 730 bubbler module	$Q = 1380\ H^{2.5}$
Outlet	90^{o} v-notch weir ISCO 730 bubbler module	$Q = 1380\ H^{2.5}$

2.3.4.4: Water Quality Data

Samples were drawn by the ISCO 6712 samplers on a volumetric flow-weighted basis and collected in 10-L HDPE composite bottles. Samples were retrieved for water quality analysis within 24 hours of rainfall cessation for rainfall depths between 2.5 mm to 51 mm (0.1 in to 2 in) when at least 70% of the storm's hydrograph was captured at all three monitoring stations. Composite bottles were agitated to resuspend settled sediment then poured into pre-specified sample bottles for analysis. Collected samples were placed on ice in a cooler and transported to the Center for Applied Aquatic Ecology (CAAE) laboratory at NC State University for analysis, located approximately 11 km (7 mi) from the research site.

At the laboratory, event mean concentrations (EMCs) were determined for the following pollutants: total suspended solids (TSS), total ammoniacal-nitrogen (TAN), total Kjeldahl nitrogen (TKN), nitrate and nitrite-nitrogen ($NO_{2,3}$), total phosphorus (TP), and orthophosphate (OP). The EMCs for organic nitrogen (ON), total nitrogen (TN), and particulate bound phosphorus (PBP) were calculated (Table 2-3).

Table 2-3. Water Quality analytes and analytical methods.

Analyte	Analytical Method	Practical Quantitation Limit
Total Suspended Solids (TSS)	Std Method 2540 D	-
Total Kjeldahl Nitrogen (TKN)	EPA Method 351.2	280 ug/L
Total Ammonia Nitrogen (TAN)	Std Method 4500 NH_3 G EPA Method 350.1	17.5 ug/L
Nitrate+Nitrite nitrogen ($NO_{2,3}$-N)	Std Method 4500 NO_3 F EPA Method 353.2	11.2 ug/L
Organic Nitrogen (ON)	TKN-TAN	-
Total Nitrogen (TN)	TKN + $NO_{2,3}$	-
Orthophosphate (OP)	Std Method 4500 P F EPA Method 365.1	12 ug/L
Particulate Bound Phosphate (PBP)	TP - OP	-
Total Phosphate (TP)	Std Method 4500 P F EPA Method 365.1	10 ug/L

2.3.5: Data Analysis

Flow rate and volume data for each monitoring station obtained during the monitoring period were compiled into an excel spreadsheet for analysis. The following metrics were used to characterize hydrologic performance: volume reduction (Eq. 1), peak flow reduction (Eq. 2), volume discharge ratio (Eq. 3), and peak flow discharge ratio (Eq. 4). Equipment failure led to the omission of several storms from the analysis.

(1) Volume Reduction (%) = $[V_{IN} - V_{OUT}] / V_{IN} \times 100$

(2) Peak Flow Reduction (%) = $[Q_{P,\,IN} - Q_{P,\,OUT}] / Q_{P,\,IN} \times 100$

(3) Volume Discharge Ratio (VDR) = V_{OUT} / V_{IN}

(4) Peak Discharge ratio (PDR) = $Q_{P,\,OUT} / Q_{P,\,IN}$

where V_{IN} is the influent volume in m^3, V_{OUT} is the effluent volume in m^3, $Q_{P, IN}$ is the greater of two peak flows entering the inlet pipes in L/s, $Q_{P, OUT}$ is the effluent peak flow rate in L/s.

Total influent EMC concentrations were a normalized (by volume) combination of inlet East and inlet Park. For instance, if inlet 'East' received 52% of the runoff volume entering the system during a storm event, total influent concentration would be calculated as: $EMC_{IN} = 0.52 * EMC_{EAST} + 0.48 * EMC_{PARK}$ for that pollutant.

Pollutant loads were calculated using Equation 5. The SCS's water quality treatment was assessed using the following metrics: Removal efficiency (Eq. 6)., pollutant load reduction (Eq. 7), and (for TN and TP) cumulative discharge loads (Eq. 8). Concentrations of the latter two pollutants were compared to water-quality thresholds established for macroinvertebrate health in North Carolina by McNett et al. (2010).

(5) Pollutant Load (mg) = EMC_i x V_i

(6) Removal Efficiency (%) = $[EMC_{IN} - EMC_{OUT}]/ EMC_{IN}$ x 100

(7) Pollutant Load Reduction (%) = $[L_{OUT}/L_{IN}]$ x 100

(8) Cumulative load normalized by annual rainfall (kg/ha/yr)

$= (\Sigma (c_i \times V_i) \times 1 \times 10^{-6})/A_{WS}/ P_{OBSERVED}/P_{ANNUAL}$

where EMC_{IN} is the influent pollutant concentrations in mg/L, EMC_{OUT} is the effluent pollutant concentrations in mg/L, EMC_i is the pollutant concentrations in mg/L, V_i is volume for storm i in L, c_i is the measured or event median concentration (mg/L) L_{IN} is the influent pollutant load in mg, L_{OUT} is the effluent pollutant load in mg, A_{WS} is the area of the watershed in ha, $P_{OBSERVED}$ is the measured rainfall during the monitoring period (mm), P_{ANNUAL} is the average annual precipitation (mm/yr) in Raleigh, NC (NOAA, 2020).

2.3.5.1: Statistical Analysis

All statistical analysis was conducted using R Statistical Software (R Core Team 2019). For analytes that were below the Minimum Detection Limit (MDL), half of the MDL was used for statistical analyses in accordance with recommendations from Helsel (1990). Data for each parameter were analyzed for normality and log-normality using the Shapiro-Wilk normality test and checked visually using Q-Q plots. Statistical significance was estimated for differences between influent and effluent characteristics using t-test if the data were normally distributed or using non-parametric Wilcoxon Signed Rank Test otherwise. Statistical analysis was conducted using an α of 0.05.

2.4: Results & Discussion

<u>2.4.1: Precipitation</u>

A total of 73 storm events ranging 4.32 mm - 92.71 mm (0.17 in - 3.65 in) were measured over the 14-month study period. The mean and median rainfall depths recorded were 18.54 mm (0.73 in) and 11.94 mm (0.47 in), respectively. The total rainfall depth throughout the monitoring period was 1343.66 mm (52.90 in) which was 122.9% of the average annual rainfall in Raleigh 1094.74 mm (43.10 in) (NOAA, 2020). Observed storm events varied with duration (0.3 - 46.7 hrs), average intensity [0.25 - 44.45 mm/hr (0.01 - 1.75 in/hr)], and peak 5-minute intensity [9.14 - 472.44 mm/hr (0.36 - 18.60 in/hr)]. The longest antecedent dry period (ADP) observed between storm events was 12.6 days. Seasonal distribution according to solstice and equinox (Spring: March 20-June 20, Summer: June 21-September 22, Fall: September 23-December 20, Winter: December 21- March 19) is presented in Table 2-4.

Table 2-4. Seasonal distribution of precipitation events occurring during the monitoring period.

Season	Events	Rainfall Depth (mm)		Duration (hrs)		Peak 5-min Intensity (mm/hr)		ADP (days)	
		Mean	Median	Mean	Median	Mean	Median	Mean	Median
Spring	9	17.78	11.68	10.61	8.30	67.05	45.72	4.49	4.20
Summer	22	16.51	10.41	5.31	3.43	121.66	35.81	3.70	1.86
Fall	19	23.11	18.79	10.76	10.63	63.75	47.24	4.71	3.40
Winter	23	16.76	14.22	11.57	9.04	45.97	35.05	4.27	3.25
Overall	73	18.54	11.93	9.96	8.11	72.39	39.62	4.29	3.15

2.4.2: Hydrologic Performance

2.4.2.1: Volume Reduction

Owing to monitoring equipment failure, paired hydrologic measurements for only 62 of the 73 storm events were successfully measured. Statistical analysis revealed that the subsurface chamber system was able to significantly reduce incoming runoff volumes throughout the monitoring period (p-value < 0.001, using Wilcox signed rank test). Mean and median volume reductions for all measured storm events were 64% and 74%, respectively. The system completely abated runoff volume for four storm events and achieved better than 50% volume reduction for 45 of the 62 (72%) storms successfully measured.

Discretizing storm events shows that volume mitigation provided by the SCS is inversely related to storm size (Table 2-5). Median volume reduction for storms up to 25.4 mm (1.0 in), between 25.4 and 51.0 mm (1.0-2.0 in), and greater than 51.0 mm (2.0 in) was 78%, 41%, and 41%, respectively. These findings suggest that: (1) the fraction of volume reduction is high for storms less than 25.4 mm (1.0 in) and (2) the system's ability to proportionally reduce runoff volume plateaus beyond the 25.4 mm (1.0 in) storm. While the sample size for storms over 25.4 mm (1.0 in) was relatively small (11 storm events), the SCS consistently reduced volumes of larger storm events including that of a 92.71 mm (3.65 in) event.

Table 2-5. Summary of changes in volume for discretized storm events.

Rainfall Depth, (mm)	**Events**	**Volume In (m^3)**		**Volume Out (m^3)**		**% Reduction**	
		Mean	**Median**	**Mean**	**Median**	**Mean**	**Median**
<25	51	74	52	31	11	71%	79%
25-51	7	399	390	258	234	40%	42%
>51	4	721	780	504	408	31%	42%
Overall	**62**	**153**	**92**	**87**	**19**	**65%**	**75%**

The mean and median volume discharge ratios (VDRs, Eq. 3) were of 0.36 and 0.26, respectively. These ratios are comparable to the 0.33 threshold suggested by Davis (2008) for LID practices. The plot of VDR vs Rainfall depth shows a direct relationship for rainfall depth with VDR, which is to be expected (Figure 2-7). VDR was greater than 1.0 for three of the observed storm events- likely due to equipment error.

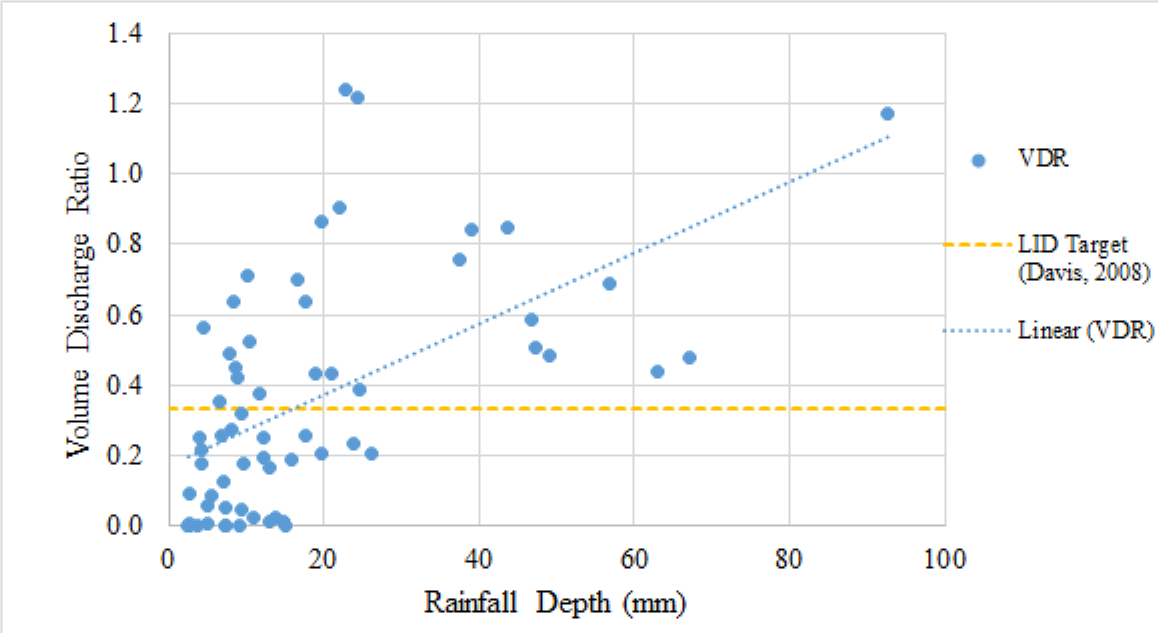

Figure 2-7. Volume discharge ratio versus Rainfall depth.

Volume reduction was primarily attributed to infiltration into the underlying sandy loam soil (HSG-B). Because the SCS was installed about 4 years before the monitoring program commenced, the transmissivity of the underlying soils could not be investigated. Nevertheless, web soil survey describes HSG-B soils as having moderate infiltration and transmissivity (Web Soil Survey, 2020). Assuming 100% of the volume reduction by the system was due to infiltration, more than 4040 m^3 (142,700 ft^3) of runoff was infiltrated throughout the monitoring period. Mean and median volume infiltrated per storm event were 65 m^3 (2,300 ft^3) and 36 m^3 (1,281 ft^3), respectively. It is worth noting that volume reduction provided by the SCS also contributes to pollutant load mitigation by lowering the quantity of polluted runoff flowing directly into ambient waters.

2.4.2.2: Peak Flow Mitigation

Peak flows entering the SCS ranged from 0.1 L/s to 311 L/s (0.004 – 10.9 ft^3/s). Peak inflows were significantly reduced (p-value < 0.0001, using Wilcoxon signed rank test) by mean and median values of 88.9% and 97.4%, respectively (Table 2-6). Mean and median flows leaving the system were 8.4 L/s (0.3 ft^3/s) and 0.8 L/s (0.03 ft^3/s), respectively. The SCS had greater than 99.9% peak flow (Q_p) reduction for 10 of 62 (16%) storm events and better than 90% reduction occurred for 46 of 62 events (74%). At least 50% peak flow mitigation was realized for 59 of 62 (95%) events.

Table 2-6. Summary of discretized peak flow reduction.

Rainfall Depth, (mm)	Events	Avg. Peak 5-min Intensity (mm/hr)	Peak Flow In (L/s)		Peak Flow Out (L/s)		Reduction	
			Mean	Median	Mean	Median	Mean	Median
<25.4	51	50	21	14	1	0.3	95%	98%
25-51	7	112	140	99	18	11	82%	83%
>51	4	275	111	83	85	65	27%	30%
Overall	**62**	**7**	**40.4**	**19.0**	**8.5**	**0.5**	**89%**	**97%**

A plot of PDR vs peak5-min rainfall intensity illustrates a positive correlation between storm intensity and peak outflow from the SCS (Figure 2-8). The mean and median PDRs (Eq. 4) were 0.11 and 0.03, respectively, which were substantially lower than the target of 0.33 suggested by Davis (2008) (Figure 2-8). The SCS provided excellent mitigation of peak flows associated with the 1-year 10-min storm intensity (97 mm/hr; NOAA, 2017). Mean and median peak flow reductions of storm events less than the 1-year 10-min intensity (n=43) were 93% and

98%, respectively. For these storm events, mean and median PDR were 0.07 and 0.02, respectively. However, the SCS was less efficient at mitigating larger storm intensities (n=12). Mean and Median peak flow reductions decreased to 68% and 85%, respectively, while mean and median PDR were 0.32 and 0.15, respectively. The PDR was greater than 1 for one storm event during the monitoring period, which was likely due to equipment failure.

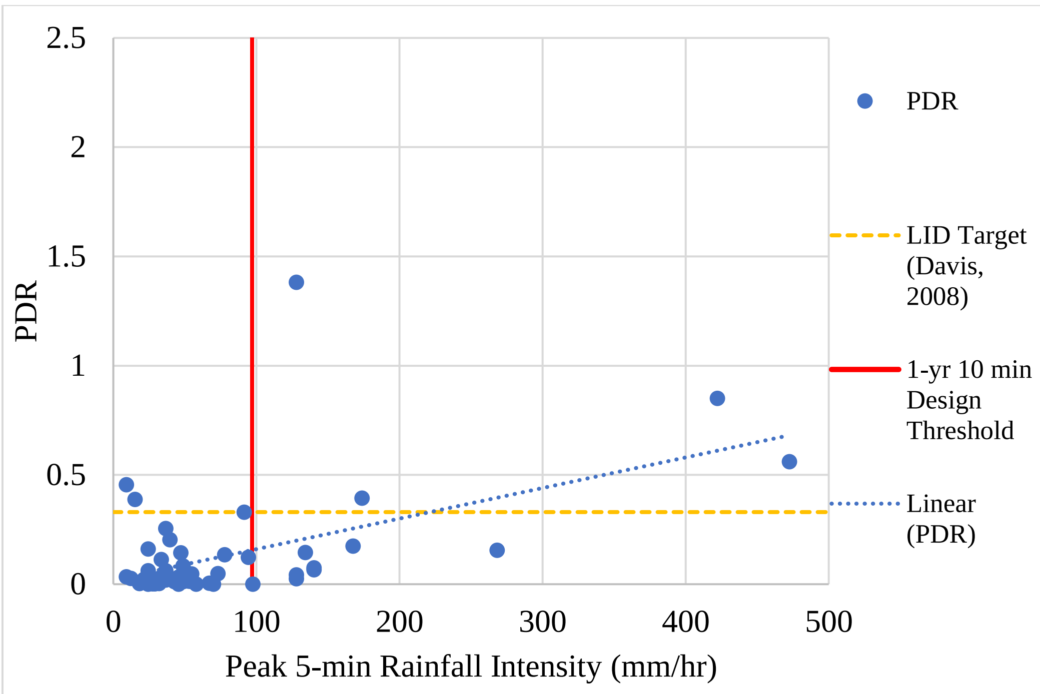

Figure 2-8. Peak discharge ratio vs peak 5-min rainfall intensity with 1-yr 10-min rainfall intensity and LID target suggested by Davis, (2008).

Peak flow reduction is attributed to the SCS's storage capacity and the design of the outlet structure, which facilitated slow and controlled release of stormwater from the system. Noteworthy factors affecting available system storage for a given event were: (1) antecedent conditions and (2) the rate of stormwater infiltration into underlying soil.

<u>2.4.3: Water Quality Performance</u>

Sixteen storm events were sampled and analyzed for analytes in Table 2-3. A summary of the sampled storm events and the results of EMC and load analysis are presented in Tables 2-7 to 2-9.

Table 2-7. Summary of precipitation data for monitored water quality events.

		Rainfall Depth (mm)		Duration (hrs)		Peak 5-min Int. (mm/hr)		ADP (days)	
Season	**Events**	Mean	Median	Mean	Median	Mean	Median	Mean	Median
Spring	6	9.91	10.16	9.5	9.0	37.1	37.3	5.31	4.25
Summer	1	8.38	8.38	0.3	0.3	18.3	18.3	1.30	1.30
Fall	3	14.73	10.41	5.5	5.4	75.2	50.3	2.13	2.10
Winter	6	30.99	20.32	15.9	9.1	57.2	45.0	2.25	2.00
Overall	**16**	**18.8**	**11.94**	**10.5**	**7.6**	**48.26**	**40.64**	**3.31**	**2.45**

Table 2-8. Summary of changes in pollutant EMC for sampled analytes.

Analyte	Influent EMC (mg/L)		Effluent EMC (mg/L)		Reduction	
	Mean	**Median**	**Mean**	**Median**	**Mean**	**Median**
TSS	77.9	33.80	4.38	3.22	86 %	91 %
ON	0.94	0.82	0.25	0.27	64 %	68 %
TKN	1.21	1.08	0.29	0.29	68 %	70 %
TAN	0.26	0.11	0.04	0.02	74 %	78 %
$NO_{2,3}$	0.20	0.16	0.47	0.42	-163 %	-155 %
TN	1.41	1.36	0.76	0.66	34 %	35 %
OP	0.02	0.02	0.01	0.01	49 %	60 %
PBP	0.11	0.10	0.03	0.03	64 %	71 %
TP	0.14	0.12	0.04	0.03	64 %	68 %

Table 2-9. Summary of changes in pollutant load for sampled analytes.

Analyte	Influent Load (g)		Effluent Load (g)		Reduction	
	Mean	Median	Mean	Median	Mean	Median
TSS	4500	2436	295	54	95 %	97 %
ON	109	59	23	5	79%	92%
TKN	127	82	27	5	78%	94%
TAN	18	9	2	0.5	90 %	95%
$NO_{2,3}$	23	14	36	9	-60 %	33%
TN	168	100	59	14	77 %	84 %
OP	5	1	1	0.2	80 %	86 %
PBP	14	9	2	0.5	86 %	95 %
TP	15	9	3	0.5	88 %	91 %

Table 2-10. Summary of changes in cumulative pollutant load for sampled analytes.

Analyte	Cumulative Load (kg/ha/yr)		Reduction (%)
	Influent	Effluent	
TSS	159.5	10.4	93%
ON	3.9	0.8	79%
TKN	4.5	0.9	80%
TAN	0.6	0.1	88%
$NO_{2,3}$	0.9	1.3	-40%
TN	6.0	2.1	65%
OP	0.1	0.03	70%
PBP	0.4	0.1	85%
TP	0.5	0.1	81%

2.4.3.1: Total Suspended Solids

The SCS and isolator row received a wide range of influent TSS concentrations (6 mg/L - 539 mg/L) and consistently reduced (statistically significant; p-value < 0.001, using Wilcox signed rank test) TSS throughout the study (Figure 2-9). Influent concentrations of TSS (mean: 77.9 mg/L, median 33.8 mg/L) were released at much lower concentrations (effluent mean: 4.38 mg/L, median 3.22 mg/L), resulting in mean and median EMC reductions of 86% and 91%, respectively. Five of the 16 monitored storms produced effluent concentrations that were below the laboratory's reporting limit of 2.5 mg/L.

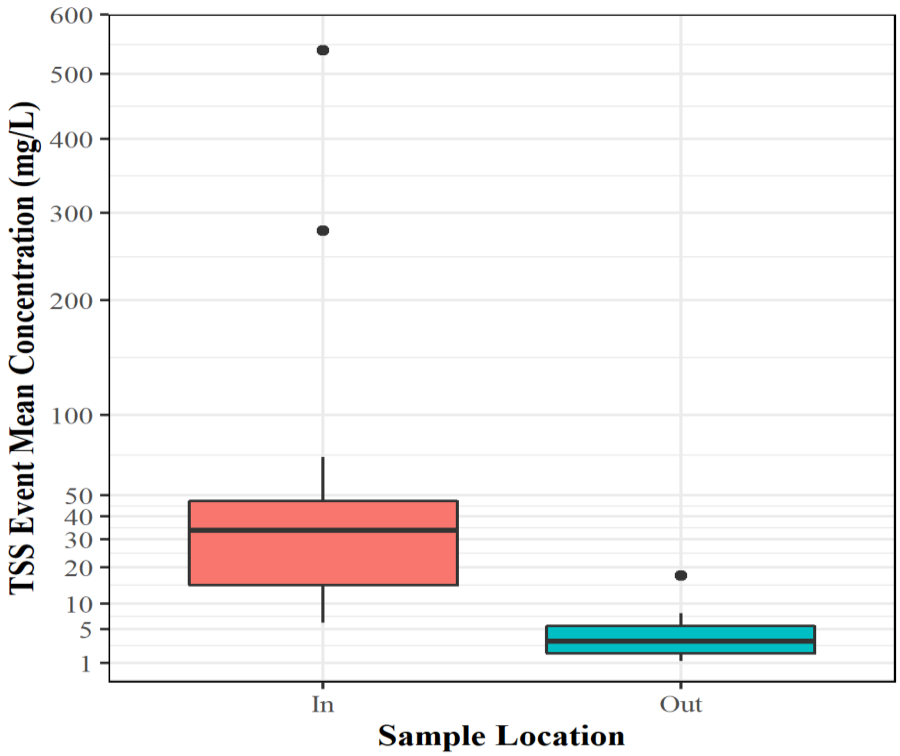

Figure 2-9. Box plot of changes in EMC for TSS**.**

When coupled with the volume mitigation previously described, the SCS significantly and substantially reduced TSS loads (p-value < 0.001, using Wilcox signed rank test). Influent loads (mean = 4500 g, median = 2436 g) were dramatically lowered (mean = 295 g, median =54 g) resulting in mean and median load removal efficiency of 95% and 97%, respectively.

NCDEQ's standards (NCDEQ, 2017b) mandate TSS removal efficiencies that must be achieved for a SCM to be considered as a primary practice in NC. Primary practices can stand alone and are the best designation possible for an SCM by NCDEQ. The SCS met NCDEQ requirements for all but six storm events, which did not meet valid testing requirements due to low influent concentrations (Table 2-11, Figure 2-10). Several events had influent concentrations that were less than what had been previously considered by Schueler (1996) as irreducible (<20 mg/L), including one event with an influent EMC of 5.99 mg/L. The SCS still provided a median removal efficiency of 87% for these six events.

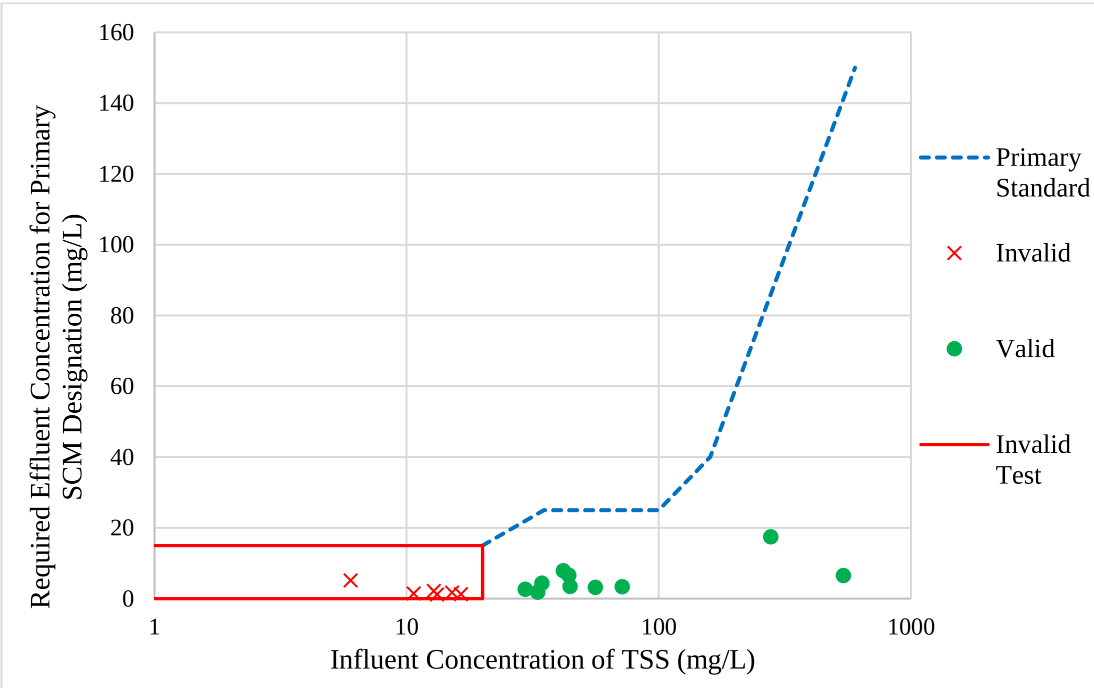

Figure 2-10. TSS treatment by the SCS plotted relative to current standards for primary SCM designation mandated by NCDEQ. To meet primary standards, inflow-outflow concentration pairs should be below the dashed line.

Table 2-11: Summary of performance according to NCDEQ requirements for primary SCMs.

Influent Concentration (mg/L)	NCDEQ Performance Standard	# of Qualifying Storm Events	SCS Performance
0-20	Invalid	6	Median: 87% removal
20-35	>29% Reduction	3	Mean: 93% reduction Median: 93% reduction
35-100	<25 mg/L effluent concentration	5	Mean: 4.85 mg/L Median: 3.39 mg/L
>100	>75% Reduction	2	Storm 1: 99% reduction Storm 2: 91% reduction

NCDEQ also specifies a maximum median effluent concentration for TSS of 35 mg/L (NCDEQ, 2017). Exceedance probability plots show that while influent concentrations exceeded the 35 mg/L target 45% of the time, effluent concentrations were below the threshold for all observed storm events (Figure 2-11). The maximum effluent concentration measured during the study period was 17.46 mg/L, which was lower than the widely used Technology Assessment Protocol- Ecology (TAPE) standards of 20 mg/L (WSDE, 2018).

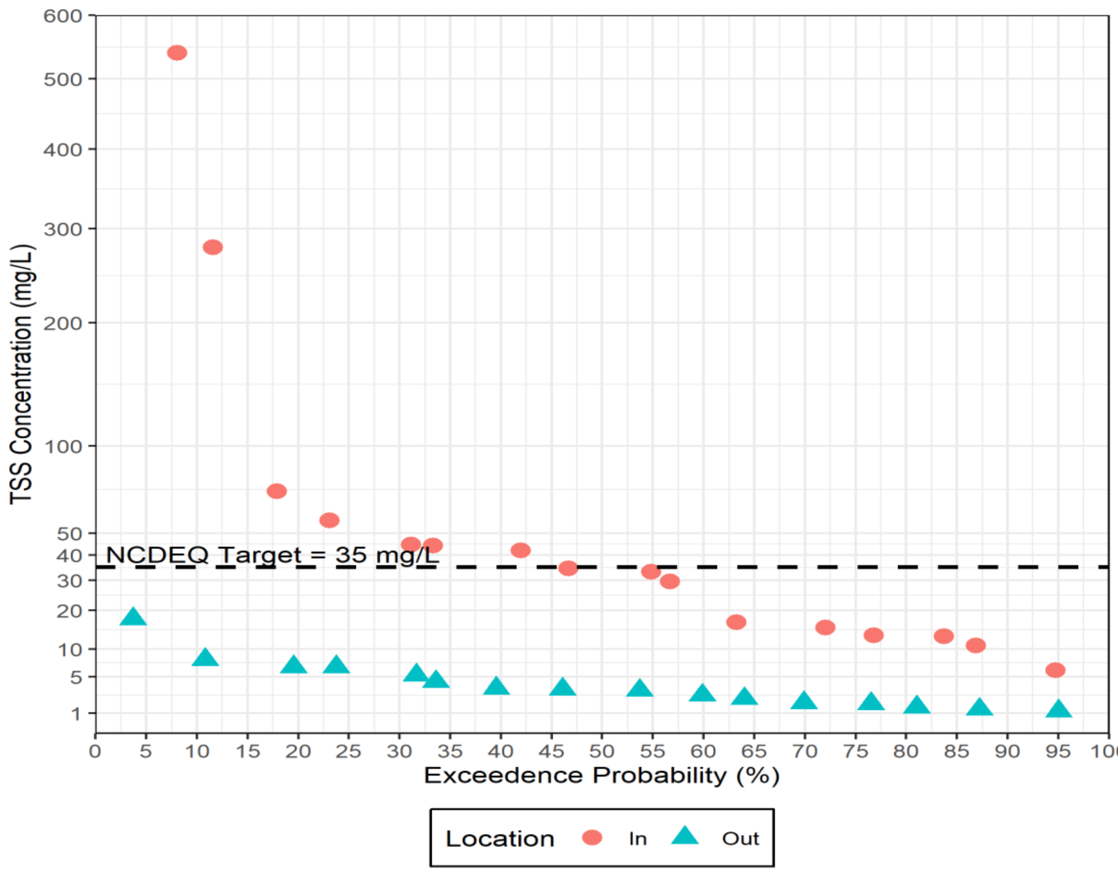

Figure 2-11. Exceedance probability for observed TSS concentrations compared to NCDEQ's (2017) maximum median effluent TSS concentration.

TSS is primarily removed from runoff via filtration in the isolator row. By capturing and straining the first flush volume, the isolator row treats the portion of runoff that is most likely to have the highest concentrations of TSS. TSS is also mitigated via sedimentation in other chambers as runoff is detained during storm events. Finally, and as with all subsequent pollutant discussions, TSS load was reduced by volume mitigation.

2.4.3.2: Nitrogen

Influent and effluent concentrations of all nitrogen species were significantly different (p-values all less than 0.0001, Wilcox signed rank test). A summary of changes in EMCs is presented in Figure 2-12 and Table 2-11.

Table 2-12. Summary of changes in EMC for nitrogen species during the monitoring period.

Nitrogen Species	Influent EMC (mg/L)		Effluent EMC (mg/L)		Reduction (%)	
	Mean	**Median**	**Mean**	**Median**	**Mean**	**Median**
ON	0.94	0.82	0.25	0.27	64	68.1
TKN	1.21	1.08	0.29	0.29	68	69.7
TAN	0.26	0.11	0.04	0.02	74	78
$NO_{2,3}$	0.20	0.16	0.47	0.42	-163	-155.2
TN	**1.41**	**1.36**	**0.76**	**0.66**	**34**	**34.9**

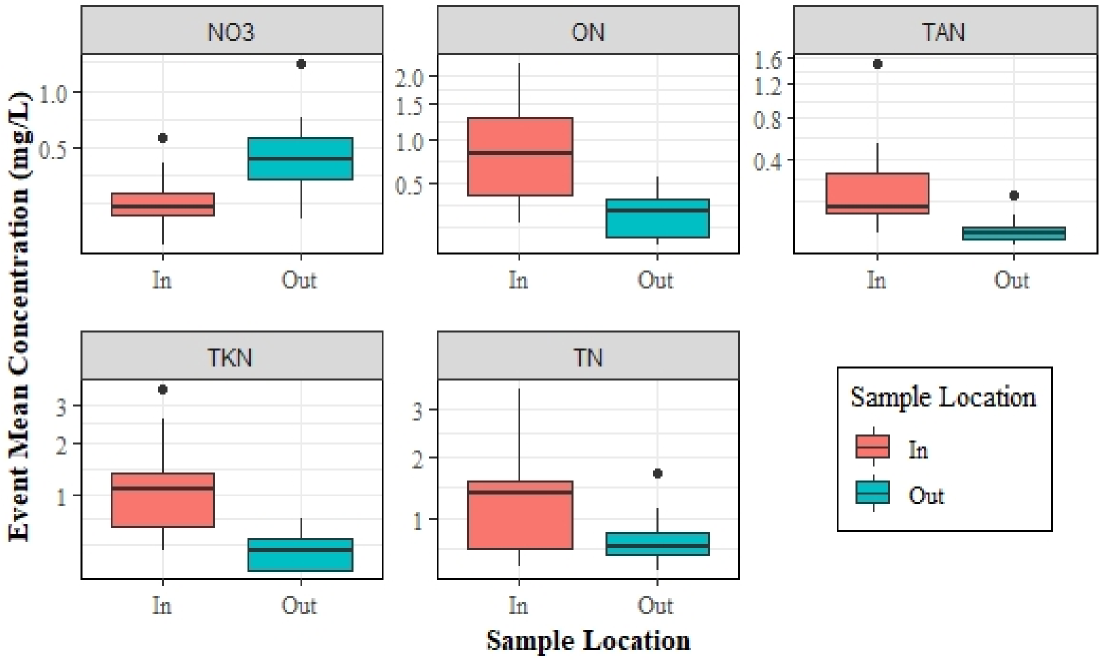

Figure 2-12. Box plot of changes in EMC for nitrogen species.

Of the multiple nitrogen species that comprise TN, organic nitrogen was predominant, with an average ON:TN of 0.66, similar to the 0.60 reported by Lusk et al. (2020) for urban residential runoff. Although the proportion of particulate organic nitrogen (PON) was not determined in this study, TN removal was mainly attributed to PON filtration in the isolator row and subsequent sedimentation in the remaining chambers (Igielski et al., 2019). Moreover, sources of $NO_{2,3}$ in this watershed - likely atmospheric deposition, vehicular exhaust, and fertilizer (Jani et al., 2020)- were supplemented by nitrification, which converts TAN to $NO_{2,3}$ in aerobic conditions via *Nitrosomonas* and *Nitrobacte*r species (Davis et al., (2010). Nitrification is the likely cause of TAN reduction and $NO_{2,3}$ export.

McNett et al. (2010) recommended target effluent TN concentrations to match receiving water quality. Their statistically-based thresholds were based upon benthos populations. Streams with "excellent" benthos were assigned a target TN concentration of 0.69 mg/L, while those with "good" benthos health had a TN concentration of 0.99 mg/L. Fifty-five percent of storm events met the "excellent" target, with 80% achieving "good" status (Figure 2-13).

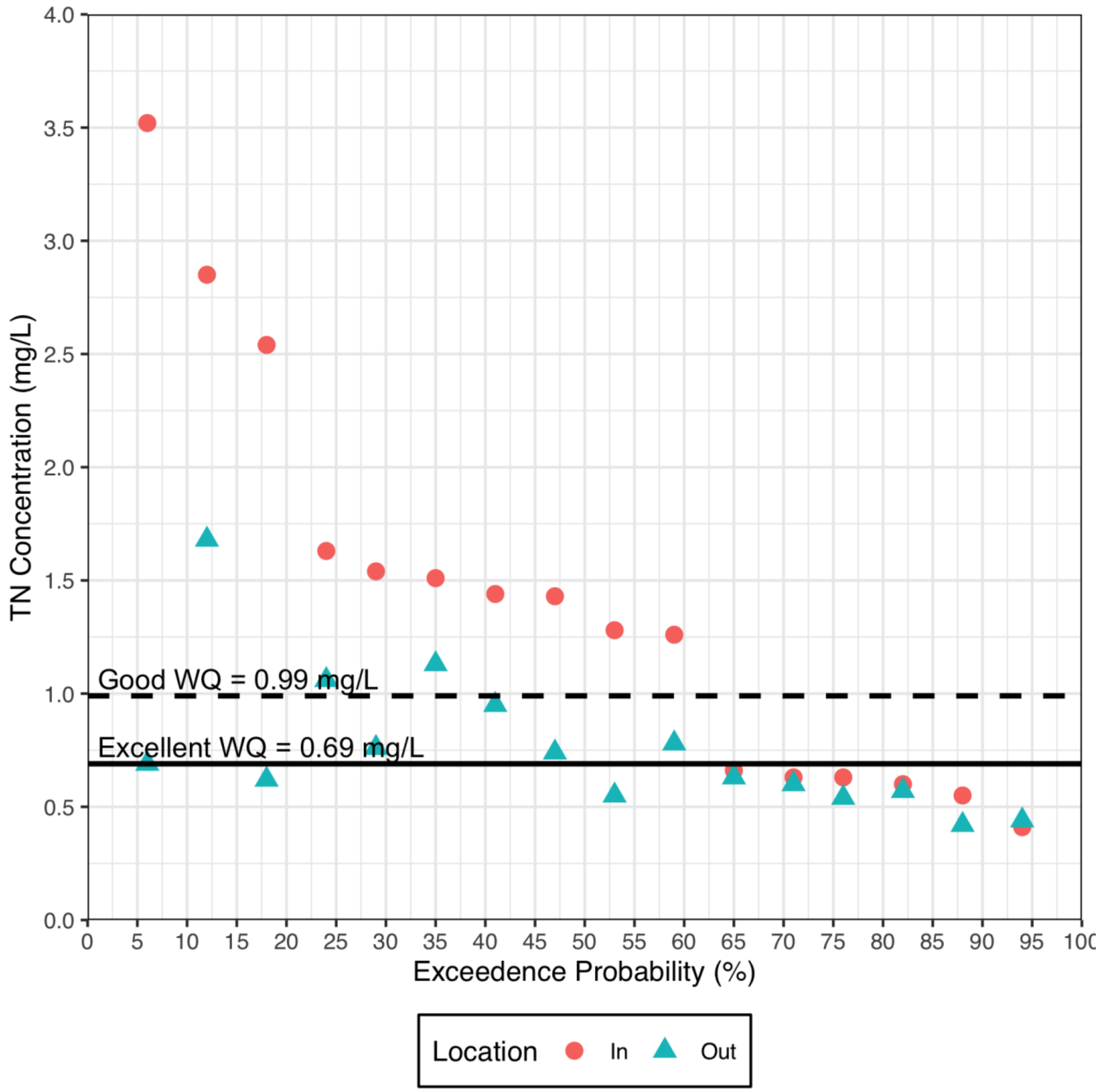

Figure 2-13. Exceedance probability plots for TN with target thresholds suggested by McNett et al. (2010).

The SCS significantly reduced TN loads (p-value < 0.0001, using Wilcox signed rank test). Mean and median removal efficiencies were 77% and 84%, respectively. Nutrient sensitive watersheds in NC have been assigned allowable nutrient loading rates to be deemed compliant with state regulation (NCDEQ, 2020). Cumulative TN loads were reduced from 6.0 kg/ha/yr to 2.1 kg/ha/yr (65% reduction), which meets nutrient loading goals for Tar-Pamlico River (4.5 kg/ha/yr; 4.0 lb/ac/yr) Jordan Lake (4.5 kg/ha/yr; 4.0 lb/ac/yr), Neuse River(4.04 kg/ha/yr; 3.6 lb/ac/yr), and Falls Lake (2.47 kg/ha/yr; 2.2 lb/ac/yr) watersheds (NCDEQ, 1999; 2003; 2009; 2010).

2.4.3.3 Phosphorus

The range of TP and PBP EMCs entering the SCS were 0.04 - 0.35 mg/L and 0.02 - 0.28 mg/L, respectively. The range of OP could not be estimated because 9 of the monitored 16 storms had both influent and effluent concentrations below the detection limit of 0.012 mg/L. For this reason, OP was also excluded from statistical analysis. Changes in EMCs of TP and PBP were, however, significant (respective p-values: 0.0004, 0.0007, using Wilcox signed rank test). Median EMC removal efficiencies were 68% and 71%, for TP and PBP, respectively. Using estimates provided by CAAE, OP removal efficiency was 60%, but overall change in EMC was very slight (< 0.01 mg/L).

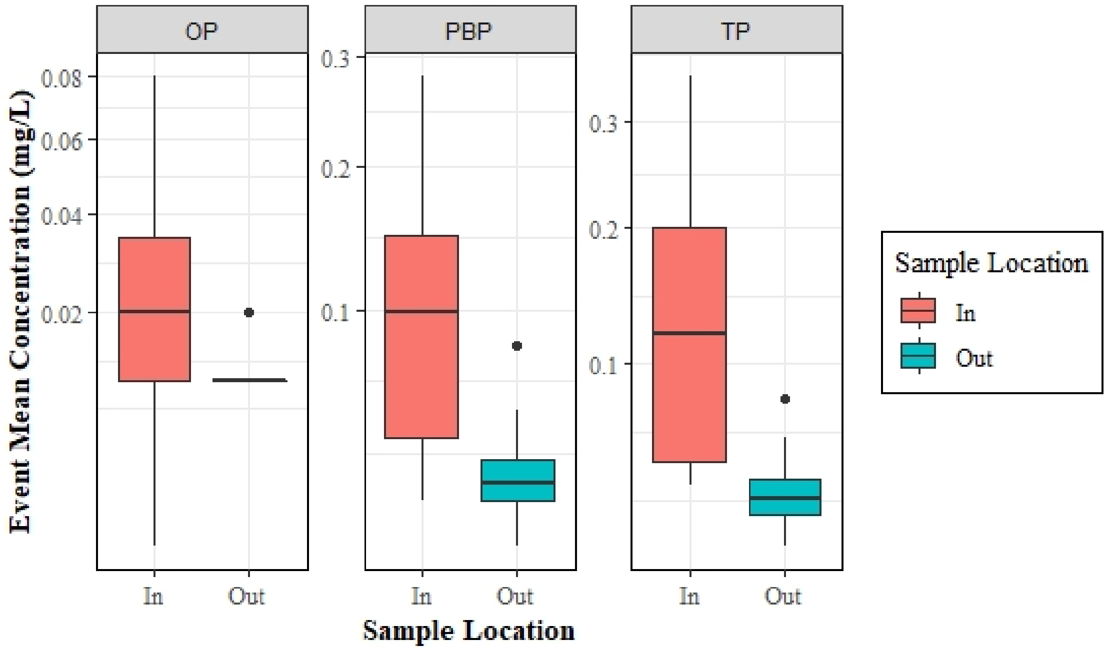

Figure 2-14. Boxplot of EMC changes for phosphorus species.

McNett et al. (2010) also recommended effluent TP concentrations for receiving waters with excellent and good benthic health. The "excellent" TP target threshold was 0.06 mg/L, while the "good" threshold was 0.11 mg/L. The SCS met the "excellent" threshold for 15 of the 16 sampled events, releasing mean and median effluent TP concentrations of 0.04 mg/L and 0.03 mg/L, respectively. While the effluent of 85% of storm events met "excellent" goals, all storms were lower than the "good" water quality threshold (Figure 2-15).

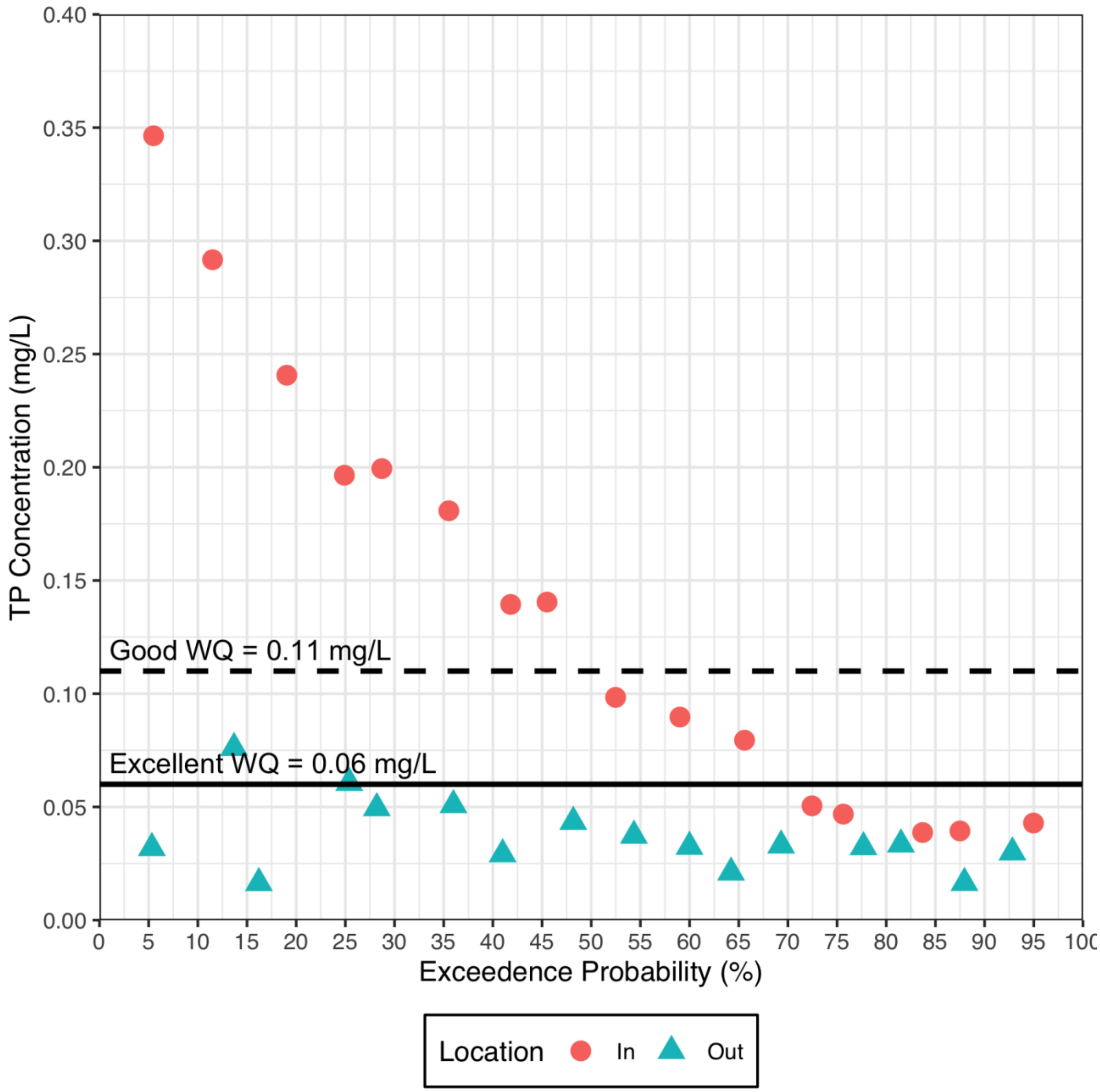

Figure 2-15. Exceedance probability plots for TP with target thresholds suggested by McNett et al. (2010).

TP loads were significantly reduced with median removal efficiency of 91% (mean = 88%) (p-value: <0.0001, using Wilcox signed rank test). The cumulative TP load leaving the SCS was reduced by 81% from 0.50 kg/ha/yr (0.45 lb/ac/yr) to 0.10 kg/ha/yr (0.09 lb/ac/yr). The effluent loading rate was less than one-half that of the nutrient loading goals for the Tar-Pamlico

River (0.45 kg/ha/yr; 0.4 lb/ac/yr) and Falls Lake (0.37 kg/ha/yr; 0.33 lb/ac/yr) watersheds (NCDEQ, 2003; 2010).

PBP comprised more than 79% of the TP concentrations. TP removal was attributed mainly to removal of particles via filtration in the isolator row and sedimentation throughout the system. Negligible changes in the dissolved forms of phosphorus are likely due to the already low concentrations entering the SCS.

2.4.6: Potential for Improvements: Internal Water Storage Zone

Researchers acknowledge that denitrification can play a key role in managing nitrogen export from SCMs (Linn et al., 2015; Morse et al., 2017; Igielski et al., 2019). Igielski et al. (2019) highlight that the denitrification process utilizes: (1) organic carbon to function as an electron donor, and (2) depleted oxygen to ensure favorable conditions to facilitate the sequential reduction of nitrate. Urban stormwater typically lacks the organic carbon source needed to facilitate the denitrification process (Sun et al., 2017), however, any lignocellulosic material (e.g., mulch and wood chips) can be used as a carbon and energy source as well as support the necessary microbial population (Igielski et al., 2019). To facilitate denitrification in SCMs, Davis et al. (2010) recommend the use of an internal water storage zone (IWSZ), which is an area of the SCM that is designed to promote anoxic conditions between storm events. In conjunction with mulch or other carbon source, an IWSZ can be implemented to facilitate the conditions necessary for the denitrification process. Moreover, inter-event storage provided by the IWSZ can also enhance volume reduction via exfiltration to underlying soils. One notable drawback of implementing the IWSZ, however, is diminished storage capacity. This can lower peak flow mitigation by inducing bypass flow for smaller, less intense storms events.

2.5: Summary & Conclusion

The StormTech subsurface chamber system was successfully monitored over a 14-month period in accordance with NCDEQ's (2017a) NEST program requirements. The results indicate substantial volume and peak flow reductions facilitated by detention, infiltration, and controlled flow release from the outlet structure. Pollutant removal is attributed to filtration in the isolator row and sedimentation throughout subsequent chambers. TSS reductions met NCDEQ (2017b) requirements for top tier (or primary) SCMs, and TP and TN reduction generally surpassed "good" water quality thresholds for nutrient sensitive watersheds in the Piedmont NC (McNett et al., 2010).

Hydrologic and water quality performance highlights are as follows:

- Peak flows and runoff volumes were significantly reduced with median reductions of 96% and 75%, respectively.
- Median TSS removal efficiency was 91%. TN and TP concentrations were both significantly reduced with median event reductions of 35% and 68%, respectively.
- Effluent nutrient loads were less than Tar-Pamlico River, Jordan Lake, Neuse River, and Falls Lake nutrient loading thresholds promulgated by NCDEQ.

These results suggest that the SCS effectively treated runoff from a moderately impervious watershed (55% impervious). The SCS is a viable MTD for watersheds requiring peak flow and TSS abatement. Future work should investigate SCS performance in highly impervious (>80%) watersheds.

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CHAPTER 3: Developing a Framework to Assess Pathogen Mitigation in Stormwater Control Measures.

3.1: Abstract

Stormwater runoff, contaminated with the fecal matter of warm-blooded animals, contributes to the degradation to surface waters by increasing the concentration of pathogenic microbes. One strategy for protecting surface water is to employ stormwater control measures (SCMs) to treat stormwater prior to its release into receiving waters. Although researchers note variability in their performance, SCMs can remove pathogens via filtration, sedimentation, phytoremediation, and adsorption. Proprietary SCMs, which are also referred to as Manufactured Treatment Devices (MTDs), are emerging as viable stormwater management options; however, researchers and water quality agencies lack the means by which to evaluate their pathogen-trapping capabilities in comparison to non-proprietary SCMs (i.e., bioretention cells, wetlands, wet ponds, etc.). The objective of this study was to develop a framework that can be used to compare and assess the performance of MTDs for the removal of three pathogen indicator species: fecal coliform, *Escherichia coli* (*E. Coli*), and e*nterococci*. A meta-analysis was conducted to extract performance data of mainly non-proprietary SCMs. Specifically, the changes in geometric mean concentrations, as well as their removal efficiencies, were compiled as reported in peer-reviewed publications. Using the collected data, SCM types were placed into three tiers based on how well they treated Fecal Indicator Bacteria (FIB). SCMs in Tier 1 were the most effective at mitigating FIB concentrations. These SCMs either (1) reliably exposed FIB to filtration (bioretention cells and infiltration basins), or in 2 of 3 cases, (2) coupled phytoremediation with sedimentation [constructed stormwater wetlands (CSWs)]. Tier 2 SCMs were less effective than SCMs in Tier 1, but still provided some FIB removal because they had reliable sedimentation (wet ponds). The CSW was included in Tier 2 for *E. Coli*. Tier 3 consists

of the SCMs that generally did not provide effective removal of FIBs (dry ponds and MTDs). The studied bioswale remains unassigned due to limited data. A framework was also constructed that relates geometric mean influent concentrations of FIB to geometric mean effluent concentrations of FIB. The framework can be used to determine whether future SCMs, including MTDs, capture and treat FIB comparably to Tier 1 SCMs.

3.2: Introduction

Pathogenic microorganisms are a major contributor to the biological degradation of ambient waters. In the United States, the current National Summary of State Information, which summarizes the conditions of surface waters as reported by each state to the United States Environmental Protection Agency (USEPA), lists pathogens as the primary cause of impairment in ‘rivers and streams’ and a top five contributor to impairment of ‘coastal shorelines, oceans and near coastal’, and ‘wetlands’ (USEPA, 2020). Contamination of water bodies designated for recreational use are particularly concerning because of increased human health risk associated with exposure. Several studies have been published noting links between gastrointestinal infection, respiratory ailments, skin irritations, among other illnesses and recreational water use (Papastergiou et al., 2012; Arnold et al., 2013; Fewtrell & Kay, 2015). In addition to public health concerns, tourism and fishing industries are also impacted as pathogen-polluted water bodies are often closed to recreational and commercial use (Dorfman et al., 2007).

Stormwater runoff is a significant contributor of pathogens to inland and coastal waters (Gaffield et al., 2003). Nearly 40% of impaired waters in the United States are attributed to stormwater runoff (Arnone & Walling 2007). Pathogens are mainly introduced to stormwater via contact with pathogen-laden fecal remains of humans and other warm-blooded domesticated or wild animals. Several studies link the development of a watershed, which is indicative of increases in human and domestic animal population density, to higher pathogenic microbe concentrations (Mallin et al., 2000; Line et al., 2008; Schoonover & Lockaby, 2006). Typical ranges of pathogen indicator species, also referred to as fecal indicator bacteria (FIB), in urban runoff are between 10^2–10^7 CFU/100 mL for *E. Coli* and 10^3–10^7CFU/100 mL for fecal coliform (Tota-Maharaj and Scholz, 2010).

The process of assessing biological water quality differs from monitoring physical and chemical pollutants. Samples are typically required to be delivered to laboratories for analysis within the maximum hold time of six hours. Consequently, many researchers and regulatory agents utilize grab sample collection to ensure samples arrive at laboratories in a timely manner. Because grab samples only represent the condition of the water at a particular point in time, statistical indicators of central tendency, such as the geometric mean, are commonly used to summarize microbial density in water (Wymer, 2007).

Under the Clean Water Act of 1972, federal and state water quality regulations have been established to address known point and non-point sources of pathogenic microbes to a water body. Storm sewers are subjected to total maximum daily loads (TMDLs), which are loading limits developed based on designated use of an impaired water body with the goal of regaining water quality integrity (USEPA, 2018). Recreational water quality criteria have also been established for fresh and marine water bodies. The USEPA recommends that states implement geometric mean limits of 35 CFU/100 mL for *Enterococci* for fresh and marine waters and 126 CFU/100 mL for *E. coli* for freshwater systems over a 30-day sampling period (USEPA, 2012).

Stormwater control measures (SCMs) are typically employed to limit the impact of urban runoff on receiving water bodies and to help achieve water quality standards. These systems employ physical, chemical, or biological mechanisms that can remove or inactivate pathogens in stormwater such as filtration, sedimentation, adsorption, desiccation, phytoremediation, and photoinactivation (Table 3-1). Environmental conditions such as temperature, pH, salinity, presence of nutrients, predation, and competition also play a major role in enhancing or inhibiting the removal of pathogen indicators (Stevik et al., 2004). Although the effectiveness of each mechanism has not yet been directly investigated, filtration and sedimentation are considered primary mechanisms owing to the realization that considerable portions of indicator organisms are bound to sediment and particulate matter (Krometis et al., 2009). However, SCMs can also be sources of pathogens by attracting wildlife and leaching or resuspending captured pathogens (Hathaway & Hunt, 2008).

Manufactured treatment devices (MTDs) are gaining popularity as viable stormwater treatment alternatives primarily for implementation in urban watersheds. These systems are uniquely designed to remove pollutants from stormwater using many of the same mechanisms utilized by non-proprietary SCMs. Therefore, MTDs likely share the ability to mitigate pathogens from urban runoff; however, their capacity to do so has been seldom explored in research. Furthermore, gauging their performance, especially in comparison to non-proprietary SCMs, is also difficult due to the absence of a performance framework. The lack of a metric to assess the FIB removal performance of proprietary systems forms the premise of this paper. The objective of this study is to develop a frame of reference that can be used to gauge the potential for FIB removal by MTDs.

Table 3-1. Common SCMs and the mechanisms they employ for pathogen removal.

Stormwater Control Measure	Description	Pathogen Removal Mechanisms
Bioretention cells	Vegetated infiltration systems designed to temporarily capture and filter stormwater through underlying engineered media	Filtration, adsorption, sedimentation, solar inactivation, and phytoremediation
Constructed Stormwater Wetlands	Vegetated systems constructed to temporarily capture and treat runoff by employing the physical, chemical, and biological mechanisms of naturally occurring wetlands.	Sedimentation, solar inactivation, and phytoremediation
Wet ponds	Permanently inundated systems that capture, store, and slowly release runoff over a 2-5-day period.	Sedimentation and solar inactivation
Infiltration basins	Depressions that are designed to capture and infiltrate runoff into native soils	Filtration, adsorption, and desiccation
Bioswales	Systems designed to filter runoff through an engineered media as it is conveyed from one point to another.	Filtration, adsorption, solar inactivation, desiccation, and phytoremediation
Dry detention ponds	Systems designed to primarily reduce peak flows by detaining and slowly releasing runoff over a 2-5-day period. Systems return to a dry state between storm events.	Sedimentation and solar inactivation
Proprietary devices	System design varies with manufacturer. Generally subsurface, modular systems that treat runoff in compact footprints.	Treatment mechanisms vary with design. Typically includes sedimentation and/or filtration through engineered media or filter fabric.

Sources: NCDEQ, (2017a); Hathaway & Hunt, (2008).

3.3: Methodology

A meta-analysis was conducted to investigate the removal of fecal coliform, *E. Coli*, and *enterococci* from SCMs. The analysis was carried out in July of 2020 using the following databases: Web of Science, ASABE Technical Library, and Google Scholar. Peer-reviewed literature was identified using different combinations of the following search terms: ‘bioretention’, ‘rain garden’, ‘wetland’, ‘wet pond’, ‘permeable pavement’, ‘proprietary system’, ‘manufactured treatment device’, ‘dry detention basin’, ‘dry pond’, ‘sand filter’, ‘infiltration system’, ‘fecal indicator bacteria’, ‘fecal coliform’, ‘*Escherichia coli*’, ‘e*nterococcus*’, ‘stormwater control measure’, ‘best management practice’, ‘stormwater’, ‘removal’, and ‘runoff’. The search strategy was to pair one indicator organism with different SCMs and the term “stormwater”. Searches had no restrictions on journals or publication dates.

To qualify for inclusion in the analysis, peer-refereed publications had to meet the following criteria:

(1) SCMs were installed in-field (i.e., no laboratory, mesocosm, or pilot studies).

(2) SCMs exclusively treated stormwater runoff.

(3) At least one of the following indicator bacteria was monitored: *E coli*, fecal coliform, *enterococci.*

(4) Changes in geometric mean concentrations of the indicator organism for paired samples were reported or able to be determined.

Peer-reviewed publications were excluded for one of the following reasons:

(1) The results for a particular SCM performance were published in multiple articles. In this case only one article was used.

(2) Systems were sited in locations of extreme climate conditions relative to North Carolina or that are significantly different from what is typical of a NC-centric data set.

(3) If it could be determined that the design of the monitored SCM was substantially deviated from the minimum design criteria (MDC) of North Carolina (NCDEQ, 2017a). For example, a bioretention cell evaluated by Hunt et al. (2012) was excluded from analysis because the media depth was less than 0.3 m (1 ft), which is less than one half the minimum (0.6 m, 2 ft) required by North Carolina's MDCs.

3.3.1: Assumptions and Limitations

Maintenance data were not typically provided. It was assumed that the reported data represent typical or average performance of the SCMs.

3.3.2: Data Analysis

Data were extracted and compiled into a Microsoft Excel® spreadsheet for analysis. Microsoft Excel® was used to calculate indicator geometric mean values for studies that provided paired influent and effluent event mean concentrations for the indicator organisms of interest (Equation 1). Removal efficiency was determined using Equation 2. The performance of each SCM type was determined by comparing the average of the reported geometric mean concentrations (i.e., percent reduction) for each SCM type.

(1) Geometric mean = $(A_1 \times A_2 \times A_3 \ldots A_n)^{1/n}$

Where: A_1, A_2, A_3, etc. are individual sample concentrations of a given data set, n is the total number of samples.

(2) Removal efficiency (%): $\frac{Geometric\ mean\ (in) - Geometric\ mean\ (out)}{Geometric\ mean\ (in)}$

3.4: Results

3.4.1: Qualifying Studies

Fourteen studies qualified for inclusion in the analysis (Table 2). All of the studies were peer-reviewed publications except for Tormey (unpublished), which is expected to be published in 2021. Collectively, 32 SCMs were evaluated: bioretention cells (7), constructed stormwater wetlands (CSWs) (7), wet ponds (8), dry detention basins (3), infiltration basins (3), bioswale (1), and MTDs (3). The three MTDs were described as underground detention-based systems, i.e., received no sunlight. MTD 1 and MTD 2 treated runoff by separating settleable and floatable solids, while MTD 3 treated runoff by capturing floatable solids and facilitating sedimentation (Hathaway et al., 2009). Because they were not provided, the geometric mean values were calculated for three studies: Birch et al. (2005), Birch et al. (2006), and Zhang et al. (2012). Summaries of the changes in geometric means for fecal coliform, *E. Coli*, and *enterococci* as reported by each study are provided in Tables 3-3 to 3-5, respectively.

3.4.2: Site Descriptions

The majority of the watersheds were moderately to highly impervious and consisted primarily of residential areas, parking lots, and roadways. Drainage areas ranged from 0.1 ha to 378.4 ha. The average number of samples collected was 12. Nine of the 14 studies were located in moist, subtropical, climate zones signifying warm and humid summer months and mild winters (NOAA, 2020). The remaining five studies can be described as being sited in temperate climate zones, which are characterized as having cooler average temperatures compared to the subtropical zones and a regular distribution of precipitation events during a year (Australian Climate Zones, 2020).

Table 3-2. Summary of characteristics of the included studies.

Study	Site Location	Climate[a]	SCM(s)	FIB[b]
Birch et al. (2005)	Sydney, Australia	Warm Temperate	Infiltration Basin	FC
Hathaway & Hunt (2012)	Willmington, NC	Moist Subtropical	Wet Pond (x2) Bioretention Cell CSW (x2)	EC, EN
Hathaway et al. (2009)	Charlotte, NC	Moist Subtropical	Dry Pond (x2) Wet Pond CSW (x2) Bioretention Cell MTD (x3)	FC, EC
Purvis et al. (2018)	Brunswick County, NC	Moist Subtropical	Bioswale	FC, EN
Krometis et al. (2009)	Durham, Chatham, NC	Moist Subtropical	Wet Pond (x2)	FC, EC EN
Price et al. (2013)	Kure Beach, NC	Moist Subtropical	Dune Infiltration Basin (x2)	EN
Mallin et al. (2002)	Willmington, NC	Moist Subtropical	Wet Pond (x3)	FC
Chandrasena et al. (2016)	Cheltenham, Australia	Mild Temperate	Bioretention Cell (x2)	EC
Petterson et al. (2016)	Melbourne, Australia	Mild Temperate	CSW	EC
Birch et al. (2006)	Sydney Australia	Warm Temperate	Dry Detention	FC
Tormey (Unpublished)	Raleigh, NC	Moist Subtropical	Bioretention Cell	FC, EC
Zhang et al. (2012)	College Park, MD	Moist Subtropical	Bioretention Cell (x2)	FC, EC
Mallin et al. (2012)	Willmington, NC	Moist Subtropical	CSW	FC
Davies & Bavor (2000)	Plumpton, Australia	Mild Temperate	CSW Wet Pond	FC, EN

[a] - Sources: NOAA (2020) and Australian Climate Zones (2020)

[b] - FC: Fecal coliform; EC- *E. Coli*; EN- e*nterococci*

Table 3-3. Changes in geometric mean for fecal coliform as reported by each study.

Study	SCM(s)	no. of Samples	Geometric Mean (in)	Geometric Mean (out)	Removal Efficiency
Birch et al. (2005)*	Infiltration Basin	4	62636[a]	1476[a]	98%
Hathaway et al. (2009)	Dry Pond 1 Dry Pond 2	9 12	1985[a] 1327[a]	2873[a] 1590[a]	-45% -20%
	Wet Pond	14	9033[a]	2703[a]	70%
	CSW 1 CSW 2	9 15	9560[a] 8724[a]	184[a] 3874[a]	98% 56%
	Bioretention Cell	19	2420[a]	258[a]	89%
	MTD 1 MTD 2 MTD 3	7 6 6	667[a] 235[a] 1472[a]	277 368[a] 2379[a]	58% -57% -62%
Krometis et al. (2009)	Wet Pond 1 Wet Pond 2	7 7	49170[b] 66217[b]	69476[b] 45446[b]	-41% 31%
Birch et al. (2006)*	Dry Pond	5	1217[a]	893[a]	27%
Tormey (unpublished)	Bioretention Cell	11	1406[a]	116[a]	92%
Zhang et al. (2012)	Bioretention Cell 1 Bioretention Cell 2	5 5	359[a] 79[a]	202[a] 110[a]	44% -40%
Mallin et al. (2012)	CSW	8	521[a]	42[a]	92%
Davies & Bavor (2000)	CSW Wet Pond	24 24	17000[a] 7900[a]	3600[a] 8100[a]	79% -3%
Purvis et al. (2018)	Bioswale	15	320[b]	111[b]	65%
Mallin et al. (2002)	Wet Pond 1 Wet Pond 2 Wet Pond 3	29 29 29	488[a] 97[a] 74[a]	70[a] 43[a] 85[a]	86% 56% -15%

* Calculated based of EMCs reported

a - Units: CFU/100 mL

b - Units: MPN/100 mL

Table 3-4. Changes in geometric mean for *E. Coli* as reported by each study.

Study	SCM(s)	no. of Samples	Geometric Mean (in)[a]	Geometric Mean (out)[a]	Removal Efficiency
Hathaway et al. (2009)	Dry Pond 1 Dry Pond 2	9 12	915[b] 655[b]	1121[b] 658[b]	-23% 0%
	Wet Pond	10	2122[b]	1153[b]	46%
	CSW 1 CSW 2	6 10	2400[b] 1295[b]	106[b] 864[b]	96% 33%
	Bioretention Cell	14	241[b]	20[b]	92%
	MTD 1 MTD 2 MTD 3	7 6 6	36[b] 4[b] 183[b]	37[b] 14[b] 196[b]	-3% -250 -7%
Krometis et al. (2009)	Wet Pond 1 Wet Pond 2	7 7	6219[b] 9403[b]	6214[b] 4848[b]	0% 48%
Tormey (Unpublished)	Bioretention Cell	11	111[b]	10[b]	91%
Zhang et al. (2012)*	Bioretention Cell 1 Bioretention Cell 2	5 5	193[a] 16[a]	128[a] 39[a]	34% -142%
Petterson et al. (2016)	CSW	5	11000[b]	581[b]	95%
Chandrasena et al. (2016)	Bioretention Cell 1 Bioretention Cell 2	20 6	6270[b] 20250[b]	260[b] 1200[b]	96% 94%
Hathaway et al. (2012)	Bioretention Cell	20	130[b]	39[b]	70%
	CSW 1 CSW 2	18 18	834[b] 425[b]	826[b] 503[b]	1% -18%
	Wet Pond 1 Wet Pond 2	15 18	2483[b] 1273[b]	62[b] 60[b]	98% 95%

* -Calculated based of EMCs reported

a - Units: CFU/100 mL

b - UnitsL MPN/100 mL

Table 3-5. Changes in geometric mean for *enterococci* as reported by each study.

Study	SCM(s)	no. of Samples	Geometric Mean (in)	Geometric Mean (out)	Removal Efficiency
Price et al. (2013)	Infiltration Basin 1 Infiltration Basin 2	24 24	278[b] 315[b]	4[b] 7[b]	99% 98%
Purvis et al. (2018)	Bioswale	15	3451[b]	1411[b]	59%
Krometis et al. (2009)	Wet Pond 1 Wet Pond 1	7 7	14627[b] 30925[b]	30378[b] 19908[b]	-108% 36%
Hathaway et al. (2012)	Bioretention Cell	20	375[b]	39[b]	90%
	CSW 1 CSW 2	18 19	1018[b] 866[b]	316[b] 510[b]	69% 41%
	Wet Pond 1 Wet Pond 2	15 18	2356[b] 274[b]	237[b] 37[b]	90% 86%
Davies & Bavor (2000)	CSW Wet Pond	24 24	6100[a] 1200[a]	900[a] 920[a]	85% 23%

a - Units: CFU/100 mL

b - Units: MPN/100 mL

3.4.3: SCM performance - Fecal Coliform

Summaries of fecal coliform removal by the SCMs are presented in Figures 3-1 to 3-3. The highest reduction of fecal coliform (98%) was achieved by the infiltration basin. Bioretention cells, CSWs, and the bioswale also effectively lowered average geometric mean fecal coliform inflow concentrations by 84%, 78%, and 65%, respectively. Collectively, wet ponds demonstrated poor removal of fecal coliform (5%). MTDs and dry ponds discharged more fecal coliform than what entered. MTDs added 27% more fecal coliform, while dry ponds added 18%.

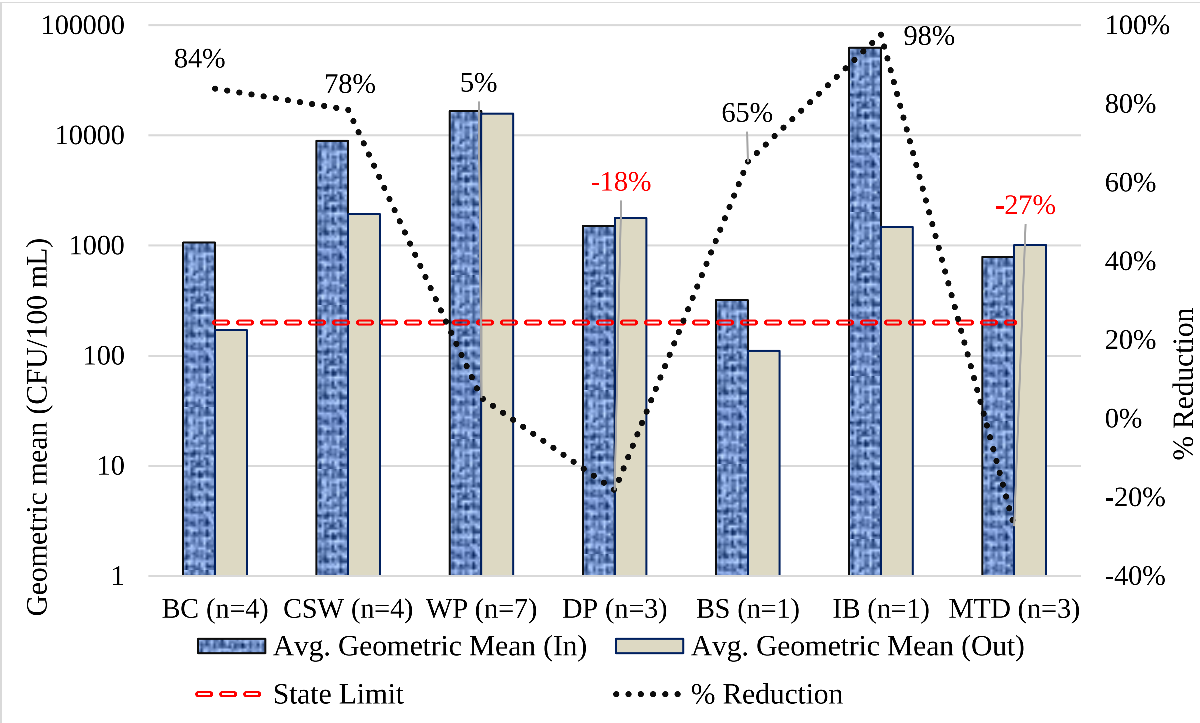

Figure 3-1. Changes in average geometric means concentrations of fecal coliform for each SCM type. Note: BC-Bioretention Cell, CSW-Constructed Stormwater Wetland, WP-Wet Pond, DP-Dry Pond, BS- Bioswale, IB- Infiltration Basin, and MTD-Manufactured Treatment Device.

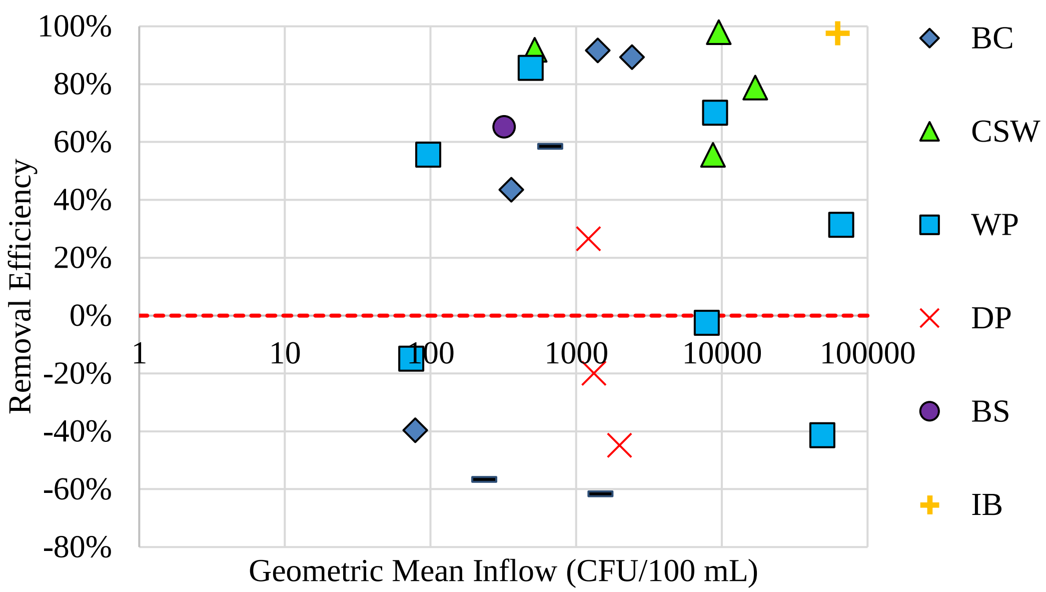

Figure 3-2. Geometric mean inflow vs removal efficiencies for all SCMs. Note: BC-Bioretention Cell, CSW-Constructed Stormwater Wetland, WP-Wet Pond, DP- Dry Pond, BS- Bioswale, IB- Infiltration Basin, MTD-Manufactured Treatment Device.

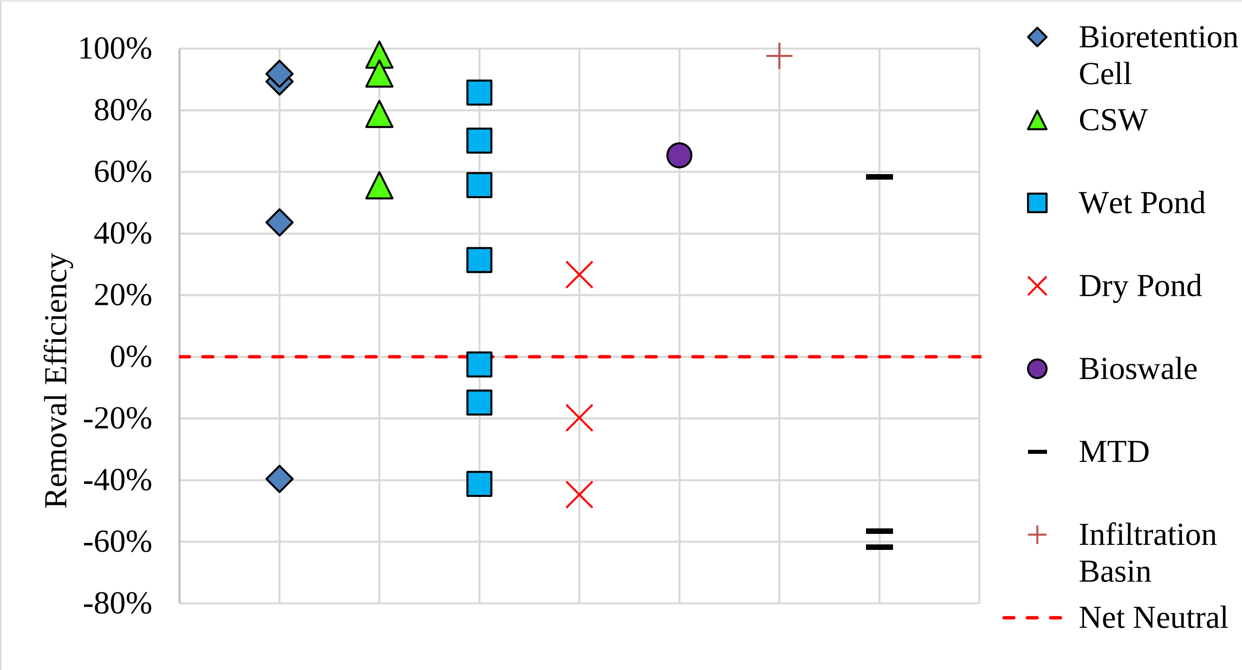

Figure 3-3. Summary of removal efficiencies of fecal coliform per SCM type.

Six SCMs [bioswale (n=1), wet pond (n=1), CSWs (n=2), bioretention cell (n=1), and infiltration basin (n=1)] released geometric mean outflows of fecal coliform that were lower than the NC limit for primary recreation of 200 CFU/100 mL, when subjected to greater than 200 CFU/100 mL geometric mean inflow (NCDEQ, 2019). Further, geometric mean outflows from three other SCMs [bioretention cells (n=2) and MTD (n=1)] exceeded the NC limit by less than 100 CFU/100 mL. Two (wet pond and bioretention cell) of the three SCMs [wet pond (n=2) and bioretention cell (n=1)] with inflow geometric mean concentrations less than 200 CFU/100 mL increased fecal coliform concentrations, but neither exceeded the state limit.

There were two distinct clusters of geometric mean inflow concentrations of fecal coliform that exceeded the NC limit; 200-2500 CFU/100 mL (12 SCMs) and 6000-10,000 CFU/100 mL (5 SCMs). For the SCMs receiving geometric means inflows ranging from 200 to 2500 CFU/100 mL, the bioswale (n=1), bioretention cells (n=3), wet pond (n=1), and CSWs (n=2) reduced fecal coliform concentrations by 65%, 86%, 86%, and 92%, respectively. For these lower concentration inflows, MTDs and dry ponds notably struggled, increasing fecal coliform concentrations by 27% and 18%, respectively. When receiving more polluted runoff (geometric mean inflows ranging 6000-10,000 CFU/100 mL), the CSWs (n=2) and the infiltration basin (n=1) lowered inflow concentrations of fecal coliform by 78% and 98%, respectively. They outperformed the wet ponds (n=2), which removed 36% of fecal coliform. Wet pond performance appeared to diminish [from 86% (n=1) to 36% (n=2)] as fecal coliform inflow concentrations increased; however, sequestration rates for the more polluted runoff varied greatly (removal efficiencies of 70% and -3%) (Figure 3-2).

Four SCMs had geometric mean inflows that were greater than 10,000 CFU/100 mL [wetland (n=1), infiltration basin (n=1), and wet ponds (n=2)]. The wetland and infiltration basin removed 79% and 98% of fecal coliform, respectively, while the performance of the wet ponds was lower and inconsistent (removal efficiencies of -41% and 34%).

3.4.4: SCM Performance - *Escherichia Coli*

E. Coli removal performance is summarized in Figures 3-4 to 3-6. Notably, *E. Coli* was more reliably removed than fecal coliform. Wet pond performance particularly improved (5% fecal coliform removal to 43% *E. Coli* removal), yet wet ponds were still outperformed by bioretention cells and CSW's, which reduced average geometric mean concentrations by 94% and 82%, respectively. MTDs and dry ponds struggled to mitigate *E. coli* and were net sources of *E. coli*. MTD's added 11% more E. Coli, while dry ponds added 13%.

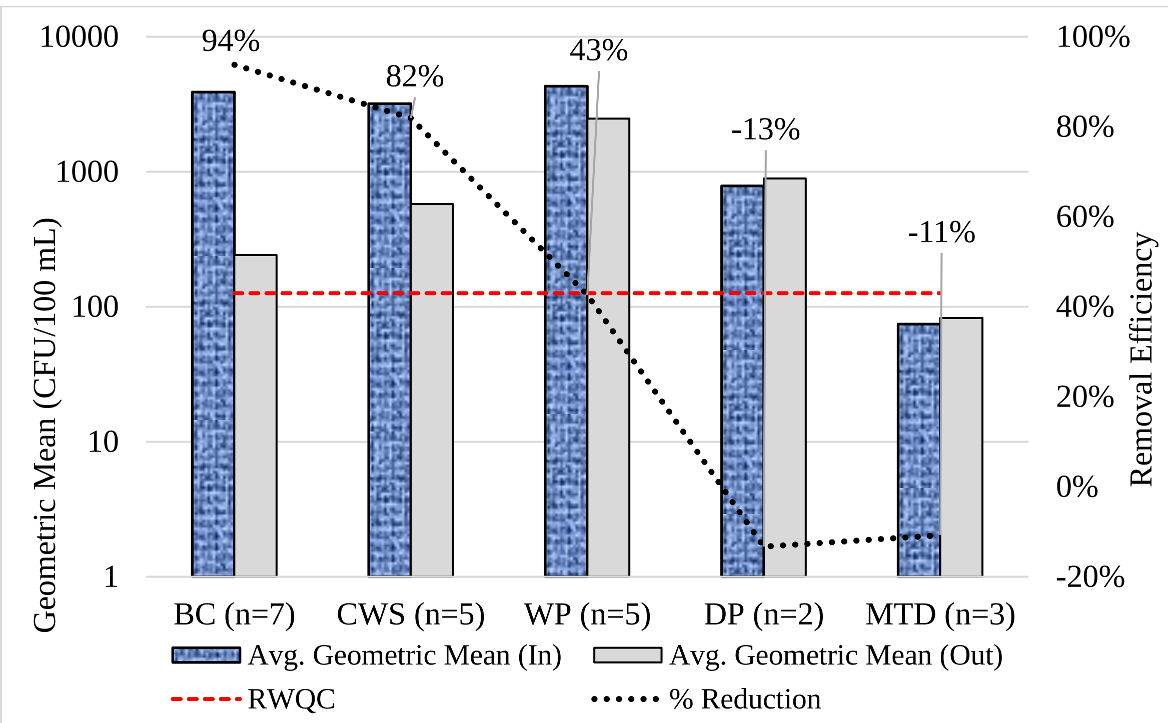

Figure 3-4. Changes in influent and effluent total geometric means of *E. Coli* for each SCM type. BC-Bioretention cell, CSW-Constructed Stormwater Wetland, DP- Dry Ponds, WP-Wet Pond, MTD- Manufactured Treatment Devices. RWQC – USEPA's Recreational Water Quality Criteria for *E. Coli* (126 CFU/100 mL).

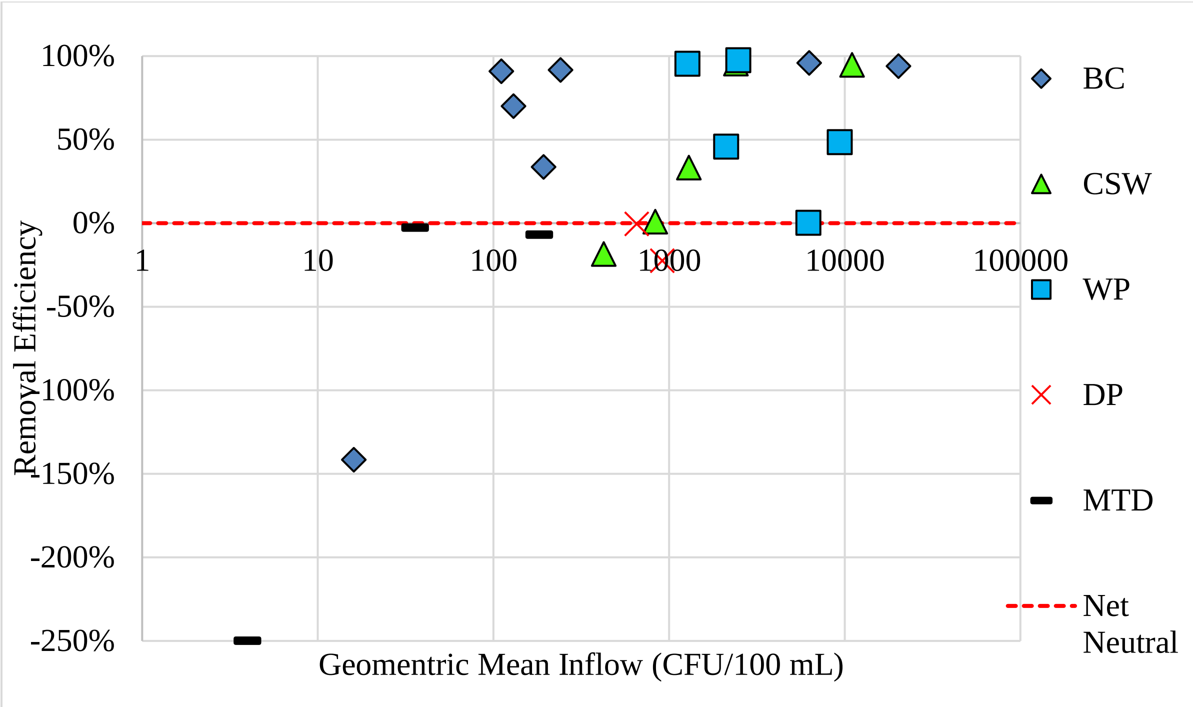

Figure 3-5. Geometric mean inflow vs removal efficiencies for all SCMs. Note: BC-Bioretention cell, CSW-Constructed Stormwater Wetland, WP-Wet Pond, DP- Dry Pond, MTD-Manufactured Treatment Device.

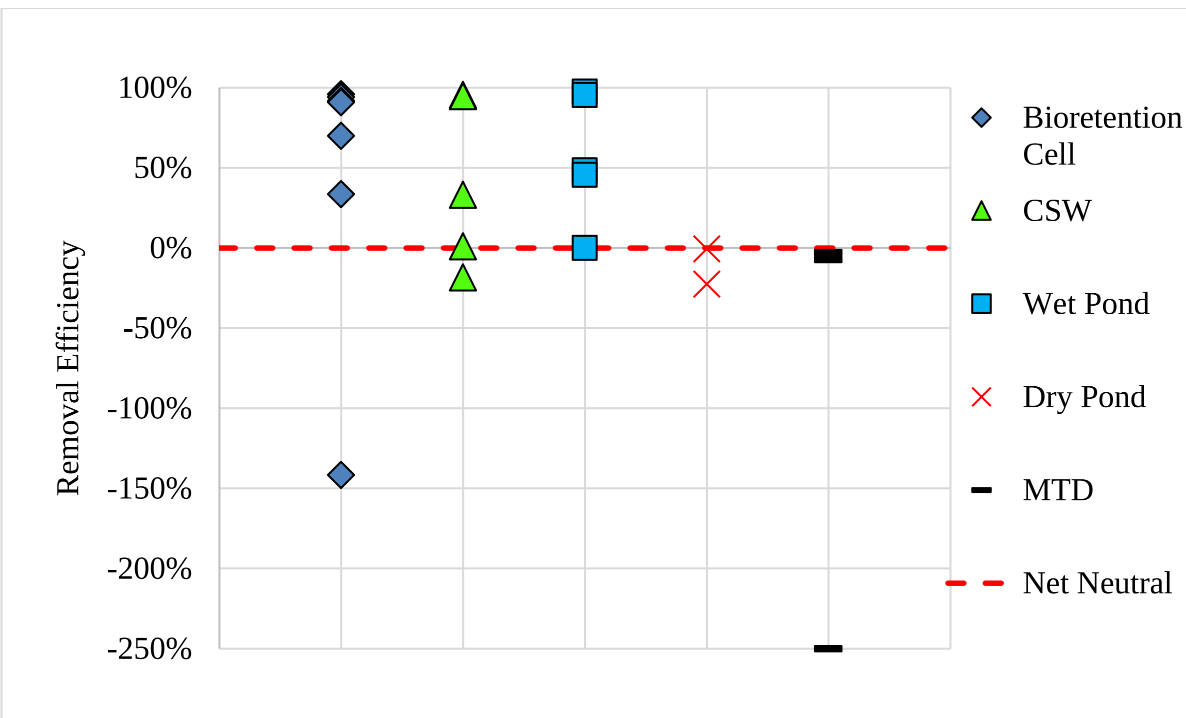

Figure 3-6. Summary of removal efficiencies of *E. Coli* per SCM type.

Five of the 18 SCMs met the USEPA's recommended standards for recreational water quality of 126 CFU /100 mL when receiving influent geometric means greater than 126 CFU/100 mL [bioretention cells (n=2), wet pond (n=1), and wetlands (n=2)]. One bioretention cell and one MTD were within 100 CFU/100 mL of the limit. Three of the four SCMs that received inflows with concentrations lower than 126 CFU /100 mL [Bioretention cell (n=1) and MTD (n=2)] yielded net increases in *E. coli*, but no geometric mean outflow concentrations were greater than the USEPA's thresholds.

As with fecal coliform, two distinct clusters of geometric means for inflows existed: 126-2,500 CFU /100 mL and 6,000-11,000 CFU/100 mL. At the lower inflow range of 126-2,500 CFU/100 mL, both wet ponds (n=3) and bioretention cells (n=3) effectively removed *E. coli,* by 78% and 67%, respectively. CSWs (n=4) reduced geometric mean inflow concentrations by 54%, but the removal efficiencies were highly variable: 96%, 33%, 1%, and -18%. Dry ponds (n=2) and MTD (n=1) were net sources of *E. coli*, increasing geometric mean concentrations by 13% and 7%, respectively.

When more polluted runoff entered the SCMs (6,000-11,000 CFU /100 mL), wet ponds (n=2) were substantially less effective and were highly variable at reducing *E. Coli* concentrations (range of 48% to 0%). One bioretention cell (n=1) and one CSW (n=1), however, had substantially higher efficiencies of 96% and 95%, respectively. One bioretention cell, which received runoff with concentrations >11,000 CFU /100 mL, had a removal efficiency of 94%.

3.4.5: SCM performance - *Enterococci*

Summaries of *enterococci* removal by the SCMs are presented in Figures 3-7 to 3-9. As with fecal coliform, infiltration basins, bioretention cells, CSWs and the bioswale effectively removed *enterococci*. The aggregate removal efficiencies for these SCMs were 98%, 90%, 78%, 59%, respectively. Collectively, wet ponds were net sources of enterococci, increasing geometric mean concentrations by 4%.

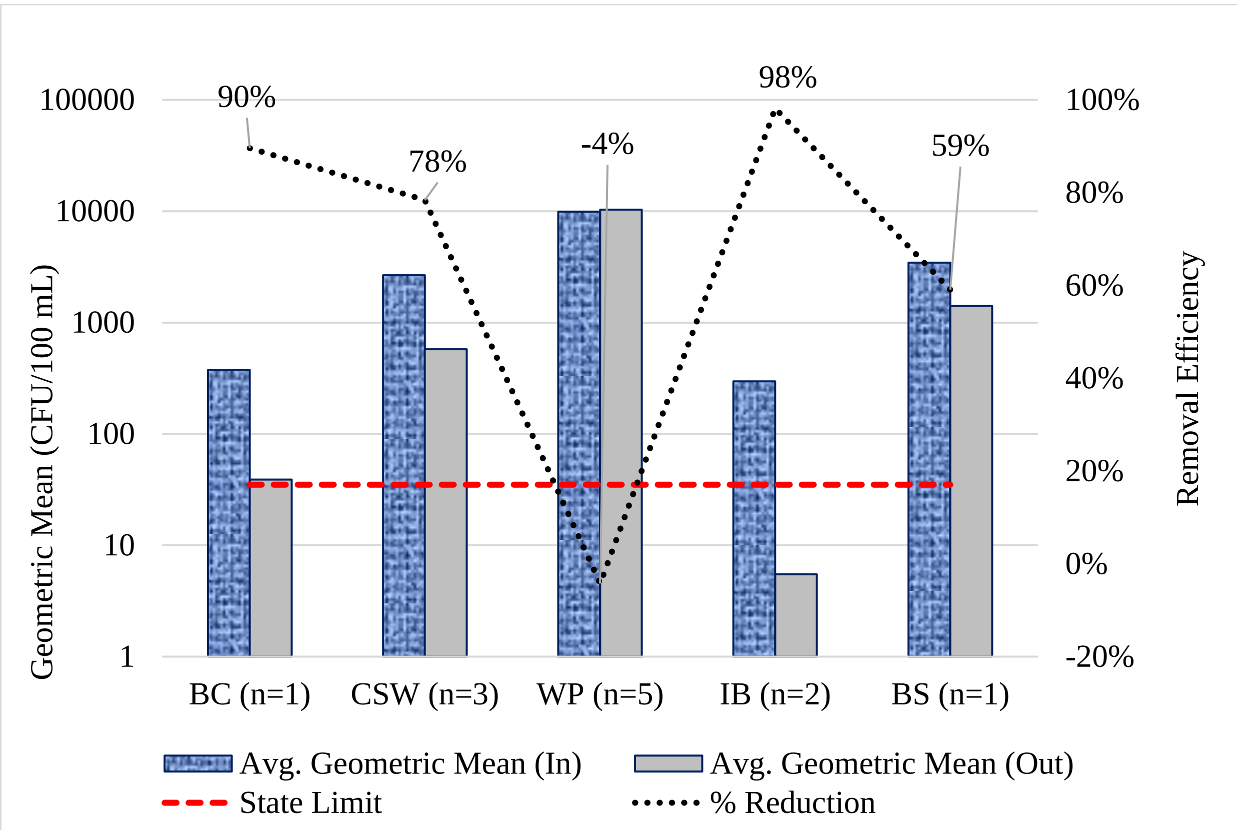

Figure 3-7. Influent and effluent total geometric means of Enterococci for each SCM type. Note: BC-Bioretention cell, CSW-Constructed Stormwater Wetland, WP-Wet Pond, IB- infiltration basin, BS- Bioswale.

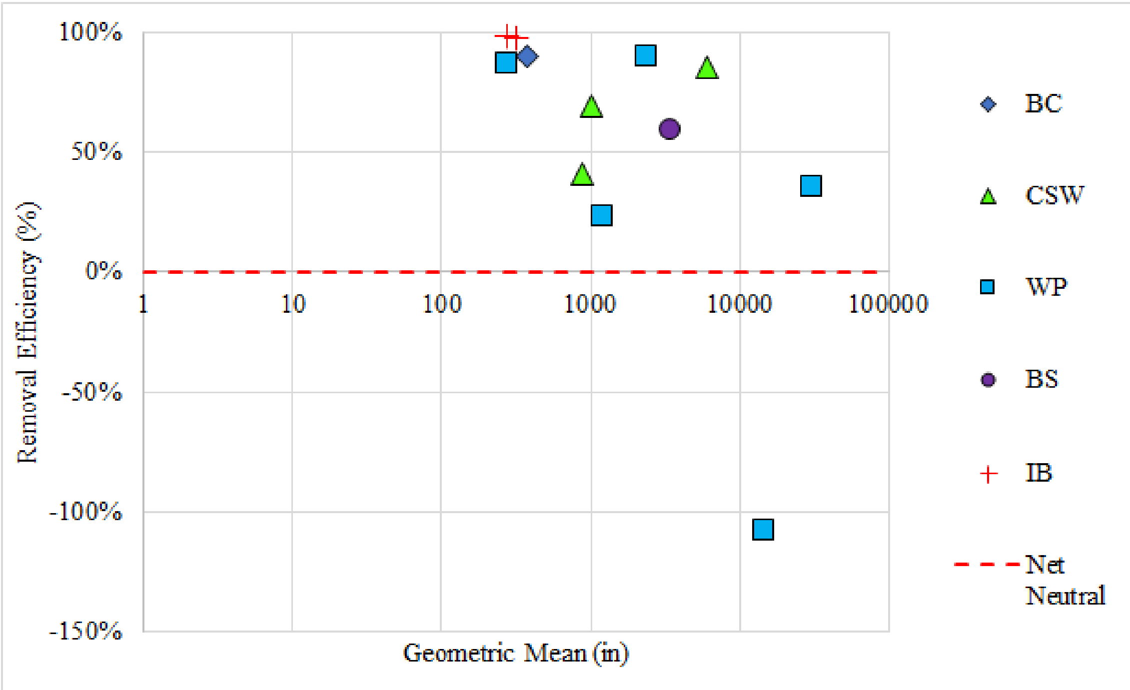

Figure 3-8. Geometric mean inflow (CFU/100 mL) vs removal efficiencies (%) for all SCMs. Note: BC-Bioretention cell, CSW-Constructed Stormwater Wetland, WP-Wet Pond, BS-Bioswale, IB- infiltration basin.

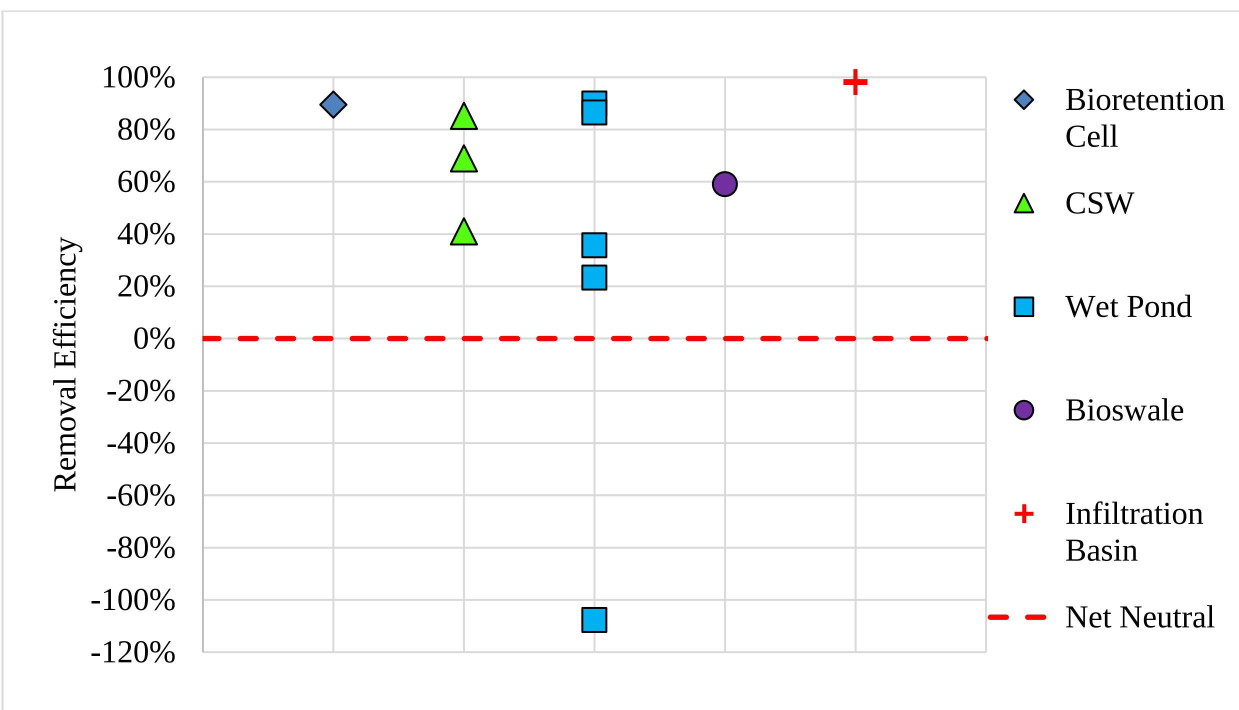

Figure 3-9. Summary of removal efficiencies of *enterococci* per SCM type.

Only the infiltration basins were able to lower *enterococci* concentrations below NC state standards of 35 CFU/100 mL, but effluent from one bioretention cell and one wet pond did have geometric means within 5 CFU/100 mL of the state limit (NCDEQ, 2019). No SCM received geometric mean influent concentrations of enterococci lower than the state limit.

Geometric mean inflow concentrations were generally clustered from 250 to 3500 CFU/100 mL. The change in geometric mean concentrations for infiltration basins (n=2), a bioretention cell (n=1), a bioswale (n=1), wet ponds (n=3), and CSWs (n=2) in this range were 98%, 90%, 59%, 69%, and 56%, respectively. Inflow geometric mean concentrations of two wet ponds and one CSW were higher than 3500 CFU/100 mL. The CSW had a removal efficiency of 85%, while the performance of the wet ponds was lower and varied (removal efficiencies of -108% and 36%).

3.5: Discussion

3.5.1: Variability in SCM Performance

3.5.1.1: Differences in SCM Design

Researchers have noted changes in the performance of bioretention and other filtration-based SCMs when significant changes in their design are present (Hathaway et al., 2012; Chandresena et al., 2016). Variation in media composition can affect properties such as particle size distribution, organic content, surface charge, and detention time, which all impact the efficiency of filtering, straining, and adsorption mechanisms (Peng et al., 2016; Rippy, 2015; Li et al., 2012). Both laboratory (column) experiments and field studies suggest that media depth plays a critical role in pathogen removal (Hathaway et al., 2012; Li et al., 2012). Hathaway et al. (2012) evaluated two bioretention cells in the same watershed with identical surface areas, storage depths, and geometric mean inflows, and found that the shallower media depth (25 cm/ 10 in) system had removal efficiencies (-108% for *E. coli* and -1% for *enterococci*) that were significantly worse than those of the deeper media system (60 cm/24 in) (70% removal for *E. Coli* and 89% removal for *enterococci*). The authors noted that differences in performance may be attributed to differences in soil moisture, inorganic content of the soil, soil temperature, and microbial sorption stemming from the differences in hydraulic function. Moreover, the inclusion of an internal water storage zone in bioretention cells- an area below the aerobic media of the SCM that can remains saturated between storm events (Igielski et al., 2019)- has been suggested to produce conditions that enhance predation, competition, and natural die-off of indicator organisms (Chandrasena et al., 2014; Li et al. 2012; Peng et al., 2016).

In detention-based SCMs where sedimentation is the dominant mechanism for FIB removal, Krometis et al. (2009) and Mallin et al. (2002) suggested that differences in outlet structure design and the length-to-width ratio of the basin can impact removal efficiency by altering hydraulic residence time. Mallin et al. (2002) notes that wet ponds with greater coverage of aquatic vegetation are likely to have better FIB removal.

Vegetation type (aquatic macrophytes) and coverage for both filtration and detention systems appear to increase an SCM's rate of FIB mitigation (Mallin et al., 2002; Chandrasena et al., 2014b). Plant species have varying biological uptake characteristics, antimicrobial root exudates, and microbial communities fostered in the rhizosphere (i.e., differences in competition and predatorial population) (Chandrasena et al., 2014).

Ultimately, it is reasonable to expect that the design of an SCM contributed directly to FIB removal performance, especially within SCM types. Better performing SCMs likely benefited from specific design characteristics, that promoted certain treatment mechanisms, which were more optimized for FIB removal.

3.5.1.2: Variability with Environmental Conditions

As living organisms, the survivability of indicator organisms is predicated on several environmental factors including temperature, salinity, dryness, availability of nutrients, and the microbial community present (Arnone & Walling, 2007). Microorganisms have preferred conditions that maximize their survival and productivity. Hathaway (2010) notes that pathogens tended to persist in moist, cool, and dark conditions. The performance of an SCM reflects its ability to expose trapped pathogens to sunlight, retain moisture, and other means of fostering the conditions that are inhospitable to indicator species. The proportion of indicator organisms partitioned to particles also impacts an SCM's performance. Researchers have noted that attachment increases an indicator organism's resistance to environmental stressors and their subjectivity to predation (Sherer et al., 1992; Krometis et al., 2009). External factors can also impact SCM performance. Andradóttir (2017) noted that the agitative action of wind can effectively hinder the settling of solids (particularly of medium silt and small clay particles) by as much as 20%. In some cases, wind can be strong enough to resuspend settled particles from the bed of stormwater ponds. Thus, wet ponds located in areas with high wind activity may have less efficient FIB mitigation than if sited elsewhere.

3.5.1.3: Comparing Coastal vs Piedmont SCMs

Elevated concentrations of indicator bacteria in surface water are of greater concern in coastal regions owing to the increased public exposure associated with coastal recreational activities. Segregating SCMs between coastal and piedmont environments revealed several key points: Firstly, geometric influent concentrations were generally higher in piedmont areas than coastal areas with the difference being most pronounced for fecal coliform (12,700 CFU /100 mL vs 300 CFU /100 mL). This may also be due to less microbial partitioning in the predominantly sandy soils of the coast (microbes tend to attach to finer soil grains), which increases FIB susceptibility to removal mechanisms (predation, solar inactivation, etc.). Secondly, SCM performance did not vary due to location. The notable exception was wet ponds and how they treated *E. coli* and *enterococci*. Wet ponds performed considerably better in coastal areas than in the piedmont. The average removal efficiencies were 96% (*E. coli)* and 88% (*enterococci)* in coastal areas and 31% (*E. coli*) and -36% (*enterococci*) in the piedmont. Hathaway and Hunt (2012) suggest that this may also be as a result of high water tables in coastal watersheds causing groundwater infiltration to dilute the effluent concentrations of these indicators. Larger grained sands, which are common at the coast, are also less prone to resuspension during a storm (Hathaway and Hunt, 2012).

3.5.2: Ranking SCMs Based on their Performance

The SCMs obtained from the meta-analysis were ranked based on their removal efficiencies (Table 3-6). SCMs in Tier 1 were the most effective at mitigating FIB concentrations and included practices that treated FIB mainly via filtration through engineered media (e.g., bioretention cell and infiltration basin). Tier 2 consists of SCMs that were less effective but still provided some FIB removal. Tier 2 includes SCMs that incorporate mainly sedimentation (e.g., wet pond). While speculative, it appears that phytoremediation may provide CSWs with a modestly preferable performance to wet ponds that allows it to be considered Tier 1 for fecal coliform and *enterococci*. CSWs were placed in Tier 2 for *E. Coli.* Tier 3 consists of the SCMs that generally did not effectively remove FIB. Notably, these SCMs did not employ filtration and only employed temporary sedimentation (i.e., dry pond and the MTDs included herein). The bioswale was not assigned to a tier because of limited data.

Table 3-6. Classification of SCM performance based on average removal efficiencies.

	Fecal Coliform	***E. Coli***	***Enterococci***
Tier 1	• Bioretention Cell • Infiltration Basin • CSW	• Bioretention Cell	• Bioretention Cell • Infiltration Basin • CSW
Tier 2	• Wet Pond	• CSW • Wet Pond	• Wet Pond
Tier 3	• Dry Pond • MTD	• Dry Pond • MTD	

3.5.3: Developing Framework for MTD Performance Evaluation

A framework was developed by referencing the procedure used to develop NCDEQ's standards for primary SCMs (NCDEQ, 2017b). This result was an iterative process that involves reaching three main goals: (1) estimating the geometric mean inflows that SCMs can reduce to NC state limits for fecal coliform and *enterococci* and USEPA's *E. Coli* recommendation for recreational waters, (2) using the collected data herein to gauge how SCMs perform with increasing geometric mean inflows, and (3) adjusting values to ensure that most Tier 1 SCMs meet the standards. The purpose of this framework is to assess whether FIB treatment observed for an MTD is comparable to what is typically achieved by "high performing" non-proprietary SCMs. Elements of the framework are presented in Tables 3-7 to 3-9 and Figures 3-9 to 3-11.

3.5.3.1: Fecal Coliform

Table 3-7. Performance standards for fecal coliform.

Geometric Mean Inflow (CFU/ 100 mL)	Performance Standard
< 200	Invalid
200-1000	200 (CFU/ 100 mL)
>1000	>60 % Reduction

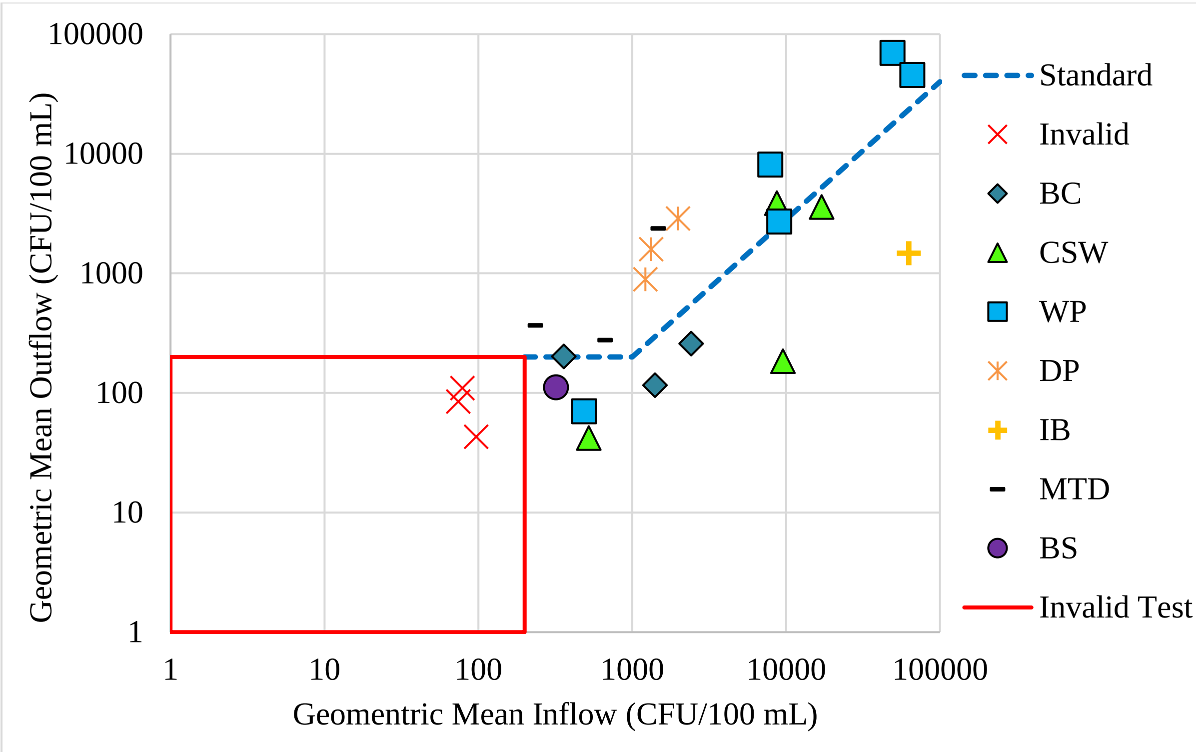

Figure 3-10. Fecal coliform performance standards populated with the SCM performance herein. Note: BC-Bioretention cell, CSW-Constructed Stormwater Wetland, WP-Wet Pond, DP- Dry Pond, BS- Bioswale, IB- infiltration basin, MTD-Manufactured Treatment Device. Tier 1 SCMs include: bioretention cell, constructed stormwater wetland, and infiltration basin.

3.5.3.2: E. Coli

Table 3-8. Performance standards for *E. Coli.*

Geometric Mean Inflow (CFU/ 100 mL)	Performance Standard
< 126	Invalid
126-1000	126 (CFU/ 100 mL)
>1000	>70 % Reduction

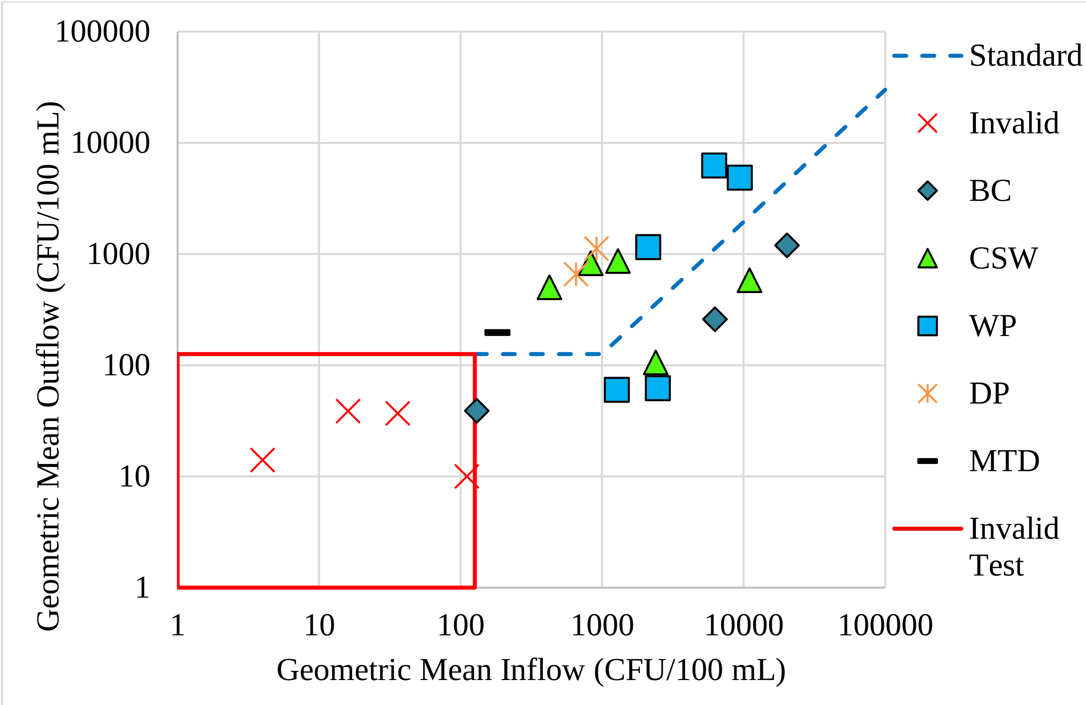

Figure 3-11. *E. Coli* performance standards populated with the SCM performance herein. Note: BC-Bioretention cell, CSW-Constructed Stormwater Wetland, WP-Wet Pond, DP- Dry Pond, MTD-Manufactured Treatment Device. Only BCs are Tier 1 in this category.

3.5.3.3: Enterococci

Table 3-9. Performance standards for *Enterococci*.

Geometric Mean Inflow (CFU/ 100 mL)	Performance Standards
< 35	Invalid Test
35-100	35 (CFU/ 100 mL)
> 100	>50 % Reduction

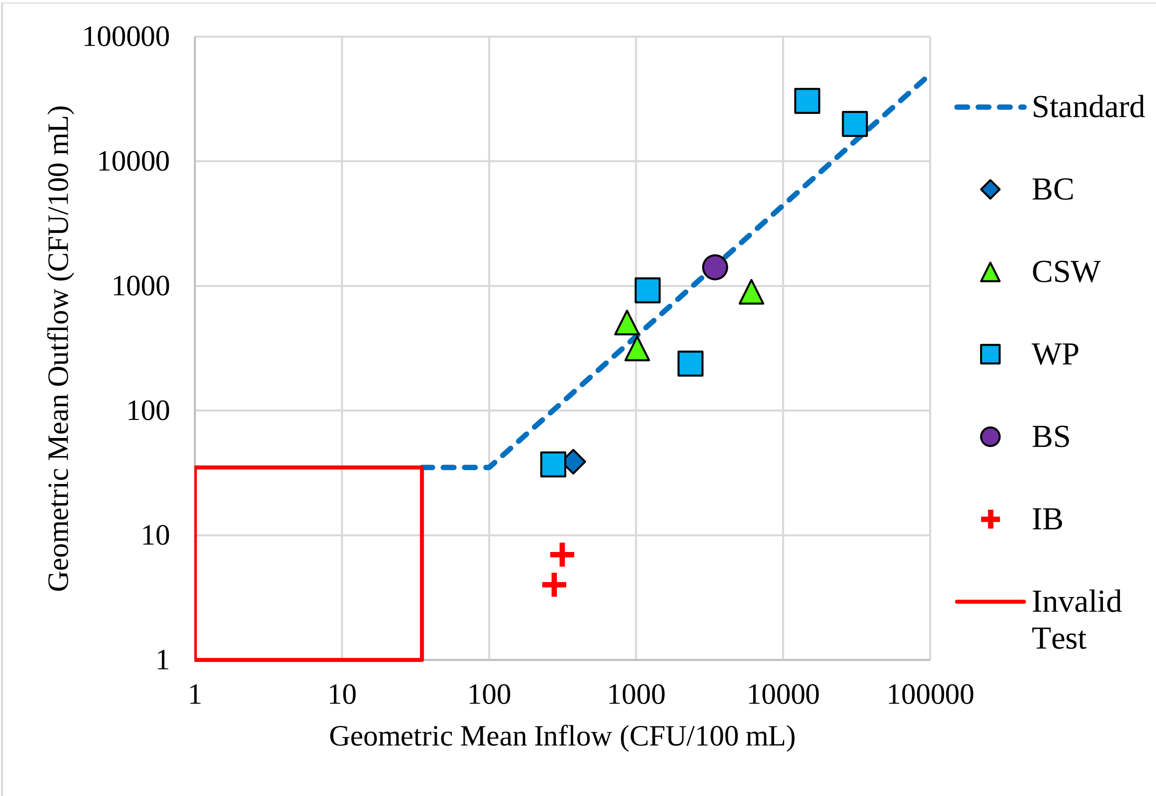

Figure 3-12. *Enterococci* performance standards populated with the SCM performance herein. Note: BC-Bioretention cell, CSW-Constructed Stormwater Wetland, WP-Wet Pond, BS-Bioswale, IB- infiltration basin. Tier 1 SCMs include: infiltration basins, bioretention cells, and constructed stormwater wetlands.

3.6: Summary and Conclusion

Comparisons of the geometric mean influent and effluent concentrations of FIB in SCMs highlighted the complex and somewhat unpredictable nature of pathogen mitigation in stormwater. Removal efficiencies varied widely across SCM types for each of the studied FIB: -62 to 98% for fecal coliform, -250 to 98% for *E. Coli*, and -108 to 99% for *enterococci*. The most effective FIB removal was achieved by SCMs that employed filtration (namely, bioretention cells and infiltration basins). Some (mostly filtration-based) SCMs discharged outflow with geometric mean concentrations that satisfied federal and state water quality regulations, although no one SCM type was able to consistently meet all water quality standards.

After filtration-based SCMs, practices that coupled sedimentation with phytoremediation were best (i.e., CSWs), followed by SCMs with standing water reliably present for sedimentation (i.e., wet ponds). SCMs that did not incorporate filtration or only employed brief sedimentation (i.e., no permanent pools) did not effectively remove FIB. Variability in SCM performance can be primarily attributed to differences in system design (e.g., storage, media depth, presence of vegetation, etc.) and treatment mechanisms in each practice.

These findings facilitated the ranking of SCMs into three tiers based on their ability to sequester FIBs. Tier 1 SCMs were the most effective at reducing FIB concentrations and included bioretention cells, infiltration basins, and for two of three FIBs, constructed stormwater wetlands. Tier 2 SCMs also provided FIB treatment but were generally less effective when compared to those of Tier 1. Tier 2 included wet ponds and, for one FIB, constructed stormwater wetlands. Tier 3 SCMs generally struggled to mitigate FIB and often increased concentrations leaving the systems. Tier 3 SCMs included dry ponds and the MTDs studied herein; these SCMs should be avoided near recreational waters. A framework for assessing the FIB removal performance was also constructed. The metrics can be used to assess whether an SCM's sequestration of FIB is in-line with Tier 1 SCMs.

The ranking of these SCMs and construction of assessment framework are limited to collected data presented herein. There are notable limitations that must be considered with the implementation of this framework. Primarily, the use of grab samples (a representation of the FIB concentration in runoff only at a specific point during a storm event) to assess changes in FIB concentrations creates some uncertainty related to SCM performance through the entirety of a storm event. This is significant because several studies have noted fluctuations in the FIB concentrations in runoff during a storm event (Chow et al., 2013; Paul-Mercado et al., 2016).

Consequently, SCM performance may, to an unknown degree, be misrepresented when using grab samples. Additionally, there was also uneven representation of each SCM type within the study; that is, some SCM types were better studied than others. It is possible that the inclusion of data from future studies would provide more insight to SCM performance and possibly adjust how an SCM type was "ranked." Nevertheless, the framework is meant to serve as the foundation for MTD evaluation and should be revised and updated with future peer-reviewed publications of SCM performance.

3.7: References

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APPENDICES

Appendix A

<u>Ancillary Water Quality Benefits of StormTech's® SCS:</u>

Thermal Buffering

By absorbing solar radiation and transferring it to runoff, urban infrastructure can be a significant source of thermal pollution to receiving streams (Thompson et al., 2008; Janke et al., 2011; Van Buren et al., 2000; Herb et al., 2008). Thompson et al. (2008) reported that the average temperature of an asphalt parking lot was 9.5 ^{o}C (17.1 ^{o}F) higher than that of sod, which translated to an average total mass heat transfer to runoff that was 3.6 times higher than that of sod (Thompson et al., 2008). Janke et al. (2011) reported that some roofs have the potential to export up to 50% as much as paved driveways. Influxes of heated runoff can alter the ambient temperature of a stream, which can adversely impact aquatic species distribution and diversity (Wang and Kanehl, 2003).

Drake et al. (2016) reported that a subsurface detention system was able to reduce inflow runoff temperatures by an average of 5 ^{o}C (9 ^{o}F) in warmer seasons (with an inconsequential change in colder seasons). Natarajan & Davis (2010) highlighted that temperature reduction in underground detention-based systems was attributed to: (1) the ambient temperature in the system, (2) heat transfer to the pipes during stormwater conveyance throughout the system, and (3) mixing of cooled runoff that may have been stored in the system in between events. Thermal conductivity of the pipes and hydraulic retention time also contributed to thermal buffering. Wardynski et al. (2013) surmised that SCMs that transmit and temporarily store stormwater in underground structures (including media) or reduce runoff volume, can mitigate thermal pollution. Although not monitored in this study, the subsurface chamber system is expected to also provide thermal buffering for receiving water bodies based on the treatment mechanisms it employs (i.e., subsurface detention & volume reduction).

Removal of Particulate-bound Pollutants

Heavy metals, fecal indicator bacteria, and hydrophobic organics are examples of three pollutants that adsorb to particulate matter (Li et al., 2012; Vogel and Moore, 2016). David et al. (2010) noted that sedimentation and filtration, which are the main pollutant removal mechanisms employed by the subsurface chamber system, are particularly important to mitigate particulate-bound pollutants. Although speculative, the higher removal of TSS and particulate phosphorus by the subsurface chamber system may be indicative that the system is also likely to reduce large portions of particulate-bound pollutants including heavy metals, FIB, and hydrophobic organics.

Appendix B

Maintenance of StormTech's® SCS

StormTech recommends biannual inspection of the isolator row in the first year of operation, with adjustments made thereafter based on observed sediment and debris accumulation. Maintenance should be performed once more than 3" (76.2 mm) of sediment buildup is measured in the isolator row during inspection. Maintenance involves scouring sediment from the isolator row using a jet-vac and vacuuming away the contaminated water until the blackwash is relatively sediment free. Records of system maintenance conducted at the Capital Oaks subsurface system could not be accessed (StormTech, 2014).