UNDERSTANDING THE RELATIONSHIP BETWEEN URBAN BEST MANAGEMENT PRACTICES AND ECOSYSTEM SERVICES

by

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Abstract

Increasing attentiveness to climate change and the dependence of human life on natural resources has spurred awareness about the detrimental impacts of human activity on the environment. Ecosystem services, or the benefits that humans derive from ecosystems, have changed more in the past 50 years than in any other comparable period in human history (Carpenter et al., 2009). The dilemma of managing the trade-off between immediate human needs and maintaining the ability of the Earth to provide ecosystem services is considered to be one of the largest challenges of this century (Foley et al., 2005). The ecosystem service concept aims to maximize the provision of services across an entire ecosystem to achieve overall ecosystem health through land management, policy, and economic decisions. The intent of this research was to improve such decisions by increasing the understanding about the relationship between urban best management practices and freshwater provision, erosion regulation, and flood regulation ecosystem services. Fifty-six land management scenarios with varying densities of BMP application were simulated using the Stormwater Management Model (SWMM). The ecosystem services resulting from these land management scenarios were quantified using indices developed by Logsdon and Chaubey (2013). Results demonstrate that the application of bioretention cells improve both freshwater provision and erosion regulation services immediately downstream from the implementation site, and an increase in erosion regulation services was observed at the greater watershed scale. There was no change in the provision of freshwater, erosion regulation, or flood regulation services observed by the application of green roofs or rain barrels at either scale of analysis.
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Dedication

I dedicate this thesis to my mother, Kimberly, who is my inspiration and guiding light.

Thank you for being my guardian angel.

I dedicate this thesis to my father, Jeffrey, who encouraged me to chase my dreams.

Thank you for your everlasting patience, guidance, and friendship.
Chapter 1 - Introduction

1.1 Problem Statement

Increasing attentiveness to climate change and the dependence of human life on natural resources has spurred awareness about the detrimental impacts of human activity on the environment. Unprecedented growth in the world population, overexploitation of the Earth’s resources, and dramatic changes in land use have generated overwhelming concern regarding the future state of the environment. Ecosystems and the services they provide have changed more in the past 50 years than in any other comparable period in human history (Carpenter et al., 2009). Croplands and pastures cover approximately 40% of the terrestrial surface in the world today (Foley et al., 2005) and according to the European Commission (2011), thirteen million hectares of tropical forests are cleared each year for agricultural use. Though the majority of these changes have been in effort to provide for the growing world population, essential environmental services, such as water and air purification, are slowly being lost. The overexploitation of land and natural resources is accountable for the loss of approximately 1.5 million hectares of arable land each year (Foley et al., 2005). This alarming decline in the state of the environment incites significant concern about the longevity of natural resources upon which the ever-growing world population is so dependent.

Current policy and scientific research is working to develop methods that encourage environmental conservation and sustainable harvesting of natural resources. The term ecosystem services is a fairly new concept that was defined by the Millennium Ecosystem Assessment (2005) as the benefits people obtain from ecosystems, and without these services, all life on Earth would cease to exist. Unfortunately, 60% of ecosystem services in the world today are being harvested in a manner that is unsustainable (Millennium Ecosystem Assessment, 2005).
The dilemma of managing the trade-off between immediate human needs and maintaining the ability of the Earth to provide ecosystem services is considered to be one of the largest challenges of this century (Foley et al., 2005). The ecosystem service concept is a new idea that applies this term to environmental management programs. This concept takes a holistic approach to maximize the provision of services across an entire ecosystem to achieve overall ecosystem health. “The ecosystem services concept offers a way to deal with and alleviate the ‘dilemma’ of land use change by incorporating effects on environment into land management, policy and economic decisions,” (Logsdon, 2011). By integrating environmental conservation with social and economic development, the ecosystem service concept incorporates the active role that human society plays in today’s environment (Chen et al., 2013).

The current challenge today is in understanding, quantifying, and valuing ecosystem services. Research must first identify and understand the trade-offs and synergies among ecosystem service provision before these services can be quantified and valued economically. Carpenter (2009) attributed the overall decline in ecosystem services to the fact that their true values are not even considered in decision making. The economic cost of replacing ecosystem services with technology would amount to almost twice the entire global gross national production (Costanza, 2012; Zari, 2012). Limited research in understanding the provision of ecosystem services, especially at larger scales such as the watershed level, has led to this lack of understanding. The ability to effectively quantify and value ecosystem services would enable ecosystem managers to provide cost-effective solutions in a synergistic manner (Millennium Ecosystem Assessment, 2005).
1.2 Objectives

The overall goal of this research is to understand the role that holistic watershed management plays in the provision of ecosystem services. The culmination of this research will improve the current understanding of the relationship between urban best management practices and the provision of ecosystem services. Specifically, fresh water provision, erosion regulation, and flood regulation are the three ecosystem services of interest. This research will answer the following questions:

1. What types of ecosystem services are provided by urban best management practices?
2. To what extent is targeted BMP implementation necessary to achieve desirable quantities of ecosystem service provision?
3. Can holistic watershed management across the rural-urban gradient improve provision of ecosystem services within the urban area?

1.3 Significance of Work

The City of Wichita, Kansas is required to improve the quality of water at several identified locations within city limits according to its current Municipal Separate Storm Sewer System (MS4) Permit and National Pollutant Discharge Elimination System (NPDES) stormwater permit. The permit lists six bodies of water within the City of Wichita as impaired (Table 1.1), which means that specific pollutants in the water body exceed the allowable loading limit and therefore do not meet water quality standards. A Total Maximum Daily Load (TMDL) is a pollutant loading limit that is established for those waters categorized as “impaired” under Section 303d of the Clean Water Act. Action must be taken to decrease the pollutant concentration levels in each impaired water body to limits set by the TMDL.
Table 1.1 Wichita impaired streams (Kansas Department of Health and Environment, 2014).

<table>
<thead>
<tr>
<th>TMDL Regulated Pollutant</th>
<th>Specific Impaired Stream to Target</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bacteria</td>
<td>Big Slough, Cowskin Creek, Chisholm Creek, Gypsum Creek, Little Arkansas River, Arkansas River</td>
</tr>
<tr>
<td>Nutrients</td>
<td>Big Slough, Cowskin Creek, Chisholm Creek, Gypsum Creek, Little Arkansas River, Arkansas River</td>
</tr>
<tr>
<td>Sediment</td>
<td>Big Slough, Cowskin Creek, Chisholm Creek, Gypsum Creek, Little Arkansas River, Arkansas River</td>
</tr>
</tbody>
</table>

The MS4/NPDES permit requires the following steps to be taken in order to address the impaired water quality status: 1) implement structural and nonstructural best management practices (BMPs) to reduce the discharge of the TMDL regulated pollutants, 2) establish measurable goals to assess the effectiveness of the TMDL BMPs, and 3) establish an alternative stormwater offsite pollution reduction program when appropriate (Kansas Department of Health and Environment, 2014). To meet the aforementioned requirements, the City of Wichita is considering an innovative off-site best management practice (BMP) implementation program to maximize the environmental and economic effectiveness of their efforts to address water quality and the NPDES permit. This program will enable targeted BMP implementation to high sediment-producing areas within the Middle-Arkansas Slate and Little Arkansas watersheds.

Prior to implementing such a program, the City of Wichita would like to understand potential benefits and tradeoffs in water quality and other ecosystem services that could be realized through an offsite BMP program. The overarching goal of this research is to realize the impact of targeted BMP implementation on ecosystem service provision within urban expanses. The types of ecosystem services provided by urban BMPs will be identified, along with the role that watershed management plays in enhancing ecosystem service provision. This research aims to identify to what extent targeted BMP implementation is necessary to achieve desirable
ecosystem service provision outcomes. This research will also evaluate the applicability of holistic watershed management across the rural/urban gradient for the management of ecosystem services. The culmination of this research should provide the City of Wichita with a clear recommendation as to how to best implement urban BMPs to improve water quality in order to meet the MS4/NPDES permit requirements.

This research additionally has wide implications for sustainable environmental management across the United States and around the world. Millions of dollars are spent each year mitigating the adverse effects of pollution and improving water quality. A greater understanding of ecosystem services and their relationship to best management practices on the watershed scale will provide private and government organizations with necessary information to make informed decisions regarding sustainable land management. It is extremely important to understand how to sustainably manage ecosystems so that the growing world population can harvest necessary resources in a way that will not leave a detrimental impact. The ability to maximize ecosystem services within an urban watershed will provide communities with the tools to maintain clean water, regulate flood control, and minimize erosion.
Chapter 2 - Literature Review

2.1 Understanding Ecosystem Services

The Millennium Ecosystem Assessment (2005) defined ecosystem services as the benefits people obtain from ecosystems. These benefits include an array of services, such as the provision of food or fresh water, and without them, all life on Earth would cease to exist. The Millennium Ecosystem Assessment (2005) classified ecosystem services into four distinct categories based on the services that they provide to humans (Table 2.1). Provisioning services provide services such as food, water, timber, or fiber while regulating services include climate change regulation and temperature control. Supporting services are considered the “behind-the-scenes” services, such as nutrient cycling. Cultural services may be recreation or spiritual services but are largely defined by those humans who use them (Millennium Ecosystem Assessment, 2005).

Table 2.1 Classification of ecosystem services (Millennium Ecosystem Assessment, 2005).

<table>
<thead>
<tr>
<th>Provisioning</th>
<th>Regulating</th>
<th>Supporting</th>
<th>Cultural</th>
</tr>
</thead>
<tbody>
<tr>
<td>• Food</td>
<td>• Climate regulation</td>
<td>• Nutrient cycling</td>
<td>• Aesthetic</td>
</tr>
<tr>
<td>• Freshwater</td>
<td>• Flood regulation</td>
<td>• Soil formation</td>
<td>• Spiritual</td>
</tr>
<tr>
<td>• Wood</td>
<td>• Disease regulation</td>
<td>• Primary production</td>
<td>• Educational</td>
</tr>
<tr>
<td>• Fiber</td>
<td>• Water purification</td>
<td></td>
<td>• Recreational</td>
</tr>
<tr>
<td>• Fuel</td>
<td></td>
<td></td>
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</tr>
</tbody>
</table>

Costanza (2012) further described ecosystem services as “the ecological characteristics, functions, or processes that directly or indirectly contribute to human well-being.” It is, however, important to discern ecosystem services from ecosystem processes and to understand that these two terms are not synonymous. Ecosystem processes describe biophysical relationships in the
environment that continue to exist regardless of human benefit (Costanza, 2012). Since ecosystem services are essentially user-defined, it is important to ensure that all ecosystem processes essential to continuation of life on Earth are included among the list of ecosystem services (Figure 2.1). Ecosystem processes only become ecosystem services once they are used by human beneficiaries (Serna-Chavez et al., 2014).

![Figure 2.1: Ecosystem services and human well-being](image)

The range of services that an ecosystem provides is often used as an indicator for ecosystem health. An ecological system is viewed as healthy if it is active and maintains its organization, autonomy, and resilience over time (Costanza, 2012). Costanza (2012) developed a method for determining ecosystem health in terms of vigor, resilience, and organization. The vigor of an ecosystem is a “measure of its activity, metabolism, or primary productivity,” while the organization of an ecosystem describes “the number and diversity of interactions between
components of a system,” (Costanza, 2012). The resilience of an ecosystem is indicated by the ability of an ecosystem to maintain its structure in the presence of stress and is based off of the length of time it takes for the system to recover as well as the magnitude of the stress event (Costanza, 2012). Ecosystem management should aim to maximize these three characteristics in order to maintain overall ecosystem health.

Unprecedented growth in the world population, overexploitation of the Earth’s resources, and dramatic changes in land use have generated overwhelming concern regarding the future state of the environment. Ecosystems and their services have changed more in the past 50 years than in any other comparable period in human history (Carpenter et al., 2009). This change has been largely in response to growing demands for food, water, timber, fiber and fuel (Millennium Ecosystem Assessment, 2005). The challenge of managing the trade-off between immediate human needs and maintaining the ability of the Earth to provide ecosystem services is considered to be one of the largest challenges of this century (Foley et al., 2005). Though the world population is predicted to level off somewhere around the middle of the 21st century, it is expected that the demand for ecosystem services will only continue to grow. Approximately one-half of global ecosystem production today is used solely to support human activities (Foley et al., 2005).

The domestication of ecosystems has led to substantial gains in overall human well-being, economic development, and technology. Humans have also experienced a significant increase in global food production during the last century. However, the Millennium Ecosystem Assessment (2005) found that 60% of ecosystem services in the world today are being harvested in a manner that is unsustainable. According to Costanza (2012), an ecosystem may be considered sustainable if it has the capacity to survive for a specific, non-infinite, period of time.
This includes maintaining ecosystem health, and the provision of ecosystem services, for the duration of that time. However the unintended consequence of the domestication of ecosystems for the purpose of human advancement has been unexpected and undesirable declines in ecosystem services (Bennett, Peterson, & Gordon, 2009). It has been observed that as human consumption of ecosystem services and goods increases, the quality and condition of the ecosystems that provide those services has declined (Figure 2.2). Overexploitation of land and ecosystem services has led to the loss of approximately 1.5 million hectares of arable land each year (Foley et al., 2005). Another unfortunate side effect of human advancement includes widespread biodiversity loss and decline in ecosystem condition, which diminishes the provision of ecosystem services (Bullock, Aronson, Newton, Pywell, & Rey-Benayas, 2011). Carpenter (2009) noted that a decline in genetic and species diversity will increase the vulnerability of ecosystem services and diminish options for sustainable land management. The European Commission (2011) considers biodiversity loss to be one of the most critical global environmental threats alongside of climate change. “Biodiversity is our life insurance, giving us food, fresh water and clean air, shelter and medicine, mitigating natural disasters, pests and diseases and contributes to regulating the climate,” (European Commission, 2011). It is clear that the continuation of ecosystem exploitation for human advancement will seriously threaten the future existence of human life on this planet.
Urbanization is a significant factor contributing to the decline in worldwide ecosystem service provision. An urban ecosystem is defined as one where “the built infrastructure covers a large proportion of the land surface, or those in which people live at high densities” (Gomez-Baggethum & Barton, 2013). More than 50% of the world’s population today lives in cities, and that percentage is expected to increase to more than two-thirds by the year 2050 (Gomez-Baggethum & Barton, 2013). Urbanization is associated with large populations, amplified energy consumption, and large impervious area, all of which generate increasingly negative impacts on the natural environment. “Despite covering only 2.7% of the world’s surface, cities are responsible for 75% of the global energy consumption, and 80% of greenhouse-gas emissions,” (Grêt-Regamey, Celio, Klein, & Hayek, 2013).
Urban areas containing more than 50% of impervious area can lose anywhere from 40 to 83% of rainfall to surface runoff, which acts as a transport mechanism for sediments and pollutants into surrounding water bodies (Berte & Panagopoulos, 2014; Grêt-Regamey et al., 2013). Yan and Edwards (2013) noted that urbanization is typically associated with an increase in flood discharge and a decrease in water quality, all of which leads to negative impacts on surrounding stream channels and the local water balance. Man-made changes to the natural environment do not even have to be very large in order to effect negative change on ecosystems. Several studies have found that biotic diversity will begin to decline when impervious area covers as little as 10% of the overall watershed area (Yan & Edwards, 2013). Thus any level of urbanization poses a threat to the health of surrounding ecosystems and, subsequently, to those humans that dwell inside them. Even as urban societies adopt an increasingly decoupled and independent existence from the environment, these cities continue to heavily depend on urban ecosystems to sustain long-term conditions for life, health, and security (Gomez-Baggethum & Barton, 2013). This independent existence has decreased the natural ability of urban ecosystems to defend themselves from extreme events such as heat waves, flooding, and water scarcity/droughts (Berte & Panagopoulos, 2014). Many scientists have attributed the rising number and intensity of super-storms, such as Hurricanes Sandy and Katrina, along with the destruction they have left behind, to the inability of ecosystems to defend themselves against natural disasters. Urban ecosystems must be able to maintain the vigor, resilience, and organizational qualities that constitute a healthy ecosystem in order to protect themselves against future extreme events. The sustainable provision of urban ecosystem services is essential for city dwellers to maintain their overall well-being especially as urbanization is expected to increase in coming years (Grêt-Regamey et al., 2013).
One of the most important drivers behind the loss of ecosystem services is land use change and intensity for agricultural use (Maes, Paracchini, Zulian, Dunbar, & Alkemade, 2012). “More land was converted to cropland in the 30 years after 1950 than in the 150 years between 1700 and 1850,” (Millennium Ecosystem Assessment, 2005). Croplands and pastures cover 40% of the terrestrial surface in the world today (Figure 2.3), and in the past 40 years, there has been an approximately 70% increase in irrigated cropland area and a 700% increase in global fertilizer use (Foley et al., 2005). According to the European Commission (2011), thirteen million hectares of tropical forests are cleared each year for agricultural use. However, this mass conversion of land for agricultural purposes has dramatically increased many provisioning services in order to meet the demand of the growing world population and thus the production of food, fuel, and fiber has skyrocketed in the past 50 years. Between 1960 and 2000, the Millennium Ecosystem Assessment (2005) found that “food production increased by roughly two-and-a-half times, water use doubled, wood harvests for pulp and paper production tripled, installed hydropower capacity doubled, and timber production increased by more than half.”
The environmental cost for such a dramatic increase in provisioning services does not come cheaply. “Modern land use practices, while increasing the short-term supply of material goods, may undermine many ecosystem services in the long run, even on regional and global scales,” (Foley et al., 2005). Biodiversity has decreased due to the loss and fragmentation of habitats and there has been a steady degradation of soil and water quality (Foley et al., 2005). Genetic diversity within agricultural crops has decreased by 75% since 1990 (European Commission, 2011). Agriculture accounts for 85% of global consumptive water use even though the provision of fresh water has become increasingly limited in recent years (Foley et al., 2005). “Ecosystem services that agricultural production depends on are being degraded or lost to the
point where current agricultural practices may not be able to be sustained into the future,” (Logsdon, 2011).

It is evident that significant changes in land use have severely altered the balance of ecosystem services in the environment, favoring increases in provisional services and causing declines in regulating and cultural services (Logsdon, 2011). Any notable decline in regulating services should ignite special concern because it foreshadows a future decline in provisioning, supporting, and cultural ecosystem services (Carpenter et al., 2009). Regardless of the reason for land use change, the motive behind the majority of alterations to the natural environment around the world are generally the same: the acquisition of natural resources for immediate humans needs, regardless of the destructive environmental impacts (Foley et al., 2005). “The consumption of ecosystem services, which is unsustainable in many cases, will continue to grow as a consequence of a likely three- to six-fold increase in global gross domestic product by 2050,” (Millennium Ecosystem Assessment, 2005). Human society would do well to recognize and use nature as a tool in order to build a resilient and sustainable society, rather than exploiting and destroying it in a way that threatens the future of our livelihood (Mitsch, 2012).

### 2.2 Ecosystem Management Strategies

In order to successfully maximize the provision of ecosystem services, one must first understand the mechanisms that drive their function and their spatial relationships. “The research evolution of ecosystem services should first define and classify ecosystem services, then research how to quantify these services, and lastly, determine how to value these quantities,” (Logsdon & Chaubey, 2013). Bennett (2009) developed a system to classify the relationships between ecosystem services based on two types of mechanisms causing them: (1) the effects of drivers on
multiple ecosystem services and (2) the interactions between ecosystem services themselves. This classification system will inevitably improve the ability to manage trade-offs and enhance synergies between ecosystem services (Bennett et al., 2009). The spatial relationships between ecosystem services also play a key role here, especially since there may be spatial dissimilarities between where services are produced and where they are used (Serna-Chavez et al., 2014). There is little known about when to expect trade-offs or synergies among ecosystem services, but recognizing the driving mechanisms behind these services and their spatial relationships will bring research one step closer to understanding such interactions.

Ecosystem management encompasses a wide variety of methods that all aim to maximize the synergies and limit trade-offs of ecosystem services (Bennett et al., 2009). First and foremost, ecosystem health should be the desired endpoint for ecosystem management (Costanza, 2012). The Millennium Ecosystem Assessment (2005) found that certain types of land management have the ability to change relationships among ecosystem services, which creates opportunities to enhance multiple ecosystem services simultaneously. “Good management of ecosystems may turn trade-offs that arise at regional scales into opportunities for synergies among ecosystem services at local scales,” (Maes et al., 2012). Spatial and urban planning, as part of the ecosystem management process, have a determinant role in affecting the distribution, quality, and use of ecosystem services and forms the basis of their conservation and enhancement (Berte & Panagopoulos, 2014).

It is important to note, however, that there are some downsides to ecosystem management. “An overly-narrow focus on a limited set of ecosystem services has led to regime shifts with unexpected sudden losses of other ecosystem services,” (Bennett et al., 2009). Carpenter (2009) also noted that actions aimed to increase a single ecosystem service typically
resulted in the reduction in other services. This type of management style can lead to negative long-term impacts on an ecosystem, with the potential to cause irreversible damage. The slow recovery of ecosystems can translate into long-term losses of ecosystem services and persistent problems for ecosystem management programs (Carpenter et al., 2009). “Past actions to slow or reverse the degradation of ecosystems have yielded significant benefits, but these improvements have generally not kept pace with growing pressures and demands,” (Millennium Ecosystem Assessment, 2005).

The ecosystem service concept is a fairly new idea that is gaining popularity among ecosystem management programs. Rather than attempt to increase the provision of a single ecosystem service, this concept takes a holistic approach to maximize the provision of services across the entire ecosystem and obtain overall ecosystem health. “The ecosystem services concept offers a way to deal with and alleviate the ‘dilemma’ of land use change by incorporating effects on environment into land management, policy and economic decisions,” (Logsdon, 2011). By integrating environmental conservation with social and economic development, the ecosystem service concept incorporates the active role that human society plays in today’s environment (Chen et al., 2013). “A sustainable system at the landscape and larger scales will most likely involve a range of human interactions from very intense agro and urban systems to highly protected areas. Determining the optimal structure of this mix is one of the most important ongoing research problems facing us today,” (Costanza, 2012).

The ecosystem service concept uses methods such as ecosystem restoration, low impact development, and green infrastructure to achieve desired outcomes. Ecosystem restoration is a traditional strategy used widely throughout ecosystem management programs to reestablish ecosystem health to a target area. The United States National Academy of Sciences defines
restoration as “the return of an ecosystem to a close approximation of its condition prior to disturbance,” (Mitsch, 2012). However, some scientists consider this definition outdated, arguing that one can no longer design a successful ecosystem as determined by preceding conditions. “Ecosystem management seeks to sustain multiple ecosystem services but often uses, as a reference point, historic conditions that are not achievable in a rapidly changing world,” (Chen et al., 2013). Mitsch (2012) provides a revised definition, calling restoration “the process of assisting recovery of an ecosystem that has been degraded, damaged, or destroyed.” Humans today are a significant determinant of ecosystem health and the design of restored ecosystems should reflect their involvement. However, regardless of how the term is defined or interpreted, restoration has repeatedly proven to be a successful tool to yield ecosystem health, which generates positive environmental impact. Bullock (2011) called ecological restoration a key approach for enhancing the provision of ecosystem services while reversing biodiversity losses. “Actions which target the restoration of ecosystems, and the maintenance of the services they provide, are likely to have positive effects on habitat and species conservation status,” (Maes et al., 2012).

Increasing pressure by the United States Environmental Protection Agency through amendments to the Clean Water Act have made low impact development and green infrastructure a preferred method to manage stormwater and associated environmental issues in a cost-effective and environmentally conscious manner (Dolowitz, Keeley, & Medearis, 2012). Low impact development (LID) is an alternative approach to stormwater management that mimics the natural, pre-development hydrology of a site through the enhancement of hydrologic controls such as infiltration, evaporation, and storage of runoff (Vogel et al., 2015). Green infrastructure (GI) is another management technique that incorporates the network of green space
in both rural and urban areas which work together to enhance ecosystem resilience, encourage conservation of biodiversity and benefit people through the maintenance and enrichment of ecosystem services (Berte & Panagopoulos, 2014). GI and LID are closely related concepts that are often used interchangeably, but the distinction between the two can be drawn along the scale of implementation with GI being applied to larger (watershed) scales and LID having a targeted, localized application (Vogel et al., 2015). One of the major benefits of LID is that it reduces overhead costs for maintenance and construction associated with traditional stormwater management techniques by focusing on prevention rather than mitigation and remediation (Ahiablame, Engel, & Chaubey, 2012). This is done through the implementation of best management practices (BMPs) such as bioretention cells, grassed swales, pervious pavement, stormwater wetlands, green roofs, or rain water harvesting. The targeted placement of these BMPs to mitigate contaminant sources in stormwater runoff is a useful approach to water quality management, based on environmental benefits and cost effectiveness (Tomer & Locke, 2011). BMPs protect water quality in the ecosystem which ultimately provides clean water for humans and species on which humans rely, and may minimize the costs of removing sediments and pollutants downstream (Vigerstol & Aukema, 2011). LID has proven to be an effective method to successfully improve water quality, manage stormwater runoff, and protect the environment (Ahiablame et al., 2012).

Watershed management programs often incorporate a mix of restoration, green infrastructure, and low impact development to improve ecosystem health on a larger scale. These programs establish a mutualistic partnership between science and public policy as a method of identifying and solving environmental issues that pertain to a specific location. Run by localized watershed groups, a watershed management program addresses water management issues
through education, research, technical assistance, direct amelioration of the problem (e.g. streambank restoration), and public advocacy (Prokopy, Mullendore, Brasier, & Floress, 2014). Approximately 77% of watershed management programs in existence today were formed at some point during the 1990s (Duram, Loftus, Adams, Lant, & Kraft, 2008). These programs were often established in response to an environmental stressor such as nonpoint-source pollutants or government regulation (Duram et al., 2008). Prokopy et. al (2014) called this a “catalyst event”, which is an event or series of events that interacts with existing conditions to generate change in the context of watershed management. The catalyst event may be intentional, stemming from government regulation (such as a TMDL), or unintentional, spurred from an event such as flooding or fire (Prokopy et al., 2014).

One of the most popular types of watershed management programs is the collaborative governance model. This model is also known under other names, such as adaptive ecosystem management or an agri-environmental program. The collaborative governance model is popular mainly because it has shifted the focus of environmental management from static structures to adaptive dynamic processes, which achieve more acceptable environmental outcomes (Chen et al., 2013). This model uses methods such as ecosystem restoration or green infrastructure to improve ecosystem health by reducing negative externalities such as nutrient runoff or soil erosion (Baylis, Peplow, Rausser, & Simon, 2007). The U.S. government has relied heavily on collaborative governance models in order to improve agri-environmental performance, although other methods such as cross-compliance and regulation have also been used (Claassen, Cattaneo, & Johansson, 2007). Collaborative governance models are successful not only for their positive environmental outcome, but because they involve a variety of stakeholders who have personal interests in the program. The National Research Council stated that “collaborative planning
involves diverse community interests within the watershed. There is no one leader and no outside expert telling people what is best for them. Rather, it is the collective effort to develop a vision and then make that vision become a reality,” (Duram et al., 2008).

Systems analysis is a method of assessment that assists with decision-making in a watershed management program to achieve water quality goals (McGarity, 2013). A SWOT analysis is one type of qualitative systems analysis that evaluates the strengths, weaknesses, opportunities, and threats of a specified area (Berte & Panagopoulos, 2014). This methodology helps decision-makers identify the actual state of the environment within a watershed through the collection of qualitative information. “The emerging field of watershed systems analysis is adapting these tools, primarily simulation, optimization, and multi-objective analysis to formulate models that inform and guide this process,” (McGarity, 2013). It has been found that watershed management programs that employ some form of a systems analysis are more successful in the long run because they have identified environmental processes and issues on a larger ecosystem scale. “System thinking is required when ecosystems are created or restored. It is not the time to think about linear cause and effects but rather the ecosystem as a whole,” (Mitsch, 2012).

There are several notable examples of successful watershed management programs that restored ecosystem health to an area using many of the aforementioned strategies and methods. The New York City Watershed Agreement of 1997 is an excellent example of a large scale watershed management program. The City of New York avoided the construction of a multi-billion dollar water filtration facility by working with communities upstream of the city to implement BMPs and encourage environmental conservation (Appleton, 2002; Grolleau & McCann, 2012). In this project, Appleton (2002) found that traditional command and control
regulation were not successful when the economic livelihood of individual farmers and other rural landowners was at stake. Thus, a bottom-up approach was taken where local stakeholders, rather than outside experts, decided the issues and discussed their options (Duram et al., 2008). Munich, Germany also developed a watershed management program based on the collaborative governance model that encouraged farmers to switch to organic farming as a way to improve drinking water quality (Grolleau & McCann, 2012). The Cheney Lake Watershed Project in south central Kansas combined the system analysis theory with the collaborative governance model to ameliorate taste and odor problems in their drinking water (Becerra, 2010). Each of these programs used a variety of methods to achieve success, although each found that equal representation, similar values, and a unified approach among stakeholders were especially important. Successful watershed management relies on making decisions through face-to-face negotiations among stakeholders; addressing all pollution sources in a watershed concurrently; and collecting extensive information to achieve consensus on pollution problems and to find win-win solutions (Borisova, Racevskis, & Kipp, 2012).

2.3 Ecosystem Service Quantification and Valuation

Many scientists and industry professionals have attempted to quantify and value ecosystem services in recent years, however there is a significant lack of information regarding the accuracy and application of methods for the quantification and valuation of ecosystem services. Even further, many industry professionals have developed economic valuation methods for ecosystem services without even first considering quantifying the actual services that are provided. This has led to inaccuracies in ecosystem service market value, and thus trade-offs and synergies are not usually negotiated effectively (Grêt-Regamey et al., 2013). Carpenter (2009)
attributed the overall decline in ecosystem services to the fact that their true values are not even considered in decision making. The economic cost of replacing ecosystem services with technology would amount to almost twice the entire global gross national production (Costanza, 2012; Zari, 2012). The ability to effectively quantify and value ecosystem services would enable ecosystem managers to provide cost-effective opportunities in a synergistic manner (Millennium Ecosystem Assessment, 2005).

The first step in measuring ecosystem services is to determine proven and reliable methods of quantification. “Several research teams have begun developing tools to quantify and visualize water related ecosystem services, arguing that if these services could be quantified, or at least visualized, stakeholders and leaders would be more likely to use this information as part of the decision-making process that would ultimately yield more sustainable choices,” (Vigerstol & Aukema, 2011). The majority of current quantification methods are quite simple and often only compare a single ecosystem service at one time rather than assessing the entire ecosystem and its services as a whole. This can lead to inaccurate information regarding trade-offs or synergies and limit ecosystem management.

Ecosystem services valuation should be completed following quantification to ensure accurate and true economic valuation. Gomez-Baggethum and Barton (2012) developed a classification system for urban ecosystem service valuation. Ecosystem services can either have a social/cultural value or an insurance value. A social or cultural value “reflects emotional, affective, and symbolic views attached to urban nature that in most cases cannot be adequately captured by commodity metaphors or monetary metrics,” (Gomez-Baggethum & Barton, 2013). Ecosystem services with insurance value play a major role in increasing the resilience and adaptive capacity of cities and their counterparts (Gomez-Baggethum & Barton, 2013). By
categorizing ecosystem services using this method, environmental managers will be able to
determine the economic impact of the loss of a particular ecosystem service in an urban
ecosystem. This information would be particularly helpful to decision-makers and policy-
makers.

The most notable work on ecosystem service quantification and valuation has been
completed by The Nature Conservancy and the University of Vermont. The Nature Conservancy,
along with Stanford University, developed a model called the Integrated Valuation of Ecosystem
Services and Tradeoffs (InVEST). InVEST is a GIS-based model that allows for ecosystem
service quantification and valuation (Natural Capital Project, 2007). Unfortunately, InVEST is
only capable of modeling one ecosystem service or function at one time and requires a
significant amount of data input that may not be readily available (Logsdon, 2011). The
University of Vermont, along with several NGO partners, developed a model called the Artificial
Intelligence for Ecosystem Services (ARIES). ARIES evaluates ecosystem service flows in a
system to quantify and value the services provided. ARIES even considers the location of human
beneficiaries and sink locations in its model (Villa, 2014). ARIES web-based interface is easy for
user-navigation, though some users have expressed difficulty with understanding the
relationships used in the model (Logsdon & Chaubey, 2013; Vigerstol & Aukema, 2011). The
programming behind ARIES is somewhat complex and unfamiliar to many researchers, which
can making using the program quite challenging (Bagstad, Semmens, Winthrop, Jaworkski, &
Larson, 2012).

Most recently physical models have become a popular method of conducting ecosystem
service quantification. “Physical models enable calculation of runoff volumes and nonpoint
pollution loads from precipitation data and land characteristics including soil types, topography,
developed land uses, and storm sewer networks,” (McGarity, 2013). The Technical Release 55 (TR-55) Urban Hydrology for Small Watersheds is one of the more simplistic physical models as it allows for single-event rainfall-runoff simulations in small watersheds. The TR-55 was developed by the Soil Conservation Service in 1975 to calculate storm runoff volume, peak rate of discharge, and storage volumes (USDA NRCS, 2015). Due to its simplicity, the TR-55 is a popular choice for modelers who intend to simulate either rural or urban small watersheds (Table 2.2). However, there are significant limitations of TR-55 such as the inability to simulate subsurface flow or lack of continuous simulation modeling (USDA NRCS, 2015). Simulation modeling of more complex watersheds should use other physical models, such as the Soil and Water Assessment Tool or the Storm Water Management Model.

The Soil and Water Assessment Tool, or SWAT, is a physically-based hydrologic model that was developed to evaluate the impact of land management on hydrology and water quality (Logsdon, 2011). SWAT allows both event and continuous simulation of watersheds at any scale, and is especially effective for agricultural areas (Table 2.2). A number of agricultural best management practices are built-in to the SWAT model for simulation. However, this focus on rural and agricultural land management makes SWAT an unlikely choice for modelers working in urban areas. Logsdon and Chaubey (2013) developed a method of quantification based off of mathematical indices to represent ecosystem service provision in a rural watershed as determined by the outputs from a SWAT model. The mathematical indices are fairly useful in determining improvements or declines in ecosystem service provision and can be applied across a variety of physical models.

The United States Environmental Protection Agency developed the Storm Water Management Model, or SWMM, to simulate quantity and quality problems associated with
runoff from urban areas (James, Rossman, & James, 2010). It is particularly adept at modeling urban hydrology and water quality cycles, including rainfall, snow melt, surface runoff, transport through the drainage network, storage and treatment, and receiving water effects (Barco, Wong, & Stenstrom, 2008). SWMM has a unique advantage over other physical models due to its ability to simulate hydraulic dynamics of artificial drainage systems (Wu, Thompson, Kolka, Franz, & Stewart, 2013). SWMM is especially effective at simulation of best management practices in urban watersheds due to the variety of built-in BMPs provided by the software (Table 2.2). However, SWMM is limited in the number of BMPs that are allowed per subcatchment, which can be a significant drawback in land management simulation. Additionally, SWMM is not quite as adept at modeling rural and agricultural watersheds. Modelers who intend to simulate rural areas should look to other physical models, such as SWAT.

Table 2.2 Comparison of popular physical models.

<table>
<thead>
<tr>
<th></th>
<th>TR-55</th>
<th>SWAT</th>
<th>SWMM</th>
</tr>
</thead>
</table>
| **Advantages** | - Low input complexity  
- Single event simulation  
- Urban and rural application | - Combined hydrologic and hydraulic model  
- Event & continuous simulation  
- Includes agricultural BMPs  
- Links to GIS | - Combined hydrologic and hydraulic model  
- Event & continuous simulation  
- Several BMP types  
- Can link to GIS  
- Effective for urban simulations |
| **Disadvantages** | - Does not consider subsurface flow  
- Minimum $t_c$ is 0.1 hour, maximum $t_c$ is 10 hours  
- Maximum 10 subwatersheds  
- No built in BMPs  
- No link to GIS  
- No continuous simulation modeling | - Medium/high input complexity  
- Limited in urban BMP choices  
- Limited use in urban areas | - Medium/high input complexity  
- Limited number of BMPs allowed per subcatchment |
Future efforts that integrate observational and modeling studies will offer the best opportunity to move conservation science and policy forward, in cooperation and partnership with stakeholders who recognize the critical importance of managing water quality (Tomer & Locke, 2011). Much of current policy contains ordinances and regulations that prevent the implementation of LID practices in some municipalities (Ahiablame et al., 2012). Overcoming legislative barriers and adopting policies that encourage sustainable development practices will be a huge step forward to achieve wide-spread acceptance. In terms of scientific research, continued field and experimental data collection is needed to evaluate LID practices over a variety of climate conditions (Ahiablame et al., 2012). Vogel et al. (2015) calls for future research to focus on the “optimization of widespread GI implementation in both time and scale, including treatment trains, watershed-based planning, and designing for the entire life cycle of the system.” Logsdon and Chaubey (2013) request research to investigate using models to simulate ecosystem functions not captured by SWAT to analyze ecosystem function and further develop indices for quantification. The enhancement of evaluation metrics and modeling techniques will improve overall system performance at all levels of implementation (including the watershed level).
Chapter 3 - Methods

3.1 Study Area

One of the main objectives of this research was to understand the role of holistic watershed management across the rural-urban gradient to improve the provision of ecosystem services within the urban area, and thus it was necessary to include the entire City of Wichita and the surrounding hinterlands within the study area (Figure 3.1; Figure 3.2). The majority of the study area lies within the Middle-Arkansas Slate Watershed (HUC #11030013), with small portions in the Little Arkansas Watershed (HUC #11030012) and the Gar-Pearce Watershed (HUC #11030010).

Figure 3.1 Location of the Gar-Pearce, Little Arkansas, and Middle-Arkansas Slate Watersheds in Kansas.
Figure 3.2 Location of the study area encompassing the City of Wichita, KS and portions of the Gar-Pearce, Little Arkansas, and Middle-Arkansas Slate Watersheds.

The total drainage region for this study area is 877 km², of which approximately 43% is used for agricultural purposes and 31% of land is in urban land uses (Figure 3.3). The Arkansas River and the Little Arkansas River flow through the study area, merging together at the center of the City. Cowskin Creek, Gypsum Creek, Chisholm Creek, Dry Creek and the Big Slough are the main tributaries to these two rivers (Figure 3.4). Out of the two rivers and five tributaries, six are listed as impaired according to the City of Wichita’s MS4/NPDES permit (Kansas Department of Health and Environment, 2014). Bacteria, nutrients, and sediment are the leading causes of impairment along various reaches of Cowskin Creek, Chisholm Creek, Gypsum Creek, Big Slough, the Little Arkansas River, and the Arkansas River (Kansas Department of Health and Environment, 2014). There are ten United States Geological Survey (USGS) gage stations operating in the watershed with continuous long-term streamflow data (Table 3.1). USGS gage #07144200 is the longest running station, with daily continuous data since 1922. USGS gage
#07144480 has supplied hourly precipitation data in addition to streamflow data since 2012. Precipitation data for the remainder of the study area was obtained from the National Oceanic and Atmospheric Administration (NOAA) National Climatic Data Center (NCDC). There are nine NCDC rain gages throughout the study area that provide continuous daily and hourly precipitation data (Table 3.2).

![Figure 3.3 Land-use percentages for the study area.](image-url)
Figure 3.4 Major rivers and tributaries in the study area.

Table 3.1 USGS gages within the study area.

<table>
<thead>
<tr>
<th>USGS Gage Number</th>
<th>Location</th>
<th>Latitude</th>
<th>Longitude</th>
<th>Date of Operation</th>
</tr>
</thead>
<tbody>
<tr>
<td>07144300</td>
<td>Arkansas River</td>
<td>37°38’36”</td>
<td>-97°20’06”</td>
<td>10/1/1934-present</td>
</tr>
<tr>
<td>07144325</td>
<td>Gypsum Creek</td>
<td>37°38’49”</td>
<td>-97°16’49”</td>
<td>3/1/1983-11/30/1984</td>
</tr>
<tr>
<td>07144330</td>
<td>Dry Creek</td>
<td>37°40’20”</td>
<td>-97°16’45”</td>
<td>10/21/1965-9/6/1966</td>
</tr>
<tr>
<td>07144301</td>
<td>Arkansas River</td>
<td>37°42’58”</td>
<td>-97°24’07”</td>
<td>2/9/2015-present</td>
</tr>
<tr>
<td>07143375</td>
<td>Arkansas River</td>
<td>37°46’53”</td>
<td>-97°23’22”</td>
<td>3/1/1987-present</td>
</tr>
<tr>
<td>07144200</td>
<td>Little Arkansas</td>
<td>37°49’56”</td>
<td>-97°23’19”</td>
<td>6/10/1922-present</td>
</tr>
<tr>
<td>07144480</td>
<td>Cowskin Creek</td>
<td>37°42’06”</td>
<td>-97°28’50”</td>
<td>4/1/2001-present</td>
</tr>
<tr>
<td>07144486</td>
<td>Calfskin Creek</td>
<td>37°40’27”</td>
<td>-97°28’49”</td>
<td>10/1/2010-present</td>
</tr>
<tr>
<td>07144490</td>
<td>Cowskin Creek</td>
<td>37°39’56.5”</td>
<td>-97°27’27.7”</td>
<td>10/1/2010-present</td>
</tr>
<tr>
<td>07144550</td>
<td>Arkansas River</td>
<td>37°32’39”</td>
<td>-97°16’31”</td>
<td>10/1/1968-present</td>
</tr>
</tbody>
</table>
Table 3.2 NCDC rain gages within the study area.

<table>
<thead>
<tr>
<th>NCDC Gage Number</th>
<th>Latitude</th>
<th>Longitude</th>
<th>Date of Operation</th>
</tr>
</thead>
<tbody>
<tr>
<td>US1KSSG0002</td>
<td>37°41’29.76”</td>
<td>-97°28’46.92”</td>
<td>6/22/2006-present</td>
</tr>
<tr>
<td>US1KSSG0003</td>
<td>37°43’29.28”</td>
<td>-97°28’42.96”</td>
<td>6/22/2006-present</td>
</tr>
<tr>
<td>US1KSSG0009</td>
<td>37°46’30.36”</td>
<td>-97°22’9.48”</td>
<td>2/01/2007-present</td>
</tr>
<tr>
<td>US1KSSG0020</td>
<td>37°40’44.4”</td>
<td>-97°30’23.04”</td>
<td>2/24/2008-present</td>
</tr>
<tr>
<td>US1KSSG0026</td>
<td>37°47’52.8”</td>
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<td>6/05/2008-present</td>
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<tr>
<td>US1KSSG0064</td>
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<td>9/22/2010-present</td>
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<td>12/01/1953-present</td>
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<tr>
<td>USW00003974</td>
<td>37°44’46”</td>
<td>-97°13’16”</td>
<td>7/01/1996-present</td>
</tr>
</tbody>
</table>

3.2 SWMM Model Development

3.2.1 Introduction to SWMM

In order to understand the impact of holistic watershed management on the provision of ecosystem services, a physical model was needed to simulate hydrologic function. The Storm Water Management Model (SWMM) is a computer modeling program used for single event or continuous simulation of runoff quantity and quality in urban areas (James et al., 2010; L. A. Rossman, 2010). SWMM was first developed by the Environmental Protection Agency (EPA) in 1971 and has undergone several revisions since that time. The EPA released the most recent version of this modeling program, **SWMM 5**, in 2004 (James et al., 2010). This latest version of SWMM provides an integrated environment for editing input data, running hydrologic and water quality simulations, and viewing the results in several different formats (L. A. Rossman, 2010).

Computational Hydraulics International (CHI) is a private company that develops and maintains professional software systems. CHI released an expanded version of the EPA-SWMM modeling program, known as **PC-SWMM**. PC-SWMM extends the capability of the original software to include many tools that improve the professional and scientific use of SWMM 5.
PC-SWMM is the modeling software that was used to model hydrology in the study area encompassing the City of Wichita and its peri-urban areas.

There are two major modeling components that are used in SWMM hydrologic simulation. The runoff component of SWMM simulates subcatchment rainfall-runoff processes and determines associated runoff and pollutant loads (L. A. Rossman, 2010). The routing component of SWMM transports runoff through a series of user-defined pipes, channels, natural waterways, storages, etc. Through these two components, SWMM has the capability to determine the quality and quantity of runoff in each subcatchment as well as the flow rate, depth, and water quality in each routing component during the simulation period (L. A. Rossman, 2010). Other hydrologic processes that SWMM has the ability to simulate include time-varying rainfall, evaporation of surface water, snow accumulation and melting, rainfall interception from depression storage, infiltration of rainfall into unsaturated soil layers, percolation of infiltrated water into groundwater layers, interflow between groundwater and the drainage system, and nonlinear reservoir routing of overland flow (L. A. Rossman, 2010). Spatial variability in the aforementioned processes is achieved by dividing each subcatchment into an assortment of smaller, homogenous areas. Each smaller sub-area contains its own fraction of pervious or impervious areas. Overland flow is then routed between these smaller sub-areas, between subcatchments, or between various entry points of the drainage system (L. A. Rossman, 2010).

SWMM treats each subcatchment surface as a nonlinear reservoir to compute the runoff component of the model. Subcatchment inflows include precipitation and runoff from upstream subcatchments, while infiltration, evaporation, and surface runoff are considered outflows (L. A. Rossman, 2010). The maximum depression storage of a subcatchment represents the capacity of the reservoir, taking into account ponding, surface wetting, and interception (L. A. Rossman,
Surface runoff occurs when the depth of water in the reservoir exceeds the maximum depression storage.

SWMM calculates the infiltration of surface water into the unsaturated soil zone of pervious subcatchments using either Horton’s Equation, the Green-Ampt Method, or the Curve Number Method. Horton’s Equation is based off of empirical observations that demonstrate an exponential decrease in infiltration from an initial maximum rate to a minimum rate over time (L. A. Rossman, 2010). The Green-Ampt Method models infiltration under the assumption that a sharp wetting front separates soil with some initial moisture content from the saturated soil above (L. A. Rossman, 2010). The Curve Number Method is based off of the NRCS (SCS) Curve Number Method for runoff estimation and assumes that the total infiltration capacity of a soil can be estimated from its tabulated curve number (L. A. Rossman, 2010). Each method requires a variety of different input parameters. SWMM gives the option for the user to select the infiltration model of their choice.

The routing component of the SWMM model is governed by conservation of mass and momentum as defined by the Saint-Venant flow equations for gradually varied, unsteady flow (L. A. Rossman, 2010). SWMM provides the user with the choice of three routing methods for model simulation; steady flow routing, kinematic wave routing, and dynamic wave routing. Steady flow routing is the simplest option available because it assumes that flow is uniform and steady within each time step in the model (L. A. Rossman, 2010). Due to its simplicity, this routing method cannot account for channel storage, backwater effects, entrance/exit losses, flow reversal or pressurized flow (L. A. Rossman, 2010). These limitations really only make steady flow routing appropriate for simulation of dendritic conveyance networks or preliminary analysis of long-term continuous simulations. Kinematic wave routing is slightly more complex than
steady flow routing. “This routing method solves the continuity equation along with a simplified form of the momentum equation in each conduit,” (L. A. Rossman, 2010). However, models using the kinematic wave routing option are also restricted for use in dendritic conveyance networks and cannot account for backwater effects, entrance/exit losses, flow reversal, or pressurized flow (L. A. Rossman, 2010). Kinematic wave routing will maintain numerical stability for simulations with moderately large time steps (on a scale of 5 to 15 minutes), and therefore is not recommended for models requiring time steps on a smaller scale (L. A. Rossman, 2010). Dynamic wave routing produces the most theoretically accurate results of the three routing methods because it solves the complete one-dimensional Saint-Venant flow equations (L. A. Rossman, 2010). Unlike the previous routing methods, dynamic wave routing has the ability to represent pressurized flow and can account for channel storage, backwater effects, entrance/exit losses, and flow reversal. Dynamic wave routing can be applied to any general network layout and SWMM will automatically adjust the user-defined time step to maintain numerical stability in the simulation (L. A. Rossman, 2010).

The SWMM modeling software has been used in both research and municipal applications around the world. Typically, SWMM is used to design drainage systems for flood control, size detention facilities for flood control and water quality, design strategies for minimizing combined sewer overflows, generate non-point source pollutant loadings, and to evaluate the effectiveness of BMPs for reducing pollutant loadings (L. A. Rossman, 2010). SWMM’s ability to accurately simulate hydrologic and hydraulic processes in both urban and rural areas makes it the preferred modeling software for this particular research project.
3.2.2 Building the SWMM Model

3.2.2.1 Pre-Processing the Data

Much of the input data for the model was first processed in ArcGIS before it was used in PC-SWMM. ArcGIS, or geographic information system, is a computer modeling software that was developed by ESRI for data analysis and interpretation (ESRI, 2015). The first step in developing the SWMM model was to delineate the study area using a digital elevation model (DEM) to determine the flow accumulation and hydrology. A 3-meter spatial resolution DEM of the study area was obtained from the United States Department of Agriculture (USDA) Natural Resources Conservation Service (NRCS) Geospatial Data Gateway (USDA NRCS, 2015). Vaze, Teng, and Spencer (2010) found that the quality of DEM-derived hydrological features are sensitive to both DEM accuracy and resolution, and that many detailed topographic properties are lost as DEM resolution becomes coarser. Thus, a 3-meter spatial resolution DEM, the highest resolution DEM available for the study area, was selected to provide topographical information for the study area (Figure 3.5). ArcSWAT, an extension of ArcGIS modeling software, was used to complete the study area delineation. ArcSWAT automates the delineation process, limiting the potential for user error and automatically calculating several necessary SWMM input parameters. The delineation process split the study area into a total of 189 subcatchments (Figure 3.6). ArcSWAT calculated the area, slope, length, and elevation for each individual subcatchment from data contained in the DEM.
Figure 3.5 USDA-NRCS DEM for the study area.

Figure 3.6 Map of study area with 189 subcatchments.

SWMM defines subcatchments as “hydrologic units of land whose topography and drainage system elements direct surface runoff to a single discharge point,” (L. A. Rossman, 2010). SWMM splits each individual subcatchment into pervious sub-areas, where infiltration
occurs, and impervious sub-areas. The impervious sub-areas are further divided into regions that contain depression storage and regions that do not (L. A. Rossman, 2010). Each subcatchment established in the delineation process was assigned a reach, which represented one of the major rivers or smaller tributaries within the study area. ArcSWAT calculated the length, slope, width, maximum elevation and minimum elevation for every reach. Each reach segment was connected to another reach either up- or downstream by a node demarking the outlet of each model subcatchment (Figure 3.7). The reaches and nodes representing the surface hydrology of the study area were used as the foundation of the development of the SWMM model.

![Figure 3.7 Reaches, nodes, and subcatchments.](image)

### 3.2.2.2 Model Construction

PC-SWMM uses a total of seven layers for model construction (Table 3.3). Of the seven layers available, only the subcatchment, junction, outfall, storages and conduits layers were used in the construction of this particular model. A description of each of the required inputs and their units is available in the PC-SWMM online user’s manual.
Table 3.3 PC-SWMM layers and required inputs.

<table>
<thead>
<tr>
<th>Layer</th>
<th>Description</th>
<th>Inputs</th>
</tr>
</thead>
<tbody>
<tr>
<td>Subcatchments</td>
<td>Area of land where topography and hydrology components direct surface runoff to a single discharge point</td>
<td>Name, Outlet, Area, Width, % Slope, % Impervious, Manning’s N, Depression Storage, Routing, Infiltration</td>
</tr>
<tr>
<td>Junctions</td>
<td>Drainage system nodes where links join together, such as the confluence of natural surface channels.</td>
<td>Name, Location, Invert Elevation, Max. Depth, Initial Depth, Surcharge Depth, Ponded Area</td>
</tr>
<tr>
<td>Outfalls</td>
<td>The terminal node of a drainage system</td>
<td>Name, Location, Invert Elevation, Tide Gate, Type</td>
</tr>
<tr>
<td>Storages</td>
<td>Storages are essentially drainage system nodes that provide a storage volume</td>
<td>Name, Location, Invert Elevation, Max. Depth, Initial Depth, Ponded Area, Storage Curve</td>
</tr>
<tr>
<td>Conduits</td>
<td>Pipes or channels that move water from one node to another in the conveyance system</td>
<td>Name, Inlet Node, Outlet Node, Shape, Max. Depth, Length, Roughness, Inlet Offset, Outlet Offset, Initial Flow, Max. Flow, Entry/Exit Loss Coefficient</td>
</tr>
</tbody>
</table>

Of the seven inputs needed to parameterize the subcatchment layer (Table 3.3), the name, outlet, area, and percent slope values were automatically calculated by ArcSWAT from the DEM in the delineation step. The subcatchment width value was determined by dividing the ArcSWAT-produced overland flow length by two (L. A. Rossman, 2010). The percent impervious of each subcatchment was calculated using data from the 2011 National Land Cover Data Set (NLCD) at a spatial resolution of 30 meters, which was obtained from the USDA-NRCS GeoSpatial Data Gateway (USDA NRCS, 2015). Of the land uses present in the study area; cultivated crops, deciduous forest, woody wetlands, open water, herbaceous, hay/pasture, emergent herbaceous wetlands, mixed forest, barren land, and shrub/scrub were all considered pervious. Estimates of the impervious area within the four remaining land-use categories (Developed Open Space, Developed Low Intensity, Developed Medium Intensity, and Developed High Intensity) were estimated by applying an impervious factor as outlined in the
National Land Cover Database (2006) (Table A.1). Values for the Manning’s roughness coefficient (Manning’s N) for overland flow were obtained from literature (Table A.4). A weighted average Manning’s N was determined for the pervious and impervious area of each subcatchment based off of the area of each land-use classification. Values of depression storage were obtained from the SWMM user’s manual (L. A. Rossman, 2010) and assigned based upon the land-use category (Table A.5). A weighted average depression storage was also determined for each subcatchment based off the area and land-use classification. The subarea routing for all subcatchments was set to pervious, indicating that the model will simulate runoff flowing from impervious areas to pervious areas before leaving the subcatchment at the outlet (L. A. Rossman, 2010) with the value of percent routed determined in calibration (Appendix D). Infiltration was modeled using the Green-Ampt equation. Values for inputs to the infiltration model (suction head, effective hydrologic conductivity, and initial deficit) were assigned based upon soil survey spatial and tabular data obtained from the USDA-NRCS GeoSpatial Data Gateway (USDA NRCS, 2015). The average soil type and associated texture within each subcatchment was determined from this dataset. Soil suction head and porosity were obtained from the SWMM User’s Manual (L. A. Rossman, 2010) and assigned based off of the soil texture class (Table A.6), while the effective hydrologic conductivity was given in the soil survey.

A combination of datasets was used to determine values for the seven inputs required to parameterize the junctions layer in SWMM (Table 3.3). SWMM describes junctions as drainage system nodes where conduits join together (L. A. Rossman, 2010). Junctions were used to represent two types of nodes in the model: nodes linking natural surface hydrology and man-made nodes linking engineered stormwater infrastructure. The invert elevation for nodes linking reaches was set to equal the maximum elevation of the downstream reach, and the ponded area
was set to equal the width of said reach. For these junction types, values of maximum depth, initial depth, and surcharge depth were ignored. The location of each junction (latitude and longitude) were obtained from the study area delineation in ArcGIS. The City of Wichita Public Works & Utilities Office of Stormwater Management provided a database of each man-made junction within the City that included all necessary information.

A single outfall for the entire model was used to represent the most downstream point in the study area. “Outfalls are terminal nodes of the drainage system used to define final downstream boundaries under Dynamic Wave flow routing,” (L. A. Rossman, 2010). Under any under routing mechanism, the outfall behaves normally as a junction. The name, location, and elevation input parameters were obtained from ArcSWAT and GIS using the same process as the junction parameters. No tide gate was assigned and the outfall type was free.

Storage units are essentially junctions in the model that provide some quantity of physical storage volume, and represent storage entities such as a catch basin or a lake (L. A. Rossman, 2010). The storages layer of the SWMM model requires seven input parameters (Table 3.3). The name, location, and invert elevation of each storage was provided by the City of Wichita. The maximum depth of each storage was calculated as the distance between the permanent water level and top berm, the elevation of which were determined using the interpolate line tool and profile graph within ArcGIS. A storage curve was developed by using the measure tool in ArcGIS to identify the area of the detention pond with minimum water and the area of the detention pond at high water. The initial depth and ponded area were both set to zero.

SWMM describes conduits as an integral part of the conveyance system that move water from one junction to another (L. A. Rossman, 2010). Dynamic wave routing was assigned as the routing mechanism to convey water through the system. The conduits layer of the SWMM model
needs twelve input values (Table 3.3). The delineation step in ArcGIS provided values for the name, inlet node, outlet node, and length input parameters for natural hydrologic channels in the study area. The geometry of model conduits representing major rivers and their tributaries was determined using the interpolate line tool and profile graph to view the cross-section of each reach in ArcGIS. The Manning’s N value for each of the natural channels in the SWMM model was set to 0.051 (L. A. Rossman, 2010). Information needed to parameterize engineered conduits in the storm sewer network were provided by the City of Wichita. The values for initial flow, maximum flow, and entry/exit loss coefficient were ignored.

The base SWMM model was constructed with only natural hydrologic channels and did not include any of the storm sewer network from the City of Wichita (Figure 3.8). Due to the extensive detail and complexity of the City of Wichita’s storm sewer network, this data was included as part of the nested model.

![Diagram of the base SWMM model.](image)
3.2.2.3 Pollutant Wash-Off Functions

SWMM can effectively simulate the generation, inflow, and transport of any number of user-defined pollutants and their impact on water quality (L. A. Rossman, 2010). Pollutant build-up and wash-off from a subcatchment is determined by mathematical functions assigned to user-defined land uses. These land uses are used to solely account for the spatial variation in pollutant build-up and wash-off rates within subcatchments (L. A. Rossman, 2010). Pollutant wash-off of total suspended solids (TSS), total nitrogen (TN), and total phosphorus (TP) were simulated in this model by assigning specific land use categories to each subcatchment within PC-SWMM. Mixed development, pavement, roofs, and undeveloped were the four land use categories used in this model. The mixed development category encompassed all developed pervious areas from NLCD land use data set. Based on the inspection of the relative area of rooftops to other impervious surfaces in the study area, 80% of the area identified as developed impervious from the NLCD land use data set was assigned to the pavement category while the remaining 20% was assumed to represent roofs. The undeveloped category represented any remaining land, such as grassland or forested areas from the NLCD land-use data set. Each land use category was assigned a wash-off function in the land-use editor for each pollutant of interest to determine overall pollutant loading in the model.

Pollutant wash-off from a given land use category occurs during wet weather periods, i.e. during a storm event, and is calculated using one of the following equations; exponential wash-off, rating curve wash-off, or event mean concentration. An exponential function was used to calculate TSS wash-off in the model. The initial parameters for the TSS exponential wash-off function were estimated using the Universal Soil Loss Equation, which is a mathematical model commonly used for describing soil erosion. The parameters for the TSS exponential wash-off
function were refined through the calibration and validation process. An event mean concentration function was used to compute TN and TP wash-off in the model. The initial parameters for the TN and TP exponential wash-off function were estimated using values obtained from literature, and then refined through the calibration and validation process.

After pollutant wash-off occurs, SWMM computes pollutant concentrations throughout the network conveyance system using water quality routing mechanisms. Pollutant concentrations at the end of a conduit are calculated by integrating the conservation of mass equation (L. A. Rossman, 2010).

3.2.2.4 The Nested Model

The nested model is an extension of the base SWMM model to include detailed hydrologic characteristics for specified subcatchments at a scale appropriate for simulation of BMP scenarios. The subcatchments chosen for the nested model were the same subcatchments where targeted BMP implementation would occur, and were focused within the watershed of a smaller tributary of the Arkansas River (Cowskin Creek). The sub-watershed of Cowskin Creek was selected for targeted BMP implementation due to its extensive issues with flooding and water quality impairment. The subcatchments selected included subcatchment 154, 169, 177, 180, and 184. Each aforementioned subcatchment was delineated again using the ArcSWAT extension of ArcGIS to divide the area into micro-subcatchments. The same methodology to build the larger base model was used to construct the SWMM model in these smaller micro-subcatchments. The only difference between the base model and the nested model was the addition of storage units, the City of Wichita’s stormwater network, and BMPs. The addition of
these components to the model allows for more accurate simulate of the hydrologic influences present throughout the watershed (Figure 3.9).

Figure 3.9 The nested model.

3.2.4 SWMM Model Calibration and Validation

The SWMM model for the study area was calibrated for streamflow, total suspended solids (TSS), total nitrogen (TN), and total phosphorus (TP). Streamflow calibration and validation for the SWMM model was conducted at node N190 on Cowskin Creek and the outfall node on the Arkansas River (Figure 3.10). Node N190 on Cowskin Creek was calibrated using data from USGS Gage Station #07144490 and the outfall node was calibrated to the USGS Gage Station #07144550 on the Arkansas River downstream of the City of Wichita. Pollutant calibration and validation for the SWMM model was conducted at the outfall node on the Arkansas River using data from EPA Station SC281 and the EPA STORET database.
The Sensitivity-based Radio Tuning Calibration (SRTC) tool in PC-SWMM was used for model calibration for streamflow. The SRTC tool allows the user to calibrate each individual model parameter to a specified level of uncertainty. The SRTC tool runs PC-SWMM two times for each parameter: once for each high and low percentage of the selected uncertainty range (CHI Support, 2014). Once run, the SRTC tool has a slider bar that allows the user to optimize the model parameters within the uncertainty range to match the simulated hydrograph with the observed (measured) hydrograph.

The SRTC tool optimizes model parameters according to the Nash-Sutcliffe Efficiency (NSE) and automatically reports the NSE value between the simulated and observed hydrographs. The Nash-Sutcliffe Efficiency (Equation 3.1) is a mathematical relationship that
determines the level of fit between a simulated and observed hydrograph (Nash & Sutcliffe, 1970).

\[
NSE = 1 - \frac{\sum_{i=1}^{n}(Q_{oi} - Q_{si})^2}{\sum_{i=1}^{n}(Q_{oi} - \overline{Qo})^2}
\]

(Equation 3.1)

Where:

- \(Q_{oi}\) = observed flow rate (m\(^3\)/s)
- \(Q_{si}\) = simulated flow rate (m\(^3\)/s)
- \(\overline{Qo}\) = mean value of observed flow rate (m\(^3\)/s)

The Nash-Sutcliffe efficiency may range from \(-\infty\) to 1, where a computed value of NSE=1 indicates a perfect match of simulated model results and observed data (Nash & Sutcliffe, 1970).

In terms of water quality, the simulated pollutant loads for each individual pollutant was compared to the observed pollutant loads using the NSE. The build-up and wash-off functions for each pollutant parameter must be adjusted to improve the NSE between the simulated pollutograph and the observed pollutograph.

### 3.3 Ecosystem Service Analysis

#### 3.3.1 Identification of Ecosystem Services

As stated in the literature review, ecosystem services are the human-defined benefits, arising from ecological characteristics, functions or processes, that people obtain from ecosystems (Costanza, 2012; Millennium Ecosystem Assessment, 2005). The types of ecosystem services that humans benefit from range from food and fiber provision to recreational enjoyment. Three ecosystem services were chosen for analysis in this research project. Fresh water provision (FWP), erosion regulation (ER), and flood regulation (FR) were the three services chosen for analysis due to their importance to the City of Wichita and their dependence on hydrologic processes.
3.3.2 Quantitative Indices

Logsdon and Chaubey (2013) developed a set of mathematical indices to represent ecosystem service provisioning for selected provisional and regulatory ecosystem services using the outputs of a physical, process-based model. These indices will provide a basis to compare changes in ecosystem service provision amongst various holistic land management scenarios in this research.

Three quantitative indices were calculated to determine changes in fresh water provision, erosion regulation, and flood regulation. Data for these calculations was obtained from the model output at three site locations within the SWMM model (Figure 3.11). The first two locations were used to determine the quantitative indices for fresh water provision and erosion regulation. The first site was located at the outfall node of the entire model along the Arkansas River, just downstream of the City of Wichita. The second site was located at node N190, which is along Cowskin Creek just downstream from the targeted BMP implementation area. The assessment of fresh water provision and erosion regulation services at the outfall of the entire model provided insight into the impact of targeted BMP implementation on the watershed scale. The evaluation of these two ecosystem services at node N190 provided information about the immediate impact of BMP implementation on a more localized scale. The difference in fresh water provision and erosion regulation between these two locations was valuable in understanding the role that targeted BMP implementation plays throughout the watershed.
The third location in the SWMM model was used to quantify and understand the impact of targeted BMP implementation on flood regulation services. This site was located at node N154-28 along Cowskin Creek. Node N154-28 represents the actual location of USGS Site #07144480 where flooding of Cowskin Creek is evaluated in real time. The node depths at node N154-28 were recorded for a random group of land management scenarios and evaluated for flooding across all of the design storm applications.

3.3.2.1 Fresh Water Provision

The fresh water provision index (Equation 3.2) was calculated as a function of water quantity and water quality (Logsdon & Chaubey, 2013).
\[ FWP_{I_t} = \left( \frac{MF_t}{MF_{EF}} \right) \left( \frac{WQI_{avg,t}}{1 + \frac{e_t}{n_t}} \right) \]  \hspace{1cm} \text{(Equation 3.2)}

\[ FWP_t = (Q_t) \times FWP_{I_t} \]  \hspace{1cm} \text{(Equation 3.3)}

Where:

- FWP = Fresh Water provision (m$^3$ of water)
- \( t = \text{time step} \)
- \( Q_t = \text{total flow in time step (m}^3) \)
- \( MF = \text{mean flow (m}^3/\text{s}) \)
- \( MF_{EF} = \text{environmental flow requirement, long-term (m}^3/\text{s}) \)
- \( qne = \text{number of times flow is less than environmental flows} \)
- \( WQI_{avg} = \text{average Water Quality Index (Equation 3.3)} \)
- \( e = \text{number of times WQI is less than one} \)
- \( n = \text{number of units in time step (i.e., 365/366 if FWP is calculated for the year on a daily basis)} \)

\[ WQI = \frac{\exp(w_1 + w_2 + \cdots + w_n)}{\exp \left[ w_1 \left( \frac{C_1}{C_{1std}} \right) + w_2 \left( \frac{C_2}{C_{2std}} \right) + \cdots + w_n \left( \frac{C_n}{C_{nstd}} \right) \right]} \]  \hspace{1cm} \text{(Equation 3.4)}

Where:

- \( WQI = \text{water quality index} \)
- \( C_1, C_2, \ldots, C_n = \text{concentrations of water quality constituents of concern} \)
- \( w_1, w_2, \ldots, w_n = \text{weights for water quality constituents of concern, where} \)
- \( \Sigma(w_1 + w_2 + \cdots + w_n) = 1 \)
- \( std = \text{standard limits for water quality constituents of concern} \)
The \( \left( \frac{M_F}{M_F^{EF}} \right) + \left( \frac{q_{nE}}{n_E} \right) \) component, or water quantity term, of Equation 3.2 was designed to equal a value of one if the environmental flow conditions are met throughout the simulation period (Logsdon & Chaubey, 2013). The environmental flow condition is a minimum flow requirement (m\(^3\)/s) set by a local governing association that is necessary to maintain environmental ecosystem health in the water body. Similarly, the \( \left( \frac{WQ_{tav.,t}}{1 + \left( \frac{e_L}{n_E} \right)} \right) \) component, or water quality term of Equation 3.2 was designed to equal a value of one if water quality standards are met throughout the simulation period (Logsdon & Chaubey, 2013). If both the water quantity term and water quality term are equal to one, then Equation 3.2 will also equal one. Thus the quantity of fresh water provision (Equation 3.3) will equal the total amount of water provided, \( Q_t \), indicating excellent fresh water provisioning service (Logsdon & Chaubey, 2013). A water quantity term or water quality term of less than one will indicate that either the environmental flow requirement or water quality standard has not been met (Logsdon & Chaubey, 2013). Thus the quantity of fresh water provision found in Equation 3.3 will be less than \( Q_t \), indicating a reduction in fresh water provision services.

### 3.3.2.2 Erosion Regulation

Though erosion is a natural process, changes in land use may increase erosion rates which will jeopardize the viability of aquatic habitats in a particular area. It is important to stabilize erosion rates in order to preserve existing land, prevent habitat degradation, and maintain water quality. The erosion regulation index (Equation 3.5) compares the current erosion rate to the allowable (or natural) rate of erosion (Logsdon & Chaubey, 2013).
\[ ERI = \exp \left( 1 - \frac{E_{\text{ann}}}{E_{\text{max}}} \right) \]  
(Equation 3.5)

Where:  
\( ERI \) = erosion regulation index  
\( E_{\text{ann}} \) = annual erosion rate (T/ha)  
\( E_{\text{max}} \) = allowable/natural rate of erosion (T/ha)

The erosion regulation index operates similarly to the fresh water provision index. If the annual erosion rate equal the allowable erosion rate, then the value of ERI will equal one (Logsdon & Chaubey, 2013). The allowable/natural rate of erosion was determined for the study area using the Universal Soil Loss Equation. An ERI value of greater than one is indication of good erosion regulation services (\( E_{\text{ann}} < E_{\text{max}} \)). An ERI value of less than one is indication of poor erosion regulation services since (\( E_{\text{ann}} > E_{\text{max}} \)).

3.3.2.3 Flood Regulation

Similar to erosion, flooding is a natural ecosystem process that can be important for ecosystems (Logsdon & Chaubey, 2013). However changes in land-use associated with urbanization and intense agriculture can increasing the occurrence of flooding events, which has the potential to be damaging to both ecosystems and human well-being. The flood regulation index (Equation 3.6) was determined from three factors: the duration of flood events, the number of flood events within a time period, and the magnitude or extent of the flood event (Logsdon & Chaubey, 2013).
\[ FRI = \exp \left[ w_1 \left( \frac{DF}{D_{FLT}} \right) + w_2 \left( \frac{QF}{Q_{FLT}} \right) + w_3 \left( \frac{FE}{F_{ELT}} \right) \right] \]  

(Equation 3.6)

Where:

- FRI = flood regulation index
- DF = duration of flood events (days)
- QF = average magnitude of flooding events (m³/s)
- FE = number of flood events per year

\[ w_1, w_2, \ldots, w_n = \text{weights for water quality constituents of concern, where } \sum (w_1 + w_2 + \ldots + w_n) = 1 \]

*The LT subscript denotes a calculation using long-term (historical) data.*

If there are no flood events within the simulation period, the flood regulation index will equal one, indicating maximum flood regulation services (Logsdon & Chaubey, 2013). However the occurrence of any flood events during the simulation period will result in an FRI value of less than one, indicating a decrease in flood regulation services (Logsdon & Chaubey, 2013).

### 3.4 Land Management Scenarios

A significant component of this research was focused on identifying and quantifying the types of ecosystem services provided by urban best management practices. Since the overarching goal of this research was to improve the potential benefits and tradeoffs in water quality in the City of Wichita through the implementation of a targeted BMP program, a variety of BMP implementation scenarios were simulated to understand the impacts on water quality. Each land management scenario was run for a 30.5-mm (1.2-inch) design storm (City of Wichita Public Works & Utilities, 2010) as well as a 5-year, 10-year, 25-year, and 100-year return frequency, SCS 24-hour Type II storm event (Table 3.4). SWMM outputs were then used to calculate the
provision of ecosystem services using the indices developed by Logsdon (2011) for each scenario.

Table 3.4 Precipitation frequency estimates for Wichita, KS [NOAA 2014].

<table>
<thead>
<tr>
<th>Average Recurrence Interval [years]</th>
<th>Duration [hours]</th>
<th>Precipitation Depth [mm(in)]</th>
</tr>
</thead>
<tbody>
<tr>
<td>Design Storm</td>
<td>24</td>
<td>30.5 (1.2)</td>
</tr>
<tr>
<td>5</td>
<td>24</td>
<td>107.7 (4.24)</td>
</tr>
<tr>
<td>10</td>
<td>24</td>
<td>126.5 (4.98)</td>
</tr>
<tr>
<td>25</td>
<td>24</td>
<td>153.7 (6.05)</td>
</tr>
<tr>
<td>100</td>
<td>24</td>
<td>198.9 (7.83)</td>
</tr>
</tbody>
</table>

Fifty-six different land management scenarios (Table 3.5) with varying applications of BMP type and density were applied to the existing SWMM model to simulate water quantity and quality impacts using a baseline scenario with no BMP implementation as the control. Each BMP was designed to handle runoff from a specified percentage of impervious area. The BMPs that were simulated in this model included bioretention cells, green roofs, and rain barrels. Bioretention cells were specified in the model to capture runoff from both pavement and roof land uses, while green roofs and rain barrels were specified to only capture runoff from roof land uses. These BMPs were selected to represent the variety of hydrologic processes BMPs employ (e.g. infiltration and storage) as well as the interests of the City of Wichita.
Table 3.5 Land management scenarios with varying BMP application. The % indicates the % of impervious surface area treated within each BMP scenario.

<table>
<thead>
<tr>
<th>Scenario</th>
<th>Bioretention Cell (%)</th>
<th>Green Roof (%)</th>
<th>Rain Barrel (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>2-11</td>
<td>10-100 (10% increments)</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>12-21</td>
<td>0</td>
<td>2-20 (2% increments)</td>
<td>0</td>
</tr>
<tr>
<td>22-31</td>
<td>0</td>
<td>0</td>
<td>2-20 (10% increments)</td>
</tr>
<tr>
<td>32-34</td>
<td>0</td>
<td>4-12 (4% increments)</td>
<td>2</td>
</tr>
<tr>
<td>35-37</td>
<td>0</td>
<td>4-12 (4% increments)</td>
<td>4</td>
</tr>
<tr>
<td>38-40</td>
<td>0</td>
<td>4-12 (4% increments)</td>
<td>6</td>
</tr>
<tr>
<td>41-42</td>
<td>20-40 (20% increments)</td>
<td>4</td>
<td>0</td>
</tr>
<tr>
<td>43-44</td>
<td>20-40 (20% increments)</td>
<td>8</td>
<td>0</td>
</tr>
<tr>
<td>45-46</td>
<td>20-40 (20% increments)</td>
<td>4</td>
<td>2</td>
</tr>
<tr>
<td>47-48</td>
<td>20-40 (20% increments)</td>
<td>8</td>
<td>2</td>
</tr>
<tr>
<td>49-50</td>
<td>20-40 (20% increments)</td>
<td>4</td>
<td>4</td>
</tr>
<tr>
<td>51-52</td>
<td>20-40 (20% increments)</td>
<td>8</td>
<td>4</td>
</tr>
<tr>
<td>53-54</td>
<td>20-40 (20% increments)</td>
<td>4</td>
<td>6</td>
</tr>
<tr>
<td>55-56</td>
<td>20-40 (20% increments)</td>
<td>8</td>
<td>6</td>
</tr>
</tbody>
</table>

3.4.1 Bioretention Cell Design Specifications

A bioretention cell is a type of infiltration-based BMP that is used to treat stormwater runoff from impervious areas in the urban landscape. This water quality and quantity control practice uses natural chemical, biological, and physical properties of plants, microbes, and soils for removal of pollutants from stormwater runoff (Environmental Services Division, 2007). Bioretention cells are known to capture runoff and reduce peak flow through natural hydrologic processes such as infiltration and evapotranspiration (Ahiablame et al., 2012). The reduction of runoff volume and peak flow rate using bioretention cells has been found to range from 40-97% (Ahiablame et al., 2012). This BMP is ideal for controlling urban stormwater runoff at the
source, therefore reducing large volumes of runoff that would otherwise be managed downstream (Mid-Ameralical Regional Council and American Public Works Association, 2012).

One of the major advantages of bioretention cells is that they have been found to have one of the highest nutrient and pollutant removal efficiencies of any BMP (Mid-Ameralical Regional Council and American Public Works Association, 2012). Research on bioretention performance in North Carolina found that total nitrogen removal ranged from 40-68% and total phosphorus removal ranged from 22-68% (Hunt & Lord, 2006). The removal of total suspended solids has been reported as high as 97% (Environmental Services Division, 2007). Additionally bioretention cells have been found to reduce metal concentrations from 30-90%, positively affect bacteria retention, and increase microbial removal (Ahiablame et al., 2012).

Bioretention cells were chosen for this study to represent the function of infiltration-based BMPs in an urban area. Each bioretention cell was designed to treat stormwater runoff and associated pollutants from 4047-m² (1-acre) of impervious area (Table 3.6). It is assumed that each bioretention cell operates with an 85% TSS removal rate, 40% TP removal rate, and 50% nitrogen removal rate for a 30.5-mm (1.2-inch) design storm (Ahiablame et al., 2012; City of Wichita Public Works & Utilities, 2010). The number of bioretention cells in each design scenario was increased or decreased accordingly to treat runoff from the designated percentage of impervious area. For example, two bioretention cells with the following design specifications would be used to treat runoff from an impervious area of 8094-m² (2-acres).
3.4.2 Green Roof Design Specifications

A green roof is a type of urban BMP that is installed on a pre-existing building rooftop, using vegetation and high quality waterproof membranes to compensate for the vegetation that was removed when the building was constructed (Ahiablame et al., 2012). This type of BMP is especially suitable for urban areas where there is limited green space to implement traditional stormwater controls (Berghahe et al., 2009). Green roofs have been found to serve a multitude of purposes, including controlling runoff volume, providing building insulation, creating wildlife
habitat, and helping to combat the urban heat island effect (Mid-America Regional Council and American Public Works Association, 2012).

Green roofs are an effective means to reduce runoff quantity. Research has shown that rainfall retention by green roofs may vary from 20-100%, though this percentage functions as a result of rainfall quantity (Mid-America Regional Council and American Public Works Association, 2012). Green roofs tend to be more effective at retaining precipitation during summer months, capturing nearly 95% of rainfall, compared to winter months where retention may be less than 20% (Berghahe et al., 2009; Mid-America Regional Council and American Public Works Association, 2012). Water quality impacts of a green roof are directly related to the design, management, and vegetation, and research has reported conflicting and inconclusive results (Ahiablame et al., 2012; Berghahe et al., 2009). It is recommended that green roofs are used in conjunction with other stormwater BMPs, such as bioretention cells, for the purpose of stormwater runoff water quality treatment (Berghahe et al., 2009).

Green roofs were chosen for simulation in this study due to the large urban land-use component. Each green roof was designed to treat stormwater runoff from 202-m² (0.05-acre) of impervious roof area (Table 3.7). This green roof should theoretically retain 100% of rainfall from a 30.5-mm (1.2-inch) design storm. Due to conflicting reports for green roof pollutant removal, this research assumed a 0% removal rate for TSS, TP, and TN. The number of green roof units in each design scenario will be increased or decreased accordingly to treat runoff for a designated percentage of roof impervious area.
Table 3.7 Green roof design values for SWMM.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Value</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>LID Usage</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Unit Area (m²)</td>
<td>60.7</td>
<td></td>
</tr>
<tr>
<td>Surface width per unit (m)</td>
<td>6.096</td>
<td></td>
</tr>
<tr>
<td>% initially saturated</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>Impervious area treated (m²)</td>
<td>202.3</td>
<td></td>
</tr>
<tr>
<td><strong>Surface</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Berm height (mm)</td>
<td>152.4</td>
<td></td>
</tr>
<tr>
<td>Vegetation volume (frac.)</td>
<td>0.8</td>
<td>(Rusenieks &amp; Kamenders, 2013)</td>
</tr>
<tr>
<td>Surface roughness</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>Surface slope (%)</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td><strong>Soil</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Thickness (mm)</td>
<td>101.6</td>
<td>(Berghahe et al., 2009)</td>
</tr>
<tr>
<td>Porosity (vol. fraction)</td>
<td>0.6</td>
<td>(Rooflite Soil,)</td>
</tr>
<tr>
<td>Field capacity (vol. fraction)</td>
<td>0.55</td>
<td>(Rooflite Soil,)</td>
</tr>
<tr>
<td>Wilting point (vol. fraction)</td>
<td>0.1</td>
<td>(Rooflite Soil,)</td>
</tr>
<tr>
<td>Conductivity (m/s)</td>
<td>1.4x10⁻⁷</td>
<td>(Rusenieks &amp; Kamenders, 2013)</td>
</tr>
<tr>
<td>Conductivity slope</td>
<td>10</td>
<td>(Rusenieks &amp; Kamenders, 2013)(L. Rossman,)</td>
</tr>
<tr>
<td>Suction head (mm)</td>
<td>3.5</td>
<td>(Rusenieks &amp; Kamenders, 2013)</td>
</tr>
<tr>
<td><strong>Drainage Mat</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Thickness (mm)</td>
<td>15.24</td>
<td>(Colbond Inc., 2006)</td>
</tr>
<tr>
<td>Void fraction</td>
<td>0.95</td>
<td>(Colbond Inc., 2006)</td>
</tr>
<tr>
<td>Roughness (Manning’s N)</td>
<td>0.022</td>
<td>(Colbond Inc., 2006)</td>
</tr>
</tbody>
</table>

3.4.3 Rain Barrel Design Specifications

Rain barrels are a type of BMP that collect stormwater runoff from impervious roof surfaces. Rain barrels operate by retaining a predetermined volume of rooftop runoff that may be stored for later reuse in a variety of applications, such as in lawn and garden irrigations (Programs and Planning Division, 1999). Rain barrels have a variety of benefits, including the reduction of stormwater runoff volume and storing water for reuse applications (especially in drought) (Sands & Chapman, 2003). Though rain barrels do not provide substantial water quality treatment, they are a low-cost, effective, and easily maintainable method for water quantity management (Programs and Planning Division, 1999). Rain barrels can be extremely effective when used in conjunction with other BMP methods, such as green roofs or bioretention cells.
Rain barrels were chosen for simulation in this study due to the significant portion of urban land-use. Each rain barrel was assumed to have a capacity of 19.3-m³ (5100-gal), which is sufficient to capture 100% of stormwater runoff from 634-m² (6825-ft²) of impervious roof area in a 30.5-mm (1.2-inch) design storm (Table 3.8). A rain barrel of this magnitude is typically referred to as a cistern, since rain barrels in practice typically hold smaller volumes of water. Pollutant removal rates were 0% for TSS, TP, and TN. It was assumed that all runoff collected in the rain barrels would be applied to pervious areas in the watershed at a later date. The number of rain barrels in each design scenario was increased or decreased accordingly to treat the designated percentage of roof impervious area.

**Table 3.8 Rain barrel design values for SWMM.**

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Value</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>LID Usage</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Unit Area (m²)</td>
<td>62.6</td>
<td></td>
</tr>
<tr>
<td>Surface width per unit (m)</td>
<td>2.18</td>
<td>(Rain Harvest Systems, 2015)</td>
</tr>
<tr>
<td>% initially saturated</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>Impervious area treated (m²)</td>
<td>634</td>
<td></td>
</tr>
<tr>
<td><strong>Surface</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Barrel height (m)</td>
<td>2.7</td>
<td>(Rain Harvest Systems, 2015)</td>
</tr>
<tr>
<td><strong>Underdrain</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Drain coefficient (m/s)</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>Drain exponent</td>
<td>0</td>
<td>(Rooflite Soil, )</td>
</tr>
<tr>
<td>Drain offset height (mm)</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>Drain delay (sec)</td>
<td>0</td>
<td></td>
</tr>
</tbody>
</table>
3.5 Statistical Analysis

SWMM operates as a deterministic model, meaning that the model will always produce the same output from a given starting condition. This essentially indicates that the final output data would always be the same whether the SWMM model were to be run five times or fifty times under the same treatment scenario. As a result, there is no variability within the model. In order to statistically analyze the results obtained under this experiment, a randomized complete block design was used. This standard statistical method divides experimental units into homogenous groups, or blocks, with treatments applied to each block. The design storms used in the model simulation (1.2-inch, 5-year, 10-year, 25-year, and 100-year) were grouped together as a block and each of the individual land management scenarios were the treatments applied to each block. This design method allows for some variability among results within the block so that the data from the SWMM model could be analyzed statistically.

Statistical Analysis System, or SAS, is a software program developed by the SAS Institute for analytics and data management. The SAS University Edition was used to develop the randomized complete block design using a generalized linear model (Appendix C). Fresh water provision index data (FWPI) and erosion regulation index (ERI) data from two locations in the model were obtained for statistical analysis. The first location was at the outfall node of the entire model along the Arkansas River downstream of the City of Wichita, and the second location was at node N190, which is along Cowskin Creek just downstream from the targeted BMP implementation site. The research question that this statistical analysis aimed to answer was: Is there a significant difference in mean values of FWPI and ERI between treatments at the two monitoring locations? Unfortunately, the mathematical structure of the flood regulation index limited the data from being analyzed statistically.
Chapter 4 - Results

4.1 Model Calibration and Validation

Streamflow calibration and validation for the SWMM model was conducted at node N190 on Cowskin Creek and the outfall node on the Arkansas River. Node N190 on Cowskin Creek was calibrated using data from USGS Gage Station #07144490 and the outfall node was calibrated to the USGS Gage Station #07144550 on the Arkansas River downstream of the City of Wichita. A total of 21 parameters were calibrated for streamflow using SWMM’s Sensitivity Radio Tuned Calibration (SRTC) tool and the Nash-Sutcliffe Efficiency (NSE) (Table 4.1). Each parameter was given an uncertainty of 100% in the SRTC tool, which optimized parameter values by simultaneously varying each within the assigned uncertainty range to maximize NSE. The final calibrated value for each parameter determined using the SRTC tool was applied to the SWMM model is available in Appendix D.

Table 4.1 Calibrated model parameters.

<table>
<thead>
<tr>
<th>SWMM Layer</th>
<th>Parameter</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Subcatchments</td>
<td>Area (ac)</td>
<td>Subcatchment area in acres</td>
</tr>
<tr>
<td>Subcatchments</td>
<td>Width (ft)</td>
<td>Width of the overland flow path for sheet flow runoff</td>
</tr>
<tr>
<td>Subcatchments</td>
<td>Slope (%)</td>
<td>Average percent slope of the subcatchment</td>
</tr>
<tr>
<td>Subcatchments</td>
<td>Imperv (%)</td>
<td>Percent of land area which is impervious</td>
</tr>
<tr>
<td>Subcatchments</td>
<td>N Imperv</td>
<td>Manning’s N for overland flow for impervious area</td>
</tr>
<tr>
<td>Subcatchments</td>
<td>N Perv</td>
<td>Manning’s N for overland flow for pervious area</td>
</tr>
<tr>
<td>Subcatchments</td>
<td>Dstore Imperv (in)</td>
<td>Depth of depression storage for impervious area</td>
</tr>
<tr>
<td>Subcatchments</td>
<td>Dstore Perv (in)</td>
<td>Depth of depression storage for pervious area</td>
</tr>
<tr>
<td>Subcatchments</td>
<td>Zero Imperv (%)</td>
<td>Percent of impervious area with no depression storage</td>
</tr>
<tr>
<td>Subcatchments</td>
<td>Percent Routed (%)</td>
<td>Percent of runoff routed between subareas</td>
</tr>
<tr>
<td>Subcatchments</td>
<td>Suction head (in)</td>
<td>Value of soil capillary suction along the wetting front</td>
</tr>
<tr>
<td>Subcatchments</td>
<td>Conductivity (in/hr)</td>
<td>Soil saturated hydraulic conductivity</td>
</tr>
<tr>
<td>Subcatchments</td>
<td>Initial Deficit (frac.)</td>
<td>Diff. between soil porosity &amp; initial moisture content</td>
</tr>
<tr>
<td>Junctions</td>
<td>Invert Elev. (ft)</td>
<td>Invert elevation of the junction</td>
</tr>
</tbody>
</table>
The NSE values obtained through calibration and validation on a daily time scale are summarized in Table 4.2, while the measured and modeled hydrographs are presented in Figure 4.1, Figure 4.2, Figure 4.3, and Figure 4.4. Daily NSE values of 0.5 or greater are acceptable for model simulation (Logsdon, 2011) and therefore the following calibration and validation values were satisfactory for streamflow prediction.

<table>
<thead>
<tr>
<th>Location</th>
<th>Calibration NSE</th>
<th>Validation NSE</th>
</tr>
</thead>
<tbody>
<tr>
<td>N190</td>
<td>0.586</td>
<td>0.63</td>
</tr>
<tr>
<td>Outfall</td>
<td>0.643</td>
<td>0.61</td>
</tr>
</tbody>
</table>

Table 4.2 NSE values for streamflow calibration and validation.
Figure 4.1 Simulated and observed streamflow at node N190 for the calibration period.

Figure 4.2 Simulated and observed streamflow at node N190 for the validation period.
Figure 4.3 Simulated and observed streamflow at the outfall node for calibration period.

Figure 4.4 Simulated and observed streamflow at the outfall node for the validation period.
SWMM model predictions of total nitrogen (TN), total phosphorus (TP) and total suspended sediment (TSS) were also calibrated and validated at the Arkansas River outfall node downstream from the City of Wichita. TSS, nitrate, total kjeldahl nitrogen, and TP data were obtained for calibration from the EPA STORET database at Arkansas River at Derby Station (Station ID: SC281), which is approximately 5.2-km upstream from the model outfall node. Nitrate and total kjeldahl nitrogen were summed to obtain estimates for TN at the site.

An individual pollutant wash-off function was assigned to the four land-use categories within the land-use editor for each pollutant of interest. Event mean concentration, or EMC (mg/L), and exponential, or EXP, were the two wash-off function types used to simulate pollutant loading (Table 4.3). Each wash-off function was calibrated by comparing the simulated pollutant loads against measured pollutant loads using the Nash-Sutcliffe Efficiency. Final wash-off functions were determined for TN, TP, and TSS for each land-use category once an acceptable NSE value was obtained (Table 4.3).

Table 4.3 Pollutant wash-off characteristics for each land use.

<table>
<thead>
<tr>
<th>Land-Use Category</th>
<th>Pollutant</th>
<th>Wash-Off Function</th>
<th>Coefficient</th>
<th>Exponent</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mixed Development</td>
<td>TN</td>
<td>EMC (mg/L)</td>
<td>5</td>
<td>-</td>
</tr>
<tr>
<td>Pavement</td>
<td>TN</td>
<td>EMC (mg/L)</td>
<td>5.2</td>
<td>-</td>
</tr>
<tr>
<td>Roofs</td>
<td>TN</td>
<td>EMC (mg/L)</td>
<td>5.2</td>
<td>-</td>
</tr>
<tr>
<td>Undeveloped</td>
<td>TN</td>
<td>EMC (mg/L)</td>
<td>4.8</td>
<td>-</td>
</tr>
<tr>
<td>Mixed Development</td>
<td>TP</td>
<td>EMC (mg/L)</td>
<td>0.8</td>
<td>-</td>
</tr>
<tr>
<td>Pavement</td>
<td>TP</td>
<td>EMC (mg/L)</td>
<td>0.88</td>
<td>-</td>
</tr>
<tr>
<td>Roofs</td>
<td>TP</td>
<td>EMC (mg/L)</td>
<td>0.88</td>
<td>-</td>
</tr>
<tr>
<td>Undeveloped</td>
<td>TP</td>
<td>EMC (mg/L)</td>
<td>0.5</td>
<td>-</td>
</tr>
<tr>
<td>Mixed Development</td>
<td>TSS</td>
<td>EXP</td>
<td>20</td>
<td>1.8</td>
</tr>
<tr>
<td>Pavement</td>
<td>TSS</td>
<td>EXP</td>
<td>40</td>
<td>2.2</td>
</tr>
<tr>
<td>Roofs</td>
<td>TSS</td>
<td>EXP</td>
<td>40</td>
<td>2.2</td>
</tr>
<tr>
<td>Undeveloped</td>
<td>TSS</td>
<td>EXP</td>
<td>10</td>
<td>1.2</td>
</tr>
</tbody>
</table>
The pollutant loading NSE values obtained through calibration and validation on a daily time scale are summarized in Table 4.4, while the measured and modeled pollutographs are presented in Figure 4.5, Figure 4.6, Figure 4.7, Figure 4.8, Figure 4.9, and Figure 4.10. Daily NSE values of 0.5 or greater are acceptable for model simulation (Logsdon, 2011) and therefore the following calibration and validation values were satisfactory for pollutant loading prediction.

Table 4.4 NSE values for pollutant calibration and validation.

<table>
<thead>
<tr>
<th>Pollutant</th>
<th>Calibration NSE</th>
<th>Validation NSE</th>
</tr>
</thead>
<tbody>
<tr>
<td>TSS</td>
<td>0.72</td>
<td>0.82</td>
</tr>
<tr>
<td>TN</td>
<td>0.65</td>
<td>0.93</td>
</tr>
<tr>
<td>TP</td>
<td>0.59</td>
<td>0.92</td>
</tr>
</tbody>
</table>

Figure 4.5 Simulated and observed TSS for the calibration period.
Figure 4.6 Simulated and observed TSS for the validation period.

Figure 4.7 Simulated and observed TN for the calibration period.
Figure 4.8 Simulated and observed TN for the validation period.

Figure 4.9 Simulated and observed TP for the calibration period.
4.2 Results and Statistical Analysis

4.2.1 Simulation Results

Each land management scenario was compared against the baseline control scenario (i.e. no stormwater BMPs) to assess quantitatively the impact of targeted BMP implementation on ecosystem service provision throughout the study area. The fresh water provision index (FWPI) and erosion regulation index (ERI) were calculated at two locations in the model. The first site was located at the outfall node of the entire model along the Arkansas River, just downstream of the City of Wichita (Figure 4.11). The second site was located at node N190, which is along Cowskin Creek downstream from the targeted BMP implementation area (Figure 4.11). The flood regulation index (FRI) was calculated for a random group of land management scenarios.
across all design storm applications at node N154-28 (Figure 4.11). Model node N154-28 is collocated with USGS site #07144480 where flooding on Cowskin Creek is evaluated in real time.

![Figure 4.11 Location of ecosystem service data collection.](image)

4.2.1.1 FWPI Provision at the Outfall Node

Those scenarios containing only bioretention cells produced the greatest FWPI values at the outfall node for the 30.5-mm (1.2-inch) design storm. The smallest values of FWPI at the outfall node for this group of land management scenarios resulted from the 100-year design storm (Table B.1). This trend indicates that system performance declines with an increase in design storm magnitude (Figure 4.12), which is understandable given the bioretention cell in this model was designed to effectively treat 4047-m$^2$ (1-acre) of runoff during a 30.5-mm (1.2-inch) design storm. It is likely that the system is overwhelmed by the larger volumes of runoff.
associated with more significant design storms and thus the fresh water provision at the outfall is diminished. FWPI values at the outfall node from the various stages of bioretention cell implementation were compared against the baseline land management scenario to quantify the change in ecosystem service provision. Though values of FWPI increased from the baseline scenario with bioretention cell application, none of the values of FWPI ever exceeded one. A value of one is a numerical indication that water quality standards have been met and that there is excellent freshwater provision service. The greatest percent increase in FWPI at the outfall among this group was observed between the baseline land management scenario and scenario 2 (10% bioretention cell implementation) for a 30.5-mm (1.2-inch) design storm (ΔFWPI=7.32%) (Table B.2).

![Changes in FWPI for Bioretention Cell Application @ Outfall Node](image)

**Figure 4.12 Changes in FWPI at the outfall node with bioretention cell application.**

There was not a significant difference in FWPI values between the baseline land management scenario and all green roof application, all rain barrel application, and all combination green roof and rain barrel application at the outfall node (Figure 4.13; Figure 4.14;
Figure 4.15. It is not surprising that there was no major change in values of FWPI since neither the green roofs nor the rain barrels in this model were assumed to provide water quality treatment in the model. Similar to the bioretention cell scenarios, all green roof, rain barrel, and green roof/rain barrel combination scenarios demonstrated greater FWPI performance at the outfall node during the 30.5-mm (1.2-inch) storm than during larger design storms (Table B.1). Again this is likely because the green roofs and rain barrels were designed to handle runoff from a 30.5-mm (1.2-inch) design storm and were unable to process the larger volumes of runoff associated with larger design storms. To quantify the impact of BMP implementation on ecosystem service provision, the all green roof application, all rain barrel application, and all combination green roof and rain barrel application were compared against the baseline land management scenario at the model outfall. In general, the percentage change in FWPI from the baseline land management scenario to this group of scenarios was zero (Table B.2).

Figure 4.13 Changes in FWPI at the outfall node with green roof application.
The combination bioretention cell and green roof scenarios performed similarly to the scenarios in which only bioretention cell were modeled. The combination bioretention cell and green roof scenarios demonstrated the greatest values of FWPI at the outfall node during the

Figure 4.14 Changes in FWPI at the outfall node with rain barrel application.

Figure 4.15 Changes in FWPI at the outfall node with combination green roof/rain barrel application.
30.5-mm (1.2-inch) design storm, with values of FWPI decreasing with increasing storm size (Figure 4.16). As stated previously, the bioretention cell in this model was designed to successfully treat 4047-m² (1-acre) of runoff during a 30.5-mm (1.2-inch) design storm, so it understandable that system performance would decrease as design storm size increased. FWPI values at the outfall node produced by various intensities of combined bioretention cell/green roof application were compared against the baseline land management scenario to quantify the change in fresh water provision. Though values of FWPI increased from the baseline with these combination bioretention cell/green roof application scenarios, none of the values were ever recorded to be FWPI ≥ 1, which is an indication of excellent freshwater provision service. The greatest percent increase in FWPI at the outfall among this group was observed comparing the baseline land management scenario to scenario 42 (40% bioretention cell, 4% green roof) with ΔFWPI=7.2% during a 30.5-mm (1.2-inch) design storm (Table B.2).

![Changes in FWPI for Bioretention Cell/Green Roof Application @ Outfall Node](image)

**Figure 4.16 Changes in FWPI at the outfall node for combination bioretention cell/green roof application.**

The combination bioretention cell, green roof, and rain barrel application scenarios performed very similarly as all other scenarios containing a bioretention cell. These combination
scenarios demonstrated the greatest values of FWPI at the outfall node during the 30.5-mm (1.2-inch) design storm, with values of FWPI decreasing with increasing storm size (Figure 4.17; Table B.1). It is likely that the system was overwhelmed with the larger volumes of runoff associated with larger design storms since the bioretention cell in this model was only designed to effectively treat 4047-m² (1-acre) of runoff during a 30.5-mm (1.2-inch) design storm. Though values of FWPI increased from the baseline scenario with the combination bioretention cell/green roof/rain barrel application scenarios, none of the values ever exceeded one, which is an indication of excellent freshwater provision service. The percent change in values of FWPI at the outfall node from the various stages of combination bioretention cell/green roof/rain barrel implementation were compared against the baseline land management scenario to quantify improvement in fresh water provision. The greatest percent increase in values of FWPI occurred between the baseline land management scenario and scenario 46 (40% bioretention, 4% green roof, 2% rain barrel) with ΔFWPI=7.2% during a 30.5-mm (1.2-inch) design storm (Table B.2).

![Changes in FWPI for Bioretention Cell/Green Roof/Rain Barrel Application @ Outfall Node](image)

**Figure 4.17** Changes in FWPI at the outfall node for combination bioretention cell/green roof/rain barrel application.
The largest percent change in FWPI at the outfall node among all of the land management scenarios occurred between the baseline scenario and scenario 2 (10% bioretention application) with $\Delta \text{FWPI}=7.3\%$ during a 30.5-mm (1.2-inch) design storm (Table B.2). This result indicates that land management scenarios with 10% runoff capture by bioretention cells will provide optimal fresh water provision at the outfall node across all sizes of design storms. However it is important to note that FWPI values did not meet or exceed one under any scenario. To achieve a FWPI value of one, the requirements in both the water quantity term and the water quality term of the index must be satisfied. This means that the minimum flow requirement for the water body of interest must be met for the water quantity term and that the simulated/measured pollutant concentrations must be below the acceptable concentration limit for the water quality term. A FWPI value of one indicates excellent freshwater provision services, and this standard was not achieved for any land management scenario at the outfall node.

4.2.1.2 ERI Provision at the Outfall Node

Those land management scenarios in which only bioretention cells were implemented were compared to the baseline land management scenario to assess the impact of BMP implementation on erosion regulation at the outfall node. As a general trend, the percent change in values of ERI increased as the percentage of runoff capture by the bioretention cells increased (Table B.4). The highest percent change in values of ERI was observed between the baseline land management scenario and scenario 11 (100% bioretention cell) with $\Delta \text{ERI}=683.7\%$ during a 30.5-mm (1.2-inch) design storm (Table B.4). The daily erosion rate for this scenario was calculated as 2111 kg/km$^2$/d (18.8 lbs/ac/d) during the 30.5-mm (1.2-inch) design storm, and the maximum allowable erosion rate was 6568 kg/km$^2$/d (58.6 lbs/ac/d). However, the percent
change in ERI values between the baseline and the 10% bioretention cell scenario
($\Delta$ERI=682.1\%) during a 30.5-mm (1.2-inch) design storm (Table B.4) was nearly the same. For comparison, the daily erosion rate for this scenario was 2130 kg/km$^2$/d (19 lbs/ac/d).

Additionally the percent change in values of ERI rose as the size of the design storm increased (Table B.4). ERI values at the outfall node exceeded 1 for all bioretention cell scenarios during the 30.5-mm (1.2-inch) design storm (Table B.3), which is an indication of excellent erosion regulation services gained by implementation of bioretention cells. There was a general decline in values of ERI as the storm size increased (Figure 4.18), which is understandable since each bioretention cell was designed to only effectively treat 4047-m$^2$ (1-acre) of runoff during a 30.5-mm (1.2-inch) design storm.

Those land management scenarios containing only green roof, only rain barrel, or combination green roof/rain barrel BMPs were compared against the baseline land management scenario to evaluate the impact of BMP application on erosion regulation at the outfall node. The three land management types performed similarly with ERI values decreasing as the magnitude
of the design storm increased (Figure 4.19; Figure 4.20; Figure 4.21). On average, the percent change in ERI values between the baseline and all green roof and/or rain barrel scenarios was negligible (Table B.4). The percent change in values of ERI among green roof scenarios was the largest between the baseline scenario and scenario 21 (20% green roof) with ΔERI=0.21% (Table B.4) during a 30.5-mm (1.2-inch) design storm. The daily erosion rate for scenario 21 was 15580 kg/km²/d (139 lbs/ac/d), which is significantly higher than that maximum allowable erosion rate of 6568 kg/km²/d (58.6 lbs/ac/d). The percent change in values of ERI among green roof scenarios decreased as the size of the design storms increased, becoming negative among larger design storm simulations. The largest percent change in ERI among rain barrel scenarios was observed between the baseline land management scenario and scenario 31 (20% rain barrel) with ΔERI=0.40% (Table B.4) during a 30.5-mm (1.2-inch) design storm. The daily erosion rate for scenario 31 was 15580 kg/km²/d (139 lbs/ac/d), which is significantly higher than that maximum allowable erosion rate of 6568 kg/km²/d (58.6 lbs/ac/d). The percent change in ERI values increased during the rain barrel scenarios as the design storm magnitude increased, which is opposite to the performance trend of the green roof scenarios. Among the combination green roof and rain barrel application scenarios, the largest percent change in values of ERI was observed between the baseline scenario and scenario 40 (12% green roof application, 6% rain barrel application) with ΔERI=0.23% (Table B.4) during a 30.5-mm (1.2-inch) design storm. The daily erosion rate for scenario 21 was 15580 kg/km²/d (139 lbs/ac/d), which is significantly higher than that maximum allowable erosion rate of 6568 kg/km²/d (58.6 lbs/ac/d). The percent change in values of ERI tended to increase during the combination green roof and rain barrel scenarios as the size of the design storm increased. Those scenarios containing only rain barrel implementation had the highest percent changes in values of ERI when compared to the baseline,
though the performance was only slightly better than the green roof implementation scenarios and the combination green roof/rain barrel implementation scenarios. The negative percent change values calculated for the some of the green roof implementation scenarios may indicate that the addition of green roofs to the system may actually inhibit ecosystem service provision in regards to erosion regulation. None of the scenarios resulted in an ERI value greater than one, which indicates that there was not beneficial erosion regulation occurring in any of these land management scenarios.

Figure 4.19 Changes in ERI at the outfall node with green roof application.
Figure 4.20 Changes in ERI at the outfall node with rain barrel application.

Figure 4.21 Changes in ERI at the outfall node with combination green roof/rain barrel application.

The combination bioretention cell and green roof application scenarios produced similar results to the all bioretention cell implementation scenarios, with ERI values at the outfall node decreasing as the size of the design storm increased (Figure 4.22). The highest percent change in combination bioretention cell and green roof application when compared with the baseline was
observed for scenario 42 (40% bioretention application and 4% green roof application). The percent change in ERI during a 30.5-mm (1.2-inch) design storm for scenario 42 was $\Delta ERI = 681.2\%$ (Table B.4). The daily rate of erosion for scenario 42 was 2130 kg/km$^2$/d (19 lbs/ac/d), which is less than the maximum allowable erosion rate. Values of $ERI \geq 1$ were calculated for all combination bioretention cell and green roof application scenarios during the 30.5-mm (1.2-inch) design storm (Table B.3), indicating the occurrence of positive erosion regulation services. As the magnitude of the design storm increased, however, ERI values fell below one, indicating that the larger design storm size potentially overwhelmed the ability of the bioretention cell/green roof system to retain pollutants from stormwater runoff.

The combination bioretention cell, green roof, and rain barrel land management scenarios performed similarly to all other scenarios containing a bioretention cell (Table B.3). In general, ERI values declined as the size of the design storm increased (Figure 4.23). The percent change in ERI values tended to increase as the size of the design storm increased, which is likely
because the baseline ERI value from the larger design storms was quite small. The largest percent change in values of ERI was observed between the baseline scenario and scenario 46 (40% bioretention cell application, 4% green roof application, and 2% rain barrel application). The ΔERI=680.5% for a 30.5-mm (1.2-inch) design storm (Table B.4). The daily erosion rate for scenario 46 was 2130 kg/km²/d (19 lbs/ac/d), which is less than the maximum allowable erosion rate.

![Changes in ERI for Bioretention Cell/Green Roof/Rain Barrel Application @ Outfall Node](image)

**Figure 4.23 Changes in ERI at the outfall node with combination bioretention cell/green roof/rain barrel application.**

The largest percent change in ERI values at the outfall among all of the land management scenarios occurred comparing the baseline scenario to scenario 11 (100% bioretention application) with ΔERI=683.65% during a 30.5-mm (1.2-inch) design storm (Table B.4). However, the percentage change in ERI between the baseline and 10% bioretention application (scenario 2) was approximately the same (682.1%) for the 30.5-mm (1.2-inch) design storm (Table B.4). ERI values for land management scenarios in which only bioretention cells were implemented during the 30.5-mm (1.2-inch) design storm exceed a value of one (Figure 4.18),
indicating excellent erosion regulation services during this storm event. Other scenarios containing bioretention cells (combined with green roofs and rain barrels) performed similarly to the only bioretention cell implementation scenarios with values of ERI above one during the smallest design storm. However, the greatest increases in ERI relative to the baseline scenario were observed in combination scenarios with the highest application of bioretention cells and lowest application of green roof and/or rain barrel. This may indicate that the addition of the green roof and/or rain barrel best management practice to the system may actually detract from the ecosystem service provision being provided by the bioretention cell.

4.2.1.3 FWPI Provision at Node N190 on Cowskin Creek

Percent change in the FWPI values were compared between land management scenarios representing increasing intensity of bioretention cell application and the baseline land management scenario in order to assess the effect of targeted BMP implementation on ecosystem service provision at node N190. Values of FWPI tended to increase as the percent of runoff capture in the bioretention cell scenarios increased (Figure 4.24). The increase in FWPI around 90%-100% impervious runoff capture is likely caused by the impervious runoff routing mechanism in SWMM. Runoff is routed from the impervious area to either pervious areas or BMPs, and it is likely that there was a greater area of BMPs than impervious area in these higher BMP scenarios. Thus, the system was able to effectively treat all impervious runoff, sending none to pervious areas or waterways, caused inflated FWPI values. However, the scenarios with higher percentage of BMP implementation are somewhat unrealistic, and it is unlikely that 90 or 100% bioretention cell application would actually be implemented. Relative to the baseline scenario, the percent change in FWPI increased 37.25% under scenario 2 (10% bioretention cell application) for the 30.5-mm (1.2-inch) design storm (Table B.6). For comparison, the largest
percent change in FWPI was observed between the baseline and scenario 11 (100% bioretention cell application) with ΔFWPI = 249.95% in the 30.5-mm (1.2-inch) design storm (Table B.6). This demonstrates that the percent change in values of FWPI increased as the volume of runoff capture by the bioretention cells increased (Table B.6). These two trends in values of FWPI and percent change of FWPI indicate that freshwater provisioning services will increase with the number of bioretention cells in a system. Values of FWPI were greater than or equal to one, which is an indication of excellent freshwater provisioning services, for the 100% bioretention cell application scenario during the 30.5-mm (1.2-inch) design storm (Table B.5). FWPI values declined as the magnitude of the design storm increased, which indicates that the larger design storms overwhelmed the ability of the system to treat stormwater runoff.

![Changes in FWPI for Bioretention Cell Application @ Node N190](image)

**Figure 4.24 Changes in FWPI at Node N190 with bioretention cell application.**

Percent changes in FWPI values between land management scenarios with varying intensities of green roof application and the baseline were compared to understand the impact of green roof implementation on ecosystem service provision. FWPI values increased as the percent of runoff capture by green roof implementation increased (Figure 4.25; Table B.5). However,
values of FWPI declined as the magnitude of the design storm increased, indicating that green roof performance deteriorated with larger volumes of runoff. The largest percent change in FWPI was observed between the baseline land management scenario and scenario 21 (20% green roof application) with ΔFWPI=21.8% during a 30.5-mm (1.2-inch) design storm (Table B.6).

![Changes in FWPI for Green Roof Application @ Node N190](image)

**Figure 4.25** Changes in FWPI at Node N190 with green roof application.

Land management scenarios with varying intensities of rain barrel application were compared against the baseline land management scenario to assess the impact of rain barrel best management practices on ecosystem service provision. The rain barrel application scenarios performed very similarly to the green roof application scenarios with FWPI values increasing as the volume of runoff capture increased (Figure 4.26). This trend is an indication that greater freshwater provisioning services are associated with a higher density of rain barrel application. The largest percent change in FWPI among this group was observed between the baseline scenario and scenario 31 (20% rain barrel application) with ΔFWPI=21.57% for a 30.5-mm (1.2-inch) design storm (Table B.6). FWPI values and the percent change in FWPI generally declined
as the size of the design storm increased, indicating that the system may have been overwhelmed with the volume of runoff generated during larger storms (Table B.5; Table B.6).

The combination green roof and rain barrel land management scenarios were compared against the baseline scenario to assess the impact of BMP implementation on ecosystem service provision. Generally, FWPI values increased as the volume of runoff capture by BMPs in this group of scenarios increased (Table B.5; Figure 4.27), indicating that greater freshwater provision is associated with higher densities of BMP application. The largest percent change in FWPI (ΔFWPI=17.52% for a 30.5-mm (1.2-inch) design storm) occurred between the baseline land management scenario and scenario 40 (12% green roof, 6% rain barrel application) which is the highest application intensity in the green roof/rain barrel combination test group (Table B.6). This result is very similar to the results obtained for the all green roof and all rain barrel land management scenarios. The percent change in FWPI values for this group of land management scenarios generally declined as the size of the design storm increased, which may mean that
system performance in terms of ecosystem service provision declined as the volume of runoff increased.

![Figure 4.27 Changes in FWPI at Node N190 with combination green roof/rain barrel application.](image)

The baseline land management scenario was compared against the combination bioretention cell and green roof land management scenarios to assess the impact of BMP implementation on freshwater provision in the study area. FWPI values generally improved as the intensity of BMP implementation increased (Table B.5; Figure 4.28). This pattern indicates that a greater density of FWPI implementation yields greater freshwater provision at node N190. The largest percent change in values of FWPI were observed between the baseline land management scenario and scenario 44, with ΔFWPI = 78.03% for a 30.5-mm (1.2-inch) design storm (Table B.6). The large value of percent change in FWPI may be because scenario 44 had the largest volume of runoff capture among this group of land management scenarios, with 40% of runoff capture by bioretention cells and 8% of runoff capture by green roofs. Another trend to note is the general decline in FWPI provision among all scenarios as the size of the design storm.
increased, indicating that the system may be overwhelmed by larger volumes of runoff produced by larger design storms.

The combination bioretention cell, green roof, and rain barrel land management scenarios were compared against the baseline land management scenario to assess the impact of this group of BMPs on freshwater provision in the study area. This group of land management scenarios performed similarly to other land management scenarios containing bioretention cells, with values of FWPI increasing as the volume of runoff capture by the system increased (Table B.5; Figure 4.29). This trend indicates that there is a general increase in freshwater provisioning services as BMP implementation intensity increases. The largest percent change in FWPI was observed between the baseline land management scenario and scenario 56, with ΔFWPI=85.08% for the 30.5-mm (1.2-inch) design storm (Table B.6). The large value of percent change in FWPI may be because scenario 56 had the largest volume of runoff capture among this group of land management scenarios, with 40% of runoff capture by bioretention cells, 8% of runoff capture by

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**Figure 4.28** Changes in FWPI at Node N190 with combination bioretention cell/green roof application.

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The combination bioretention cell, green roof, and rain barrel land management scenarios were compared against the baseline land management scenario to assess the impact of this group of BMPs on freshwater provision in the study area. This group of land management scenarios performed similarly to other land management scenarios containing bioretention cells, with values of FWPI increasing as the volume of runoff capture by the system increased (Table B.5; Figure 4.29). This trend indicates that there is a general increase in freshwater provisioning services as BMP implementation intensity increases. The largest percent change in FWPI was observed between the baseline land management scenario and scenario 56, with ΔFWPI=85.08% for the 30.5-mm (1.2-inch) design storm (Table B.6). The large value of percent change in FWPI may be because scenario 56 had the largest volume of runoff capture among this group of land management scenarios, with 40% of runoff capture by bioretention cells, 8% of runoff capture by
green roofs, and 6% of runoff capture by rain barrels. Another trend to note is the general decline in FWPI as the magnitude of the design storm increased for all mixed land management scenarios (Table B.6). This pattern indicates that the system is overwhelmed by the larger volumes of runoff produced storms exceed the 30.5-mm (1.2-inch) design storm, and therefore is unable to effectively treat runoff to the same level.

![Changes in FWPI for Bioretention Cell/Green Roof/Rain Barrel Application @ Node N190](image)

**Figure 4.29 Changes in FWPI at Node N190 with combination bioretention cell/green roof/rain barrel application.**

4.2.1.4 ERI Provision at Node N190 on Cowskin Creek

The baseline land management scenario was compared to the land management scenarios with only bioretention cell application to understand the impact of targeted BMP implementation on erosion regulation services. As a general trend, ERI values increased as the volume of runoff capture by the system increased (Figure 4.30). The largest percent change in ERI (ΔERI=23.01%) was observed between the baseline scenario and scenario 11 for the 30.5-mm (1.2-inch) design storm (Table B.8). The daily rate of erosion for scenario 11 during the smallest design storm was 95 kg/km²/d (0.85 lbs/ac/d), which is significantly less than the maximum
allowable rate of erosion at 6569 kg/km²/d (58.6 lbs/ac/d). The large value of percent change in ERI may be because scenario 11 had the largest volume of runoff capture among this group of land management scenarios, with 100% of runoff capture by bioretention cells. This indicates that higher erosion regulation may occur with greater bioretention cell implementation. A decline in the percent change of ERI was observed as the size of the design storm increased. This pattern indicates that bioretention cells are more effective at treating runoff from smaller storms, and may be overwhelmed with processing the larger volumes of runoff associated with more significant design storms.

![Figure 4.30 Changes in ERI at Node N190 with bioretention cell application.](image)

Land management scenarios with green roof implementation were compared to the baseline land management scenario to assess the impact of green roof implementation on erosion regulation services. Generally ERI values and the percent change in ERI compared to the baseline increased slightly as the volume of runoff capture by the green roof system increased (Figure 4.31). The largest percent change in ERI (ΔERI=3.24%) was observed between scenario 21 and the baseline scenario for the 30.5-mm (1.2-inch) design storm (Table B.7; Table B.8).
The daily rate of erosion for scenario 21 was 1177 kg/km²/d (10.5 lbs/ac/d), which is less than the maximum allowable rate of erosion. The large value of percent change in ERI may be because scenario 21 had the largest volume of runoff capture among this group of land management scenarios, with 20% of runoff capture by green roofs. Though the percent change in ERI is quite small, this result still indicates that there may be an association between the slight increase in percent change of ERI and a higher green roof density.

![Changes in ERI for Green Roof Application @ Node N190](image)

**Figure 4.31 Changes in ERI at Node N190 with green roof application.**

The rain barrel land management scenarios were compared to the baseline land management scenario to understand the effect of rain barrels on erosion regulation services. This group of land management scenarios produced similar results as the green roof land management scenarios with ERI values and the percent change of ERI increasing as the volume of runoff capture in each scenario increased (Figure 4.32). Comparison between the baseline land management scenario and scenario 31 resulted in the largest percent change of ERI among the group, with ΔERI=4.42% during a 30.5-mm (1.2-inch) design storm (Table B.8). The daily rate of erosion for scenario 31 was 1177 kg/km²/d (10.5 lbs/ac/d), which is less than the maximum
allowable rate of erosion. This result is likely because scenario 31 had the highest rain barrel implementation, with 20% of runoff capture occurring. This indicates that the slight increase in ERI may be due to the higher implementation density of rain barrels.

![Changes in ERI for Rain Barrel Application @ Node N190](image)

**Figure 4.32 Changes in ERI at Node N190 with rain barrel application.**

The land management scenarios with combination green roof and rain barrel implementation were compared against the baseline land management scenario to assess the impact of these two BMPs together on erosion regulation services. This group of scenarios exhibited similar behavior as the all green roof and all rain barrel land management scenarios, with ERI values and percent change in ERI increasing as the volume of runoff capture increased (Figure 4.33). Scenario 40 (12% green roof capture, 6% rain barrel capture) had the largest percent change in ERI when compared to the baseline land management scenario with ΔERI=3.24% during a 30.5-mm (1.2-inch) design storm (Table B.7; Table B.8). The daily rate of erosion for scenario 40 was 1177 kg/km²/d (10.5 lbs/ac/d), which is less than the maximum allowable rate of erosion. Scenario 40 had the highest percentage of runoff capture among this
group of scenarios, indicating that the slight increase in ERI may be associated with higher densities of BMP application.

![Figure 4.33 Changes in ERI at Node N190 with combination green roof/rain barrel application.](image)

The combination bioretention cell and green roof land management scenarios were compared against the baseline land management scenario to assess the impact of these two BMPs on erosion regulation services. There was a slight increase in values of ERI and in the percent change of ERI as the volume of runoff capture by the system increased (Figure 4.34). Scenario 44 (40% bioretention cell capture, 8% green roof capture) exhibited the largest percent change in ERI when compared with the baseline scenario, with ΔERI=21.70% for a 30.5-mm (1.2-inch) design storm (Table B.7; Table B.8). The daily rate of erosion for scenario 44 during the smallest design storm was 95 kg/km²/d (0.85 lbs/ac/d), which is significantly less than the maximum allowable rate of erosion at 6569 kg/km²/d (58.6 lbs/ac/d). Similar to previous results, this pattern indicates that higher erosion regulation provision may be associated with increased application of BMP implementation. The percent change of ERI decreased as the size of the
design storm increased, which may be an indication of system decline in the ability to process and treat larger volumes of runoff. This is understandable, however, since the bioretention cell and green roof for this model were designed to effectively treat runoff from 4047-m² (1-acre) of impervious area during a 30.5-mm (1.2-inch) design storm.

Figure 4.34 Changes in ERI at Node N190 with combination bioretention cell/green roof application.

The baseline land management scenario was compared against the combination bioretention cell, green roof, and rain barrel land management scenario to understand the impact of these BMPs on erosion regulation services. Similar to other land management scenarios containing bioretention cells, this group exhibited a general increase in ERI values and in the percent change of ERI as the volume of runoff capture by the system increased (Figure 4.35). Scenario 56 demonstrated the highest percent increase of ERI when compared to the baseline scenario with ΔERI=21.91% during a 30.5-mm (1.2-inch) design storm (Table B.7; Table B.8). Scenario 56 had the highest application of BMPs among this group of scenarios, with 40% bioretention cell implementation, 8% green roof implementation, and 6% rain barrel
implementation. This result indicates that a general increase in erosion regulation services may be associated with higher densities of BMP application. Another trend observed among this group was that the percent change in ERI declined from 30.5-mm (1.2-inch) design storm to the 100-year design storm. This result may indicate that the system was overwhelmed by larger volumes of runoff associated with more significant design storms, and therefore the BMPs in the system were unable to effectively treat runoff to the same level of efficiency.

![Changes in ERI for Bioretention Cell/Green Roof/Rain Barrel Application @ Node N190](image)

**Figure 4.35 Changes in ERI at Node N190 with combination bioretention cell/green roof/rain barrel application.**

### 4.2.1.5 FRI Provision at Node N154-28 on Cowskin Creek

The flood regulation index (FRI) for group of land management scenarios selected at random (scenarios 1, 2, 5, 12, 15, 22, 25, 33, 42, and 47) was calculated and compared against the baseline land management scenario to assess the impact of targeted BMP implementation on FRI. The magnitude, duration, and occurrence of flood events for each land management scenario across the five design storms was recorded and used to calculate the FRI value. A general increase in values of FRI was observed as the volume of runoff capture increased among...
the land management scenarios; however, scenarios containing bioretention cells produced higher values of FRI than those without (Figure 4.36). The largest percent change in values of FRI was observed between the baseline land management scenario and scenario 5 (40% bioretention cell application) with ΔFRI=0.458% (Table B.9). However, none of the values of FRI were observed to be greater than or equal to one, which would be an indication of excellent flood regulation services.

![Figure 4.36 Changes in FRI with varying scenarios of land management.](image)

### 4.2.2 Statistical Analysis

Fresh water provision index data (FWPI) and erosion regulation index (ERI) data from two locations in the model were obtained for statistical analysis. The first location was at the outfall node of the entire model along the Arkansas River downstream of the City of Wichita, and the second location was at node N190, which is along Cowskin Creek just downstream from the targeted BMP implementation site. FWPI and ERI data from this experiment was analyzed
using a generalized linear model with randomized complete block design. This method divided the experimental units in homogenous groups, or blocks, with treatments applied to each block. The design storms used in the model simulation (1.2-inch, 5-year, 10-year, 25-year, and 100-year) were grouped together as a block and each of the individual land management scenarios were the treatments applied to each block. The research question that this statistical analysis aimed to answer was: Is there a significant difference in mean values of FWPI and ERI between treatments at the two monitoring locations?

### 4.2.2.1 FWPI Analysis at the Outfall Node

The FWPI data at the outfall node was adjusted in the generalized linear model with a beta distribution and evaluated using the Tukey-Kramer method for all pairwise comparisons. Each pairwise comparison was analyzed at a Type I error rate of 5% with the null hypothesis, $H_0: \mu_1=\mu_2=...=\mu_{56}$, and the alternative hypothesis, $H_1: \mu_1\neq\mu_2\neq...\neq\mu_{56}$. The statistical analysis failed to reject the null hypothesis in favor of the alternative hypothesis based on a Type I error rate of 5%, and therefore there was not sufficient evidence to conclude that a significant difference exists among mean values of FWPI at the outfall node between land management scenarios (Figure 4.37).
4.2.2.2 ERI Analysis at the Outfall Node

The ERI data at the outfall node was adjusted in the generalized linear model with a gamma distribution and evaluated using the Tukey-Kramer method for all pairwise comparisons. Each pairwise comparison was analyzed at a Type I error rate of 5% with the null hypothesis, $H_0: \mu_1=\mu_2=...=\mu_{56}$, and the alternative hypothesis, $H_1: \mu_1\neq\mu_2\neq...\neq\mu_{56}$. The following results were obtained from this analysis (Figure 4.38):

- The statistical analysis failed to reject the null hypothesis in favor of the alternative hypothesis based on a Type I error rate of 5% when comparing scenarios 1, 12, 13, 14, 15, 16, 17, 18, 19, 20, 21, 22, 23, 24, 25, 26, 27, 28, 29, 30, 31, 32, 33, 34, 35, 36, 37, 38, 39, and 40 (hereby referred to as Group 1). Therefore there was not sufficient evidence to suggest that mean values of ERI at the outfall node are significantly different between land management scenarios.
• The statistical analysis failed to reject the null hypothesis in favor of the alternative hypothesis based on a Type I error rate of 5% when comparing scenarios 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 41, 42, 43, 44, 45, 46, 47, 48, 49, 50, 51, 52, 53, 54, 55, and 56 (hereby referred to as Group 2). Therefore there was not sufficient evidence to suggest that mean values of ERI at the outfall node are significantly different between land management scenarios.

• The statistical analysis rejected the null hypothesis in favor of the alternative hypothesis based on a Type I error rate of 5% when comparing the scenarios between those in Group 1 and those in Group 2. There was, therefore, sufficient evidence to suggest that mean values of ERI at the outfall node are significantly different between land management scenarios in these two groups.

These findings indicate that there was not a significant difference among the mean values of ERI at the outfall node between the baseline condition, all green roof application, all rain barrel application, and all combination green roof/rain barrel application scenarios. There was also not a significant difference in the mean values of ERI at the outfall node between all of the bioretention cell application, all combination bioretention/green roof application, and all of the combination bioretention/green roof/rain barrel application scenarios. There was a significant difference, however, among the mean values of ERI at the outfall node when comparing any scenario containing a bioretention cell to the baseline condition, all green roof application, all rain barrel application, and all combination green roof/rain barrel application scenarios.
4.2.2.3 FWPI Analysis at Node N190

The FWPI data at node N190 was adjusted in the generalized linear model with a gamma distribution and evaluated using the Tukey-Kramer method for all pairwise comparisons. Each pairwise comparison was analyzed at a Type I error rate of 5% with the null hypothesis, $H_0$) $\mu_1=\mu_2=...=\mu_{56}$, and the alternative hypothesis, $H_1$) $\mu_1\neq\mu_2\neq...\neq\mu_{56}$. The following results were obtained (Figure 4.39):

- The statistical analysis failed to reject the null hypothesis in favor of the alternative hypothesis based on a Type I error rate of 5% when comparing scenarios 1, 12, 13, 14, 15, 16, 17, 18, 19, 20, 21, 22, 23, 24, 25, 26, 27, 28, 29, 30, 31, 32, 33, 34, 35, 36, 37, 38, 39, and 40 (hereby referred to as Group 3). Therefore, there was not sufficient evidence to conclude upon a significant difference among mean values of FWPI at node N190 between these Group 3 land management scenarios.
• The statistical analysis failed to reject the null hypothesis in favor of the alternative hypothesis based on a Type I error rate of 5% when comparing scenarios 2, 3, 4, 5, 41, 42, 43, 45, 47, 49, 51, 53, and 55 (hereby referred to as Group 4). There was not sufficient evidence to conclude upon a significant difference among mean values of FWPI at node N190 between these Group 4 land management scenarios.

• The statistical analysis failed to reject the null hypothesis in favor of the alternative hypothesis based on a Type I error rate of 5% when comparing scenarios 3, 4, 5, 6, 41, 42, 43, 44, 45, 46, 47, 48, 49, 50, 51, 53, 54, and 55 (hereby referred to as Group 5). Therefore, there was not sufficient evidence to conclude upon a significant difference among mean values of FWPI at node N190 between these Group 5 land management scenarios.

• The statistical analysis failed to reject the null hypothesis in favor of the alternative hypothesis based on a Type I error rate of 5% when comparing scenarios 4, 5, 6, 41, 42, 43, 44, 45, 46, 47, 48, 49, 50, 51, 52, 53, 54, and 55 (hereby referred to as Group 6). There was not sufficient evidence to conclude upon a significant difference among mean values of FWPI at node N190 between these Group 6 land management scenarios.

• The statistical analysis failed to reject the null hypothesis in favor of the alternative hypothesis based on a Type I error rate of 5% when comparing scenarios 4, 5, 6, 42, 43, 44, 45, 46, 47, 48, 49, 50, 51, 52, 53, 54, and 55 (hereby referred to as Group 7). Therefore, there was not sufficient evidence to conclude upon a significant difference among mean values of FWPI at node N190 between these Group 7 land management scenarios.
• The statistical analysis failed to reject the null hypothesis in favor of the alternative hypothesis based on a Type I error rate of 5% when comparing scenarios 5, 6, 7, 42, 44, 46, 48, 50, 52, 54, and 56 (hereby referred to as Group 8). There was not sufficient evidence to conclude upon a significant difference among mean values of FWPI at node N190 between these Group 8 land management scenarios.

• The statistical analysis failed to reject the null hypothesis in favor of the alternative hypothesis based on a Type I error rate of 5% when comparing scenarios 7, 8, and 56 (hereby referred to as Group 9). There was not sufficient evidence to conclude upon a significant difference among mean values of FWPI at node N190 between these Group 9 land management scenarios.

• The statistical analysis failed to reject the null hypothesis in favor of the alternative hypothesis based on a Type I error rate of 5% when comparing scenarios 8 and 9 (hereby referred to as Group 10). Therefore, there was not sufficient evidence to conclude upon a significant difference among mean values of FWPI at node N190 between these Group 10 land management scenarios.

• The statistical analysis rejected the null hypothesis in favor of the alternative hypothesis based on a Type I error rate of 5% when comparing the scenarios between those in Group 3 to all other scenarios. There was, therefore, sufficient evidence to conclude upon a significant difference among the mean values of FWPI at node N190 between the Group 3 land management scenarios and all other scenarios.

• The statistical analysis rejected the null hypothesis in favor of the alternative hypothesis based on a Type I error rate of 5% when comparing scenario 2 to scenarios 6, 7, 8, 9, 10, 11, 44, 46, 48, 50, 52, 54, and 56. There was sufficient evidence to conclude upon a
significant difference among mean values of FWPI at node N190 between the aforementioned land management scenarios.

- The statistical analysis rejected the null hypothesis in favor of the alternative hypothesis based on a Type I error rate of 5% when comparing scenario 3 to scenarios 8, 9, 10, 11, 52, and 56. There was sufficient evidence to conclude upon a significant difference among mean values of FWPI at node N190 between the aforementioned land management scenarios.

- The statistical analysis rejected the null hypothesis in favor of the alternative hypothesis based on a Type I error rate of 5% when comparing scenario 41 to scenarios 7, 8, 9, 10, 11, and 56. There was sufficient evidence to conclude upon a significant difference among mean values of FWPI between the aforementioned land management scenarios.

- The statistical analysis rejected the null hypothesis in favor of the alternative hypothesis based on a Type I error rate of 5% when comparing scenarios 4, 43, 45, 47, 49, 51, 53, and 55 to scenarios 7, 8, 9, 10, and 11. There was sufficient evidence to conclude upon a significant difference among mean values of FWPI between the aforementioned land management scenarios.

- The statistical analysis rejected the null hypothesis in favor of the alternative hypothesis based on a Type I error rate of 5% when comparing scenarios 5, 6, 42, 44, 46, 48, 50, 52, and 54 to scenarios 8, 9, 10, and 11. There was sufficient evidence to conclude upon a significant difference among mean values of FWPI between the aforementioned land management scenarios.

- The statistical analysis rejected the null hypothesis in favor of the alternative hypothesis based on a Type I error rate of 5% when comparing the scenarios 7 and 56 to scenarios 9,
There was sufficient evidence to conclude upon a significant difference among mean values of FWPI between the aforementioned land management scenarios.

- The statistical analysis rejected the null hypothesis in favor of the alternative hypothesis based on a Type I error rate of 5% when comparing the scenario 8 to scenarios 10 and 11. There was sufficient evidence to conclude upon a significant difference among mean values of FWPI between the aforementioned land management scenarios.

- The statistical analysis rejected the null hypothesis in favor of the alternative hypothesis based on a Type I error rate of 5% when comparing scenarios 9, 10, and 11. There was sufficient evidence to conclude upon a significant difference among mean values of FWPI between the aforementioned land management scenarios.

These findings indicate that there was not a significant difference among the mean values of FWPI at node N190 between the baseline, all green roof application, all rain barrel application, and all combination green roof/rain barrel application scenarios. A significant difference was found to exist among the mean values of FWPI at node N190 when comparing any scenario containing a bioretention cell to the baseline condition, all green roof application, all rain barrel application, and all combination green roof/rain barrel application scenarios. When comparing bioretention cell application scenarios (10%-60%), all combination bioretention cell/green roof application, and all bioretention/green roof/rain barrel application, no significant difference was found among the mean values of FWPI at node N190 between treatments. A significant difference was found to exist among the mean values of FWPI at node N190 when comparing all treatment scenarios to bioretention application scenarios ranging from 70-100%, and when comparing between bioretention application scenarios ranging from 70-100%.
4.2.2.4 ERI Analysis at Node N190

The ERI data at node N190 was adjusted in the generalized linear model with a gamma distribution and evaluated using the Tukey-Kramer method for all pairwise comparisons. Each pairwise comparison was analyzed at a Type I error rate of 5% with the null hypothesis, $H_0: \mu_1 = \mu_2 = \ldots = \mu_{56}$, and the alternative hypothesis, $H_1: \mu_1 \neq \mu_2 \neq \ldots \neq \mu_{56}$. Results of this analysis are summarized in the following (Figure 4.40):

- The statistical analysis failed to reject the null hypothesis in favor of the alternative hypothesis based on a Type I error rate of 5% when comparing scenarios 1, 12, 13, 14, 15, 16, 17, 18, 20, 21, 22, 23, 24, 25, 26, 27, 28, 29, 30, 31, 32, 33, 34, 35, 36, 37, 38, 39, and 40 (hereby referred to as *Group 11*). There was not sufficient evidence to conclude upon a significant difference among mean values of ERI at node N190 between these Group 11 land management scenarios.

Figure 4.39 Mean values of FWPI at node N190.
• The statistical analysis failed the null hypothesis in favor of the alternative hypothesis based on a Type I error rate of 5% when comparing scenarios 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 41, 42, 43, 44, 45, 46, 47, 48, 49, 50, 51, 52, 53, 54, 55, and 56 (hereby referred to as Group 12). There was not sufficient evidence to conclude upon a significant difference among mean values of ERI at node N190 between these Group 12 land management scenarios.

• The statistical analysis rejected the null hypothesis in favor of the alternative hypothesis based on a Type I error rate of 5% when comparing the scenarios between those in Group 11 and those in Group 12. There was, therefore, sufficient evidence to conclude upon a significant difference among the mean values of ERI at node N190 between the land management scenarios in these two groups.

These findings indicate that there was not a significant difference among the mean values of ERI at node N190 between the baseline condition, all green roof application, all rain barrel application, and all combination green roof/rain barrel application scenarios. There was also not a significant difference in the mean values of ERI at node N190 between all of the bioretention cell application, all combination bioretention/green roof application, and all of the combination bioretention/green roof/rain barrel application scenarios. There was a significant difference, however, among the mean values of ERI at node N190 when comparing any scenario containing a bioretention cell to the baseline condition, all green roof application, all rain barrel application, and all combination green roof/rain barrel application scenarios.
4.2.3 Discussion

Land management scenarios containing a bioretention cell produced the highest values of fresh water provision, erosion regulation, and flood regulation on average when compared to all other scenarios tested with the SWMM model. The bioretention cell not only provided detention storage, but treatment of total suspended solids, total nitrogen, and total phosphorus in the stormwater runoff. Green roofs and rain barrels did not provide any form of water quality treatment in the SWMM model, which likely contributed to lower ecosystem service indices for land management scenarios containing these BMPs.

Evaluation of fresh water provision at the outfall node indicated that there was not a significant difference in fresh water provision between any of the land management scenarios. None of the values of FWPI were calculated to be greater than or equal to one, indicating that there was poor fresh water provision at the outfall of the study area. This result implies that
targeted BMP implementation within the Cowskin Creek subwatershed did not improve fresh water provision throughout the overall study area.

Assessment of erosion regulation services at the outfall indicated significantly higher values in ERI for all land management scenarios containing a bioretention cell relative to those without. This indicates that improvement in erosion regulation services at the outfall of the study area was associated with the application of bioretention cells as a watershed management strategy. In addition, there was no significant difference in values of erosion regulation among land management scenarios containing bioretention cells. This means that, theoretically, a land management scenario containing 10% bioretention cell application will obtain the same erosion regulation provision as a land management scenario containing 100% bioretention cell application at the watershed scale examined. Lastly, values of erosion regulation were calculated to be greater than or equal to one for bioretention-based land management scenarios for the 30.5-mm (1.2-inch) design storm. This was an indication of excellent erosion regulation services because the annual erosion rate is less than the allowable erosion rate. The ERI value drops below one for larger design storms, indicating that the bioretention cell system may be unable to effectively treat the larger volumes of runoff associated with design storms of greater magnitude.

Changes in fresh water provision services at node N190 on Cowskin Creek were much more dramatic than changes observed at the outfall node. Ecosystem service provision generally increased as the percentage of runoff capture by the best management practices in each land management scenario increased. However, those land management scenarios containing bioretention cells demonstrated the greatest fresh water provision when compared to all other scenarios. There was no significant difference between the baseline scenario and those scenarios containing only green roofs and/or rain barrels. The fresh water provision index was observed to
be greater than one for 100% bioretention cell application, indicating excellent water quantity and water quality at the observation node during a 30.5-mm (1.2-inch) design storm. All other intensities of bioretention cell application provided significant improvement compared to the baseline land management scenario even though their values of FWPI were calculated to be less than one. Unlike the outfall node, a significant difference was found to exist comparing the values of FWPI between land management scenarios containing bioretention cells. However this is likely because node N190 is just downstream of the targeted BMP implementation area, so the effects of BMP application are amplified.

Evaluation of the changes in erosion regulation at node N190 on Cowskin Creek were also more apparent compared to the changes in erosion regulation observed at the model outfall. There was a significant difference between land management scenarios with bioretention cells and land management scenarios without, again indicating that the presence of a bioretention cell contributes to optimal ecosystem service provision. Differences in values of ERI at node N190 between the performance of the baseline land management scenario and those scenarios containing only green roofs and/or rain barrels were not significant. Land management scenarios containing bioretention cells consistently demonstrated ERI values greater than or equal to one for all design storm magnitudes, indicating excellent erosion regulation services at this observation location. The scenario with 100% bioretention cell application demonstrated the highest value of ERI among all of the land management scenarios.

Assessment of the changes in flood regulation at node N154-28 found that those land management scenarios with more intense BMP application had the highest FRI values, though the change in FRI values among scenarios compared to the baseline was negligible. Similar to the results obtained evaluating FWPI and ERI changes, land management scenarios containing a
bioretention cell exhibited the highest values of FRI. The largest percent change in FRI was 0.46%. None of the values of FRI were observed to be greater than or equal to one, indicating that the capacity of bioretention or other BMPs implemented in this study to reduce flooding was limited. An explanation for this result is that the SWMM model only predicted flooding to occur for design storms of greater magnitude, during which the hydrologic regulating functions of BMPs is diminished significantly.
Chapter 5 - Broader Impacts

Publications such as the Millennium Ecosystem Assessment and the EU 2020 Biodiversity Strategy were published in response to growing concern about the state of the environment. Human activity has made significant progress in providing the growing world population with necessary resources such as food, fuel, and fiber. Unfortunately these advancements have cost the world in terms of environmental health, since singularly focused management strategies aim to maximize a single ecosystem service rather than maintaining overall ecosystem service provision. Ecosystem health is determined by the range of various services that an ecosystem provides and is often characterized in terms of vigor, resilience, and organization (Costanza, 2012). Ecosystems that lack these characteristics, such as those that only provide a single ecosystem service, are not sustainable long-term.

The projected implications of climate change have sparked action to reduce human impact on the environment and build up healthy ecosystems. Ecosystems that lack characteristics such as vigor, resilience, and organization have proven to be unable to defend themselves from extreme events such as heat waves, flooding, and water scarcity/droughts (Berte & Panagopoulos, 2014). Many scientists have attributed the rising number and intensity of super-storms, such as Hurricanes Sandy and Katrina, along with the destruction they have left behind, to the inability of ecosystems to defend themselves against natural disasters. Climate change projections predict that there will be substantial changes in precipitation patterns around the world, with larger and more intense storm events becoming the norm. Municipalities will have to adapt their management strategies to the extremes – cities must have the capability to mitigate larger storm events to prevent flooding while also maintaining the ability to capture and retain water during periods of extended drought. Research that aims to understand the interactions
between ecosystem services and identifies preferential management strategies in response to a changing climate will be a key component in restoring ecosystem health around the world.

The overall goal of this research was to understand the role that holistic watershed management plays in the provision of ecosystem services. This research aimed to identify the types of ecosystem services provided by urban best management practices, as well as understand the extent to which targeted BMP implementation is necessary to achieve desirable quantities of ecosystem service provision. Lastly, this research intended to recognize if holistic watershed management across the rural-urban gradient could improve the provision of ecosystem services within the urban area. The outcome of this experiment will provide further insight into methods of successful management for the provision of fresh water, flood regulation, and erosion regulation. This knowledge will contribute towards building healthy urban ecosystems that possess the vigor, resilience, and organization to thrive in the changing climate.

This research experimented with three different types of urban best management practices within the study area. These urban BMPs were applied across a variety of scenarios as part of a targeted implementation program in a smaller sub-watershed within the greater study area. Bioretention cells were chosen to represent the function of infiltration-based urban BMPs. Green roofs and rain barrels were the remaining two urban BMPs chosen for simulation. Bioretention cells demonstrated excellent provision of fresh water and erosion regulation services immediately downstream from the implementation site on Cowskin Creek. Farther downstream at the outfall node, however, only erosion regulation services were apparent. The difference in ecosystem services demonstrated by bioretention cells at node N190 and the outfall node was likely due to the distance between locations and spatial disconnectivity. Bioretention cells did not provide any flood regulation services at either location.
Neither green roofs nor rain barrels demonstrated any fresh water, erosion regulation, or flood regulation provisioning services at either location. There did not appear to be any significant changes comparing the baseline land management scenario and scenarios implementing only these two types of BMPs. Rain barrels do not provide any sort of water quality treatment and that may account for their lack of ecosystem service provision in the model simulation. Current literature on the quantity of water quality treatment by green roofs is inconsistent, and therefore they were not designated for stormwater treatment in model simulations. It is important to note that both green roofs and rain barrels provide other types of ecosystem services that were not quantified in this experiment, so they should not be entirely discounted because of these conclusions. Some additional ecosystem services provided by green roofs include the addition of insulation to existing buildings to reduce heating/cooling costs, the reduction of building albedo to counteract the urban heat island effect, and the addition of green space that may be used as a recreational/cultural/spiritual service. Additional ecosystem service indices should be developed that quantify the supplementary services provided by urban best management practices.

The extent of ecosystem service provision diminished greatly from node N190 in Cowskin Creek downstream to the outfall node of the model along the Arkansas River. Along Cowskin Creek it was apparent that an increase in freshwater provision and erosion regulation services is associated with an increase in bioretention cells. However, at the outfall node, implementation of bioretention cells capturing 10% of runoff achieved the same quantity of erosion regulation services as bioretention cells capturing 100% of stormwater runoff. This is significant since there is much less space, time, and cost associated with constructing bioretention cells to treat 10% of impervious area versus 100% of impervious area. Thus it is
important to identify if there is difference between the location of ecosystem service provision (i.e. at the bioretention cell) and the beneficiary of the ecosystem service (i.e. at the outfall node) in future applications of bioretention cell installation so that watershed management goals are met.

The extent of ecosystem service provision diminished significantly as the size and magnitude of the storm event increased. Since each BMP was designed to successfully treat 100% of runoff from the 30.5-mm (1.2-inch) design storm, it is logical that each land management scenario exhibited the highest ecosystem service provision during this storm size. The amount of fresh water provision and erosion regulation decreased as the size of the design storm increased, with the worst performance for each scenario exhibited during the 100-year design storm. As municipalities aim to increase the resilience and autonomy of cities in response to a changing climate, a first step may be to increase the size of the design storm to a larger storm event. BMPs that are designed to successfully treat stormwater runoff from the 5-year or 10-year storm will be more successful in treating runoff associated with the 100-year storm event than the BMPs that were used in this experiment.

Holistic watershed management across the rural-urban gradient was somewhat successful in this experiment. Improvement in erosion regulation services were observed at all locations across the rural-urban gradient, though there was no noticeable change across the watershed in freshwater provision or flood regulation. Urban stormwater BMPs may need to be implemented on a larger scale than was tested in this experiment in order to achieve desired results in freshwater provision and flood regulation. It is, therefore, important to identify the types of ecosystem services and the location of beneficiaries in a management program before applying holistic watershed management. Increased research in the area of holistic watershed management
should identify whether this management strategy is beneficial in connecting the urban-rural environment to improve watershed-scale ecosystem health.

Future research in understanding the relationship between urban best management practices and ecosystem services should:

1. Identify the spatial relationships between ecosystem services to maintain ecosystem service provision throughout a watershed from the implementation site to the beneficiary.

2. What scale of targeted BMP implementation is necessary to achieve desired results in freshwater provision and flood regulation?

3. Expand analysis to other types of ecosystem services that were not able to be quantified in this modeling study.

4. Account for stream channel erosion occurring in natural channels within the watershed.

5. How does ecosystem service provision increase if BMPs are designed for larger storm events (instead of the City of Wichita’s 30.5-mm design storm)? How does this change in design impact ecosystem service provision across all storm events?

It is important to note that these results did not take into account the performance of best management practices during a specific time of year. Simulations also assumed no antecedent moisture content. Modeling simulations that take into account climatic variations throughout the year as well as the effect of antecedent moisture content on the BMP-ecosystem service relationship could provide additional valuable information on the subject.

Appleton, A. F. (2002). *How New York City used an ecosystem services strategy carried out through an urban-rural partnership to preserve the pristine quality of its drinking water and save billions of dollars*. Tokyo.


Environmental Services Division. (2007). Bioretention manual. Prince George's County, Maryland: Department of Environmental Resources.


Rooflite Soil. *Rooflite certified green roof media, specifications: Rooflite extensive MCL.*


## Appendix A - Reference Tables

### Table A.1 Subcatchment impervious factor values (U.S. Department of the Interior, 2015).

<table>
<thead>
<tr>
<th>Land-Use Category</th>
<th>% Impervious</th>
<th>Impervious Factor</th>
</tr>
</thead>
<tbody>
<tr>
<td>Developed, Open Space</td>
<td>&lt;20%</td>
<td>0.1</td>
</tr>
<tr>
<td>Developed, Low Intensity</td>
<td>20-49%</td>
<td>0.35</td>
</tr>
<tr>
<td>Developed, Medium Intensity</td>
<td>50-79%</td>
<td>0.65</td>
</tr>
<tr>
<td>Developed, High Intensity</td>
<td>80-100%</td>
<td>0.9</td>
</tr>
</tbody>
</table>

### Table A.2 Manning's N overland flow values.

<table>
<thead>
<tr>
<th>Land-Use Category</th>
<th>Manning’s N</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cultivated Crops</td>
<td>0.035</td>
</tr>
<tr>
<td>Deciduous Forest</td>
<td>0.1</td>
</tr>
<tr>
<td>Developed, Open Space</td>
<td>0.015</td>
</tr>
<tr>
<td>Developed, Low Intensity</td>
<td>0.015</td>
</tr>
<tr>
<td>Developed, Medium Intensity</td>
<td>0.012</td>
</tr>
<tr>
<td>Developed, High Intensity</td>
<td>0.012</td>
</tr>
<tr>
<td>Woody Wetlands</td>
<td>0.07</td>
</tr>
<tr>
<td>Open Water</td>
<td>0.015</td>
</tr>
<tr>
<td>Herbaceous</td>
<td>0.13</td>
</tr>
<tr>
<td>Hay/Pasture</td>
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</tr>
<tr>
<td>Emergent Herbaceous Wetlands</td>
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</tr>
<tr>
<td>Mixed Forest</td>
<td>0.1</td>
</tr>
<tr>
<td>Barren Land</td>
<td>0.03</td>
</tr>
<tr>
<td>Shrub/Scrub</td>
<td>0.06</td>
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### Table A.3 Depression storage values.

<table>
<thead>
<tr>
<th>General Land-Use Category</th>
<th>Land-Use Category</th>
<th>Depression Storage [mm (in)]</th>
</tr>
</thead>
<tbody>
<tr>
<td>Impervious Surfaces</td>
<td>Open Space, Low Intensity, Medium Intensity, High Intensity (Impervious)</td>
<td>1.905 (0.075)</td>
</tr>
<tr>
<td>Lawns</td>
<td>Open Space, Low Intensity, Medium Intensity, High Intensity (Pervious)</td>
<td>3.81 (0.15)</td>
</tr>
<tr>
<td>Pasture</td>
<td>Cultivated Crops, Herbaceous, Hay/Pasture, Barren Land</td>
<td>5.08 (0.20)</td>
</tr>
<tr>
<td>Forest Litter</td>
<td>Deciduous Forest, Mixed Forest, Shrub/Scrub</td>
<td>7.62 (0.30)</td>
</tr>
<tr>
<td>Soil Texture Class</td>
<td>Suction [mm (in)]</td>
<td>Porosity (fraction)</td>
</tr>
<tr>
<td>--------------------------</td>
<td>-------------------</td>
<td>---------------------</td>
</tr>
<tr>
<td>Sand</td>
<td>49.02 (1.93)</td>
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</tr>
<tr>
<td>Loamy Sand</td>
<td>60.96 (2.4)</td>
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</tr>
<tr>
<td>Coarse(Sandy)-Loam</td>
<td>109.98 (4.33)</td>
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</tr>
<tr>
<td>Loam</td>
<td>88.9 (3.5)</td>
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<tr>
<td>Silt Loam</td>
<td>169.93 (6.69)</td>
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<tr>
<td>Urban Land</td>
<td>219.96 (8.66)</td>
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</tr>
<tr>
<td>Fine(Clay)-Loam</td>
<td>210.06 (8.27)</td>
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<tr>
<td>Silty Clay Loam</td>
<td>270 (10.63)</td>
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</tr>
<tr>
<td>Sandy Clay (Loam)</td>
<td>240.03 (9.45)</td>
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<tr>
<td>Fine(Clay)-Silty</td>
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<tr>
<td>Fine(Clay)</td>
<td>320.04 (12.6)</td>
<td>0.475</td>
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## Appendix B - Calculated Results

Table B.1 Values of FWPI at the outfall node on the Arkansas River.

<table>
<thead>
<tr>
<th>Scenario #</th>
<th>30.5-mm</th>
<th>5-Year</th>
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<th>25-Year</th>
<th>100-Year</th>
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<td>1</td>
<td>0.3255</td>
<td>0.0281</td>
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<td>0.0513</td>
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<td>0.0365</td>
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<td>0.0202</td>
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<td>0.0365</td>
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<td>0.3493</td>
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<td>0.0365</td>
<td>0.0352</td>
<td>0.0202</td>
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<td>6</td>
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<td>0.0365</td>
<td>0.0352</td>
<td>0.0202</td>
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<tr>
<td>7</td>
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<td>0.0364</td>
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<td>8</td>
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Table B.4 Percent change in values of ERI at the outfall node along the Arkansas River.
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Appendix C - SAS Code

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proc univariate data=Outfall_FWPI;
    histogram FWPI;
run;

proc glimmix data=Outfall_FWPI;
   class Scenario Rainfall;
   model FWPI = Scenario /dist=beta;
   random Rainfall;
   lsmeans Scenario/ pdiff adjust=tukey ilink;
   output out=residuals residual=residual predicted=predicted;
run;
quit;

data Outfall_ERI;
   input Scenario $ Rainfall $ ERI;
   datalines;
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   2 1.2inch 1.967111952
   3 1.2inch 1.967644633
   4 1.2inch 1.968264683
   5 1.2inch 1.968792752
   6 1.2inch 1.969305669
   7 1.2inch 1.969799616
   8 1.2inch 1.9701783
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proc univariate data=Outfall_ERI;
histogram ERI;
run;

proc glimmix data=Outfall_ERI maxopt=2000;
class Scenario Rainfall;
model ERI = Scenario /dist=gamma;
random Rainfall;
lsmeans Scenario/ pdiff adjust=tukey ilink;
output out=residuals residual=residual predict=predicted;
run;
quit;

data N190_FWPI;
input Scenario $ Rainfall $ FWPI;
datalines;
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histogram FWPI;
run;

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class Scenario Rainfall;
model FWPI = Scenario /dist=gamma;
random Rainfall;
lsmeans Scenario/ pdiff adjust=tukey ilink;
output out=residuals residual=residual predicted=predicted;
run;
quit;

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run;

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histogram ERI;
run;

proc glimmix data=N190_ERI maxopt=2000;
class Scenario Rainfall;
model ERI = Scenario /dist=gamma;
random Rainfall;
lsmeans Scenario/ pdiff adjust=tukey ilink;
output out=residuals residual=residual predicted=predicted;
run;
quit;
Appendix D - Model Code

[TITLE]
Final project. Baseline control, no LIDS (Scenario 1)

[OPTIONS]
;;Options            Value
;;------------------ ------------
FLOW_UNITS           CFS
INFILTRATION         GREEN_AMPT
FLOW_ROUTING         DYNWAVE
START_DATE           07/01/2015
START_TIME           00:00:00
REPORT_START_DATE    07/01/2015
REPORT_START_TIME    00:00:00
END_DATE             07/02/2015
END_TIME             00:00:00
SWEEP_START          07/01
SWEEP_END            07/02
DRY_DAYS             15
REPORT_STEP          0:15:00
WET_STEP             0:15:00
DRY_STEP             0:15:00
ROUTING_STEP         1
ALLOW_PONDING        YES
INERTIAL_DAMPING     PARTIAL
VARIABLE_STEP        0.75
LENGTHENING_STEP     0
MIN_SURFAREA         12.566
NORMAL_FLOW_LIMITED  BOTH
FORCE_MAIN_EQUATION  H-W
LINK_OFFSETS         DEPTH
MIN_SLOPE            0
MAX_TRIALS           8
HEAD_TOLERANCE       0.005
SYS_FLOW_TOL         5
LAT_FLOW_TOL         5

[EVAPORATION]
;;Type          Parameters
;;-------------- ---------
MONTHLY        0 0 0 0 0 0 0 0 0 0 0 0
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DRIY_ONLY      YES

[RAINGAGES]
;;           Rain    Time   Snow   Data
;;Name        Type      Intrvl Catch   Source
;;---------- ----------- ------ ------- --------
;Design Storm (Wichita)
SCS_24h_Type_II_1.2in INTENSITY 0:15  1.0 TIMESERIES SCS_24h_Type_II_1.2in
;5-year, 24 hour design storm (Wichita)
SCS_24h_Type_II_4.24in INTENSITY 0:15  1.0 TIMESERIES SCS_24h_Type_II_4.24in
;10-year, 24-hour design storm (Wichita)
SCS_24h_Type_II_4.98in INTENSITY 0:15  1.0 TIMESERIES SCS_24h_Type_II_4.98in
;25-year, 24-hour design storm (Wichita)
SCS_24h_Type_II_6.05in INTENSITY 0:15  1.0 TIMESERIES SCS_24h_Type_II_6.05in
;100-year, 24-hour design storm (Wichita)
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211
Nitrogen MG/L 0 0 0 0 0 0 NO *
Phosphorus MG/L 0 0 0 0 0 0 NO *
TSS MG/L 0 0 0 0 0 0 NO *

[LANDUSES]

;; Name Cleaning Fraction Last Interval Available Cleaned

; Pervious land cover that has been disturbed by urbanization (parks, lawns, etc.)
MixedDevelopment 0 0 0

; Impervious cover including paved parking lots, roads, etc.
Pavement 0 0 0

; Impervious cover including rooftops from residential homes and buildings.
Roofs 0 0 0

; Undisturbed land.
Undeveloped 0 0 0

[COVERAGES]

;; Subcatchment Land Use Percent

100 MixedDevelopment 3.657
100 Pavement 0.379
100 Roofs 0.095
100 Undeveloped 95.869
101 MixedDevelopment 8.855
101 Pavement 4.029
101 Roofs 1.007
101 Undeveloped 86.108
102 MixedDevelopment 16.5
102 Pavement 8.305
102 Roofs 2.076
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103 MixedDevelopment 17.856
103 Pavement 8.559
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103 Undeveloped 71.445
104 MixedDevelopment 44.449
104 Pavement 14.057
104 Roofs 3.514
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105 MixedDevelopment 15.765
105 Pavement 2.279
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106 MixedDevelopment 42.753
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[Symbols]
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