SUSTAINABILITY ANALYSIS OF DISTRIBUTED, GREEN STORMWATER INFRASTRUCTURE

BY

JAMES FRANCIS CANNING

THESIS

Submitted in partial fulfillment of the requirements
for the degree of Master of Science in Civil Engineering
in the Graduate College of the
University of Illinois at Urbana-Champaign, 2018

Urbana, Illinois

Advisor:

Assistant Professor Ashlynn S. Stillwell

ABSTRACT

As urban areas continue to grow in population and density of development, stormwater management has become a higher priority for municipal leaders, civil engineers, and residents. Traditional gray infrastructure, i.e., large, piped conveyance networks, effectively removed stormwater runoff and all its associated problems (especially water quality issues) from urban areas beginning in the early 20th century and was heralded as a public health breakthrough. However, gray infrastructure networks have not kept pace with the growth of cities, resulting in a variety of issues including increased peak runoff flow and volume, flooding, increased pollutant loading, degraded water quality, and enhanced urban heat island effect. Green stormwater infrastructure (GSI) mimics pre-development conditions in urban areas attempting to manage excess stormwater runoff. GSI can take many different forms, but its primary functions are to infiltrate stormwater and treat the water on-site, thus reducing runoff volume and pollutant loading downstream. Rain gardens, permeable pavement, green roofs, bioretention cells, street trees, swales, and grass strips are all examples of GSI. Distributed, green networks can provide additional environmental benefits beyond hydrologic improvements, including economic and social co-benefits. GSI networks present an attractive opportunity for decision makers to achieve necessary hydrologic improvements and possibly provide residents with additional co-benefits.

In this analysis, GSI is investigated in several capacities. First, GSI is taken out of its traditional urban context and analyzed in an agricultural setting. “Agricultural green infrastructure” is sought after as a potential solution to increasing water quality issues upstream of Des Moines, Iowa, USA, representing similar water quality issues throughout much of the agricultural Midwest. In these areas, heavy fertilizer use on agricultural fields leads to increased nitrate loading in nearby surface waters, presenting a risk locally to people using such water as a source of drinking water. Two main proposals are put forward: 1) a widespread, agricultural green infrastructure buildout is proposed throughout the Raccoon River Watershed (RRW) upstream of Des Moines; and 2) treating the water at a nitrate removal facility is investigated as the centralized, gray approach. Both scenarios theoretically treat the polluted water to a high enough quality to consume as drinking water, but they differ in co-benefits and cost. The distributed, green approach is estimated to cost between \$135 – \$160 million: maintaining the current new nitrate removal facility costs around \$71 million, but building a new facility can cost up to \$184 million. However, building widespread riparian buffers offers increased employment benefits, improves environmental conditions, and helps address the rural-urban divide spreading throughout the United States. Although agricultural green infrastructure might present a high initial cost, it is still less costly than building a new facility sometime in the future and provides comparatively more benefits.

Distributed, green networks can provide significant co-benefits to communities, and these co-benefits are the focus of the second portion of analyses. Specifically, social co-benefits in urban areas are investigated. Many GSI calculators, tools, and frameworks exist to help engineers and planners design GSI projects, and yet the literature does not speak to the inclusion of social co-benefits in these GSI calculators. A qualitative analysis is performed exploring whether 21 GSI calculators, tools, and frameworks mention and/or attempt to quantify social co-benefits. Of these 21 calculators, only five mention social co-benefits, and merely two of these attempt to quantify such benefits. A paradigm shift is suggested, one in which social co-benefits are included at the planning stages of GSI projects to ensure that such benefits are available to people engaging the GSI.

One derivative of such social co-benefits is the advancement of environmental justice, defined as the equitable distribution of both environmental ills and environmental benefits, regardless of race, income, or any other socioeconomic quality. Historically, low-income and minority neighborhoods have not been locations for environmental outreach. This trend is statistically analyzed in Chicago, IL and Philadelphia, PA, two national leaders in GSI installation, through regression and non-parametric testing, respectively. Results in Philadelphia indicate that the city seems to be achieving its stated goal of investing in “environmental justice” communities, based on median household income and race. Overall, results support the conclusion that open-source data can indeed be analyzed to track environmental justice efforts, an important social co-benefit of GSI installation.

GSI is a relatively new concept, but it has already been adopted in cities throughout the entire globe. It offers many benefits beyond those benefits provided by traditional gray infrastructure, and GSI is integral to many cities' long-term improvement plans. The present analyses add to the growing body of knowledge concerning GSI through analyzing GSI in an agricultural context and addressing its potential social co-benefits.

To my mother, for her unwavering support throughout all my years. And to my father, who worked tirelessly to set up mine and my siblings' successes.

ACKNOWLEDGMENTS

I would like to thank my advisor, Ashlynn S. Stillwell, for her patience, guidance, and genuine support throughout my master's degree. Thank you for taking a chance with the volunteer from Tucson. I would also like to thank Joshua M. Peschel for his support in getting me started as a graduate student. I would be remiss not to thank the fabulous members of the Stillwell Research Group. Thank you all for the guidance throughout my master's, and especially thank you for the many, many laughs we shared. Lastly, I would like to thank all my friends and family — you all are truly dear to my heart, and I cannot thank God enough for your continued presence in my life. This work was supported by a CEE Fellowship and the Department of Civil and Environmental Engineering at the University of Illinois at Urbana-Champaign.

TABLE OF CONTENTS

LIST OF ABBREVIATIONS

ACS	American Community Survey
BMP	Best Management Practice
CNT	Center for Neighborhood Technology
CRP	Conservation Reserve Program
CSO	Combined Sewer Overflow
EPA	Environmental Protection Agency
EQIP	Environmental Quality Incentive Program
DEM	Digital Elevation Model
DMWW	Des Moines Water Works
GIS	Geographic Information Systems
GSI	Green Stormwater Infrastructure
PLS	Pure Live Seed
RRW	Raccoon River Watershed
SWPP	Source Water Protection Program
TMDL	Total Maximum Daily Load
TWI	Topographic Wetness Index
USDA	United States Department of Agriculture
USGS	United States Geological Survey
VISES	Victoria Institute of Strategic Economic Studies
WQT	Water-Quality Trading
WWTP	Wastewater Treatment Plant

LIST OF SYMBOLS

ha	Hectare
β	Slope
A_s	Specific catchment area, units of $m^2\ m^{-1}$
mg/L	Milligrams per liter
kg	Kilogram

CHAPTER 1

INTRODUCTION

In both urban and agricultural areas throughout the globe, stormwater can be an integral part of the local hydrologic cycle. Precipitation has many positive effects, perhaps the greatest of these being the growth of vegetation and food for human consumption. However, stormwater can also have negative effects if, for example, there is excess stormwater and the local conditions cannot handle the large volume of water. Managing stormwater is crucial in both agricultural and urban contexts, and such management is the focus of this thesis.

1.1 Agricultural Runoff and Water Quality Implications

Agricultural intensification, including the widespread use of fertilizer and livestock manure, has led to increased inputs, improved agronomic practices, and enhanced crop varieties [240]. Intensifying agriculture is necessary to meet the demand for crops from an increasing global population (projected 2.3 billion person increase by 2050), but it comes with potentially hazardous environmental effects [85]. Dabrowski et al. [48] found that the water required to address water quality degradation in agriculture is similar to the water quantity required to produce crops, indicating the extent to which fertilizer intensification affects water quality. Impairment of waters due to agricultural runoff is a problem for many rural and urban areas in the United States — a 2010 United States Geological Survey (USGS) study found that nitrates were above recommended levels in 64 percent of surface water and shallow groundwater monitoring wells throughout the United States [225]. Riparian buffers are analyzed as one agricultural green infrastructure practice intended to reduce nutrient levels. Buffers prevent degradation of surface water through uptake of nitrates by the vegetation, as well as facilitating conditions necessary for denitrification and phosphorus removal [83, 105]. Nitrogen, phosphorus, and sediment pollution challenges exist in the Raccoon River Watershed (RRW) [156], representing one of the most nitrate-polluted watersheds in the United States. Jones et al. [125] analyzed 2016 data to determine that Iowa contributes 41 percent of the nitrate load to the Gulf of Mexico. Thus, the RRW is a fitting location for the analysis because of its large contribution to degraded water quality. Furthermore, it is the setting of a recently-decided, contentious lawsuit between Des Moines Water Works (DMWW) and three upstream drainage districts (*Bd. of Water Works Trs. of Des Moines v. Sac Cty. Bd. of Supervisors as Trs. of Drainage Dists. 32, 42, 65, 79, 81, 83, 86, No. 5:15-cv-04020, 2015 WL 1191173*). The lawsuit is a microcosm of a larger theme throughout the United States: a rural-urban divide that precipitates deadlock over environmental issues. The final ruling stated that water leaving the drainage districts was not regulated, and the districts did not have to recompense DMWW monetarily for having to treat the nitrate-laden water. The RRW is unique in the extent to which nitrate pollution has led to greater issues, and thus it was chosen as the focus of the analysis.

The RRW, as a source of pollutants both for Des Moines residents and the Gulf Hypoxia Zone, is analyzed to determine where riparian buffers would be most effective throughout the watershed and what scale of cost is associated with installing buffers. Furthermore, there have been no studies, to the author's knowledge, contrasting centralized gray infrastructure and distributed green infrastructure in an agricultural context. Many studies analyze green versus gray infrastructure in an urban setting, not an agricultural setting [79, 207, 226]. The following questions are addressed:

- What are the ramifications of using agricultural green infrastructure for nitrate reduction?
- Where might agricultural green infrastructure be placed, and how much will it cost?

The present analysis is added to the greater body of knowledge in gray and green infrastructure comparisons.

1.2 Social Co-Benefits of Green Stormwater Infrastructure

Distributed green spaces offer benefits in many different contexts, and they are an alternative to centralized, gray solutions to managing water quality and quantity. Such green networks originally sprouted from urban applications, termed “green stormwater infrastructure” (GSI). GSI offers a multitude of environmental, economic, and social benefits, and researchers have quantified many of the hydrologic (environmental) benefits [e.g., 21, 146, among many others]. Further research has shown complementary environmental and economic co-benefits from some forms of GSI, such as energy savings through the use of green roofs [69, 239, 263]. However, fewer findings have been published concerning the potential social co-benefits. This work aims to 1) assess the state of the art in quantifying social co-benefits of green infrastructure, and 2) analyze the environmental justice impacts of green infrastructure installation.

Social co-benefits concern the ways in which people *interact* with GSI, how people *perceive* GSI, and emotional/physical/mental health benefits derived from engaging GSI. A portion of this thesis analyzes various existing stormwater calculators regarding quantification of social co-benefits of GSI. The following question provides the motivation for this section of the thesis:

- To what extent are social co-benefits included in planning for GSI projects, as reflected by available calculators?

1.3 Environmental Justice

One sought-after derivative of social co-benefits is environmental justice. Environmental justice is defined as an equitable distribution of environmental benefits and environmental ills regardless of race, income, or social status [97], i.e., “fairness in the distribution of environmental well being” [84]. Di Chiro [56] suggests that the ‘environment’ need not be a pristine, unharmed, and removed nature; instead, she broadens environment to include “the place you work, the place you live, the place you play.” Just as public goods such as libraries and schools or private goods such as grocery stores can be distributed differently between affluent and impoverished areas, ecosystem services (i.e., the benefits of GSI) can similarly be differently (even "unjustly") distributed. This broad sense of environmental justice includes equitable distribution of access to GSI and the benefits it offers. GSI presents an excellent opportunity to merge the hydrologic improvements often required of green spaces with the goal of increasing access to green space for marginalized communities.

Anecdotal criticisms claim that GSI projects are often implemented in more affluent urban areas or areas favoring more racial homogeneity. This spatial challenge is expored by statistically analyzing available GSI and socioeconomic data to determine whether GSI installations are in fact associated with certain socioeconomic trends. In the second part of the analysis, statistical analyses of GSI and socioeconomic data are conducted to assess levels of environmental justice, demonstrated for two major cities located in the United States: Chicago, IL and Philadelphia, PA. Chicago is a national leader in GSI installation, especially regarding green roofs [133]. Similarly, Philadelphia has an ambitious GSI plan ('Green City, Clean Waters') wherein the city will implement 25 years' worth of GSI projects [191]. There is a significant precedent in research using geographic information systems (GIS) to assess environmental justice within a city, and the approach employed here similarly uses GIS [47, 75, 99]. The following questions is of import in this portion of the thesis:

- How might existing socioeconomic data be used to analyze GSI projects and their implications on environmental justice?

The present analyses support further application of GSI in both agricultural and urban contexts, improving not only hydrologic conditions but the livelihoods of the people that interact with the GSI. Additional background on GSI, stormwater management, and environmental justice is presented in Chapter 2. In Chapter 3, a full analysis is performed on the feasibility of utilizing riparian buffers as agricultural green infrastructure in the Raccoon River Watershed with the goal of reducing nitrate pollution. Results can help inform future solutions and policies regarding agricultural runoff. In Chapter 4, a review is performed that seeks out how many currently available GSI calculators reference social co-benefits. A paradigm shift is suggested wherein social co-benefits are included in the planning stages of GSI projects, and a consequent statistical analysis of Chicago and Philadelphia provides an example of such monitoring. Chapter 5 outlines conclusions and proffers potential applications in the GSI arena.

CHAPTER 2

BACKGROUND

2.1 Green Stormwater Infrastructure

GSI is an alternative to traditional gray infrastructure in urban areas, i.e., piped conveyance networks that transport stormwater away. GSI mimics pre-development conditions, allowing stormwater to naturally infiltrate into soil, reducing runoff volumes, peak flows, and pollutant loads. Bioretention cells, green roofs, tree trenches, and pervious pavement are all examples of GSI. Many American cities feature combined sewer systems, and when there are large rain events, the system is overloaded with sewage and stormwater, oftentimes causing a combined sewer overflow (CSO). The Clean Water Act states that cities must have a permit to legally discharge water in a CSO [1]. GSI is one approach to helping solve the CSO problem in urban areas. Many engineers, landscape architects, and others have defined "green infrastructure." Benedict and McMahon [19] state that GSI is "an interconnected network of green space that conserves natural ecosystem values and functions and provides associated benefits to human populations." Tzoulas et al. [244] claim that GSI can be "considered to comprise of all natural, semi-natural and artificial networks of multifunctional ecological systems within, around and between urban areas." In most cities, GSI cannot completely replace the need for gray infrastructure; however, a distributed network of GSI certainly reduces the burden placed on gray infrastructure systems, many of which are outdated [221].

There are various types of GSI, including bioretention areas, permeable pavement, urban trees, infiltration trenches, and green roofs. These different types of GSI have different mechanisms through which they function, as well as different physical forms and benefits. Bioretention areas, often in the form of rain gardens, are sunken areas designed to capture and accept large amounts of stormwater. Rain gardens are populated with native and water-intense vegetation, perennial vegetation, and trees. Their benefits include increased groundwater recharge, peak flow attenuation, decreased surface runoff, and pollutant removal [58]. Davis [53] experimentally found that bioretention cells are effective for decreasing peak flows, reducing stormwater volumes, and delaying peak flows. Kim et al. [139] physically demonstrated how bioretention areas remove pollutants in runoff through "sedimentation, filtration, and sorption on mulch and soil layers, plant uptake, and biodegradation by soil microorganisms."

Green roofs are another type of GSI. They are created through structural enhancements to rooftops that then support a layer of soil and a layer of "green" vegetation. There are two main types of green roofs; 'intensive' roofs with thick soil layers (and thus more structural requirements), and 'extensive' roofs with thinner soil layers [58]. Fioretti et al. [74] showed that green roofs' greatest stormwater benefits include volume reduction (through stormwater storage in the media and evapotranspiration by the plants), peak flow attenuation, and increased concentration time. Green roofs have also been shown to reduce urban heat island effects and are an effective means of addressing the effects of climate change in today's cities [233, 262]. A further manifestation of GSI is permeable pavement. There are many methods of constructing permeable pavement, but the high-level idea is that the pavement sits above a layer of soil and gravel that infiltrate stormwater runoff. Permeable pavements are suitable in low-traffic areas such as alleys and parking lots, as well as bike/pedestrian paths. Their benefits include recharging groundwater, pollutant removal, and reducing runoff volumes [86].

There are many options for cities to utilize rights-of-way for GSI implementation, including bioretention cells, infiltration basins, or retention trenches. "Infiltration devices" include retention basins and infiltration trenches [86]. Infiltration trenches are a form of GSI that can reduce runoff volume and improve water quality through sorption and filtration [5, 64, 210]. Infiltration trenches are compact and can fit in many urban environments, providing another tool in GSI projects. Holman-Dodds et al. [106] confirmed that such devices reduce peak flow and runoff volume, as well as improving groundwater quality through pollutant removal. Urban trees are an environmentally and economically effective form of GSI. Urban trees can provide water quality protection during longer, less intense rainfall events that are responsible for the majority of pollutant washout [266]. Tree roots penetrate compacted urban soils and can increase infiltration in those soils by as much as 63 percent [14]. Canopy interception reduces runoff volume, prevents pollutant washout, and reduces soil erosion [10]. An additional environmental benefit of many of these vegetative forms of GSI is the sequestration of air pollution [6, 181].

Philadelphia, a focus of this thesis, encourages the use of GSI through several avenues. Property owners are responsible for all stormwater within their surface parking lots. Small lots must be reviewed by the City Planning Commission to ensure the lot's catch basins and drains will accommodate runoff, while large lot owners might be required to submit a stormwater plan to the Philadelphia Water Department [190]. Every commercial parking lot is required to design at least 10 percent of the total lot area to be landscaped. The City Planning Commission explicitly urges developers to utilize GSI in the design manual, including porous pavement, solar power parking lot canopies, porous asphalt comparison, bioretention cells, and curb cuts [190]. The City Planning Commission and the Philadelphia Streets Department require curb cuts in residential driveways and garages. Furthermore, Philadelphia has realized the social importance of maintaining streetscape devoid of front parking/garages and thus requires that developers apply for a permit for front parking. These requirements can ensure that developers are more thoughtful when deciding the quantity of impervious space they want to introduce to an area [190].

Kaminker et al. [129] discuss how policy makers can create optimal conditions for investment in green infrastructure. The authors cite a shortage of objective information and data to evaluate infrastructure investments and their inherent risks. Investment can come from a variety of sources, but one of the authors' main foci is private investment. According to Hewes [102], covering as little as 1 percent of large buildings in mid- to large-sized cities in the United States with green roofs would create over 190,000 jobs, divert billions of gallons of stormwater from sewer systems, and provide billions of dollars in revenue to suppliers and manufacturers of materials related to green roofs. To encourage increased private participation in such GSI projects, barriers to implementation can be removed via education, incentives, social influences, and inclusive participation [229, 230]. Mere encouragement from policy to create GSI is often deficient motivation for developers to include GSI as part of their construction/retrofitting plans. The U.S. Environmental Protection Agency (EPA) insists that incentives and regulations ought to be used in tandem to cause widespread acceptance and implementation of GSI. Specifically, the EPA suggests that an incentive program can initiate GSI regulation through introducing design criteria and technology; consequently, private landowners will apply such designs, and later the municipality can introduce mandatory legislation [155]. Combining data collection, straightforward policy, and incentives can lead to greater investment in GSI.

2.2 Social Co-Benefits of Green Stormwater Infrastructure

Cities throughout the United States (and the globe) continue to grow in population and physical size, and thus they experience accompanying land use changes. Engineering systems simultaneously change in complexity and technological advancement. Stormwater management is one such system: in the early 20th century, engineers utilized 'gray' infrastructure (i.e., large, piped conveyance networks) to quickly and efficiently transport stormwater runoff out of cities. This "Promethean approach" satisfied the need to convey water away from urban cores, but it essentially took the problems associated with stormwater (e.g., flooding, water quality degradation, disease vector transport) and moved them downstream to the next community [131]. Continuing into modern times, increased urban and suburban development has led to problems such as degraded water quality, urban heat island effect, sewer system malfunction, and flooding [204]. Stormwater runoff is a leading contributor to water pollution throughout the United States [42, 177], and the growth of impervious surfaces increases storm runoff in regard to volume and peak flow [88, 255]. Developed areas incur changes in runoff temperatures and produce larger magnitudes of runoff from increased impervious surfaces [205]. Streams in developed areas are subject to increases in peak volume, discharge, and pollutant loading [161, 212, 214]. Furthermore, thermally-enriched runoff from impervious surfaces causes spikes in stream temperatures [127, 253] that can affect streams for up to 10 percent of summer days [178, 257]. Gray stormwater management offered a solution to storm runoff for the better part of the 20th century, but maintaining and expanding such gray systems cannot keep pace with the immense growth of cities.

Extensive research has described potential effects of human engagement with green spaces, including decreased stress levels [62, 130, 249], improved physical health through effects like decreased asthma [157], and enhanced mental health [227]. The presence of GSI has also been shown to decrease the occurrence of crime [19, 140]. A review by Velarde et al. [249] found that simply viewing natural landscapes has a positive impact on health. Although there are published studies showing such social co-benefits, findings can be highly site-specific, and the study methods (often surveys or behavioral observation) can be expensive to implement. And yet, the social co-benefits are nonetheless present, albeit sometimes challenging to quantify.

Lafortezza et al. [142] analyzed the intersection of physical and psychological well-being and interacting with green spaces in urban spaces and found that longer and more frequent visits to green spaces generated significant improvements in well-being among users. Their results supported the idea that green spaces can alleviate the suffering caused by periods of heat stress. Parks offer many co-benefits beyond hydrologic improvements, including physical well-being through activity [25, 158], as well as psychological and physiological benefits through reducing stress and lowering blood pressure [96, 170, 248]. Some studies have found that low-income and minority neighborhoods are located in areas with few or no parks, depraving those residents of the many benefits parks offer [2, 222, 259]. Exposure to green spaces has been linked to healthier cortisol levels [200] and reduced rates of attention-deficit and hyper-activity disorder (ADHD) in children [261]. The human benefit of green spaces for such children extends to economic benefits as well: Faber Taylor and Kuo [71] found that children with ADHD who took a 20-minute walk in a park had significantly better concentration than similar children who walked in an urban or residential setting. Browning et al. [30] calculated that this could save Americans nationwide up to $228 million in eliminated medication costs. Kuo [141] performed a multi-year study comprising of over 1,300 person-space observations in inner-city Chicago and found that trees and grass cover were linked to numerous social health indicators, including stronger ties among neighbors, greater sense of safety, healthier patterns of children's play, fewer property crimes, and fewer violent crimes. Browning et al. [30] quantified these social benefits and found that the

Illinois Department of Corrections could save up to $162,000 annually from reduced crime if the housing units in Kuo's study were outfitted with green spaces. The presence of myriad social co-benefits of GSI provide a motivation for the analyses in this thesis.

2.3 Environmental Justice

Historically, environmental justice was not considered during changes in land use and cover, leading to some marginalized communities disproportionately suffering environmental ills [123]. Often times, ethnic minority and low-income neighborhoods face such environmental injustice [28], which is only made worse by economic insecurity [184]. Examples include racial minorities subjected to higher levels of air pollution in Oakland, California [75], as well as impoverished neighborhoods having less access to green spaces in Milwaukee, Wisconsin [103]. Lejano and Iseki [145] found that the locations of hazardous waste treatment, storage, and disposal facilities are significantly correlated with Latino residential areas in Los Angeles. A two-year study of youth in Indiana found that greenness was inversely related to body mass index [18], a key implication for environmental justice given the lack of opportunities for physical activity in minority and low-income neighborhoods [77]. Many studies have shown that low-income households are more often exposed to harmful pollution from road transportation compared to higher income households [91, 92, 176]. Uneven social geographies have been shown to produce a phenomenon known as 'thermal inequity,' or a disproportionate impact of urban heating on lower-income and ethno-racially marginalized populations [168, 169]. Thermal inequity is based on two conditions: greater exposure to climate change related temperature increases and/or urban heat island effects and the lower socio-economic and/or ethno-racial minority status of that population. Instances of environmental injustice abound throughout the United States and elsewhere.

Although minority and low-income neighborhoods often bear the burden of environmental injustice, Baptiste et al. [13] suggest that people living in low-income communities are more willing to implement GSI. Such community ownership of GSI depends on public engagement during planning and installation of GSI [128], which does not always occur. New York has undertaken concerted efforts to provide green spaces in its low-income and visibly minority neighborhoods as a means of preventing the negative effects of high temperatures [202]. Going forward, involving marginalized communities in GSI planning and targeting benefits for such communities present important steps toward advancing environmental justice goals.

Ernstson [67] analyzed the ecological connectivity of all green spaces and GSI within a region, as well as a community's protective and management capacities, as one procedure to empower municipalities to further environmental justice. Analyzing whether more GSI is required in impoverished areas is necessary to better connect an entire regional complex of green spaces. Furthermore, fostering ownership of those spaces is an important factor in the success of environmental justice efforts. Several studies have shown that detailed knowledge of local communities, as well as continued involvement, is crucial for long-term success of GSI implementation [7, 134, 172, 218]. Baptiste et al. [13] concluded that it is important to understand the level of understanding of GSI among local residents. Environmental justice efforts can also be analyzed through a "value articulation" lens. Value articulation of GSI co-benefits features the scientists or experts providing substantiation for the articulated values, as well as local actors and artifacts (maps, scientific reports) that describe GSI and its values [68, 224]. In this context, empowering municipalities to choose the right agents of articulation and artifacts is an important portion of addressing social co-benefits.

Baptiste et al. [13] performed an extensive survey on two neighborhoods near Syracuse, New York, and found that GSI initiatives should be targeted for people who desire to improve the aesthetic of their neighborhood and personal space, people whose financial commitments will not be strained, and, significantly, low-income people. The positive health effects of GSI are shown in several studies to be highest among residents in lower socioeconomic groups [54, 170]. Focusing on such communities for investment in GSI provides part of the backdrop and motivation for this thesis.

CHAPTER 3

AGRICULTURAL GREEN INFRASTRUCTURE

3.1 The Issue of Nitrate Pollution

Nitrates pose a threat to both human and aquatic life. Consuming nitrates has been shown to lead to two major health issues in humans, namely infant methemoglobinemia (or "blue baby syndrome") and cancers of the digestive tract [90, 193]. People with weakened immune systems (babies, the elderly, and sick persons) are most susceptible to suffering sicknesses due to ingesting nitrate-polluted drinking water. Nitrates are also toxic to various fish species and can lead to degraded health and increased mortality rates [94, 111, 213], in addition to contributing to eutrophication that further harms wildlife [100].

Nitrate pollution levels in the RRW upstream of Des Moines, Iowa, USA (Figure 3.1) are some of the highest in the nation [208]. Water flowing from the RRW is used as drinking water for the Des Moines metropolitan area, an area serving more than 500,000 people. Nitrate levels were consistently high enough (continually above the federal 10 mg/L maximum contaminant level) to force DMWW, the region's drinking water utility, to construct the world's largest nitrate removal facility [208]. Nitrates are costly to remove from water; DMWW spends between \$4,000 and \$7,000 each day to operate its nitrate removal facility [46]. Hatfield et al. [98] showed that the nitrate issue in the RRW has worsened over time; since the 1970s, levels have increased, even with stagnant fertilizer use over the previous 15 years. Agricultural runoff is accountable for water quality violations in many other lakes, streams, and estuaries [60, 110, 182], and yet regulation of such runoff is not a viable policy solution. The Clean Water Act explicitly states in §502(14) that agricultural fields are exempt from federal permitting, meaning that farmers voluntarily introduce best management practices on their fields to reduce water pollution.

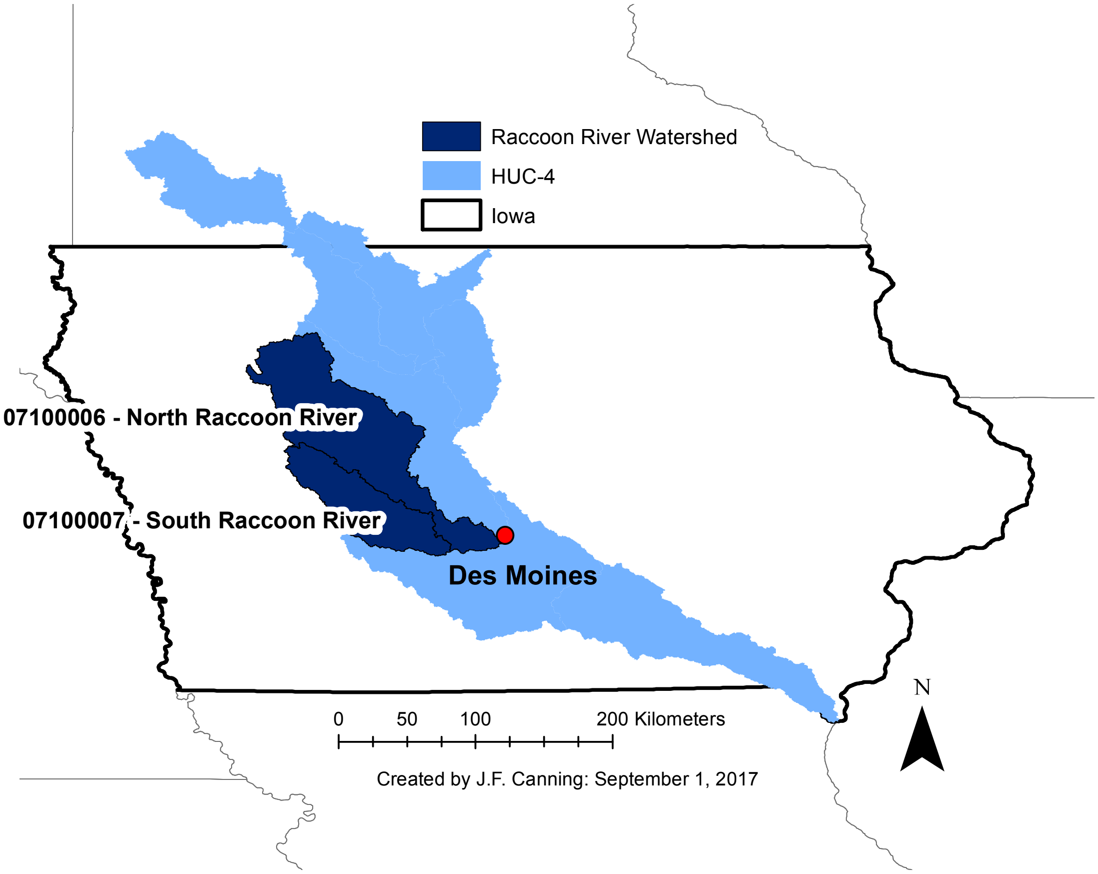


Figure 3.1: The Raccoon River Watershed (RRW), situated upstream of Des Moines, Iowa, USA, has some of the highest nitrate levels in the nation. Such high nitrate levels have led to negative environmental, economic, and social effects throughout the region and beyond.

Although DMWW operates a centralized nitrate removal facility, there are distributed solutions to preventing nitrate pollution. The idea of 'agricultural green infrastructure' is introduced in the form of riparian buffers. Agricultural green infrastructure functions similarly to its urban counterpart, infiltrating runoff and reducing the amount of pollutants reaching downstream. Its performance depends on topography, soil health, storm events, and other climatic conditions.

Nitrogen and phosphorus pollutants have contributed vastly to the growing Gulf of Mexico Hypoxia Zone, a dead zone that was measured at over 20,000 km^2 in 2008 [194]. A hypoxia zone is typified of low dissolved oxygen levels, caused by nitrogen and phosphorus, that place large amounts of stress on aquatic ecosystems [150]. The Gulf of Mexico Hypoxia Zone is the second largest on the planet after the Baltic Sea Hypoxia Zone [34, 151], and it represents an immense water quality issue that impacts wildlife, and consequently humans, throughout the Gulf region [49, 87]. Agriculture in the Midwest is a significant driver of Gulf Hypoxia; land covering over 50 percent of America's farms drains into the Mississippi River Basin. Reducing agricultural runoff and thereby water pollution would reduce the size and toxicity of the Gulf Hypoxia Zone, and such goals are the motivation for various nutrient load reduction programs throughout the 12 states comprising the Mississippi River Basin [e.g., 113, 118, 167]. In many of these states, both rural and urban communities are required to manage nutrients polluting their water. Many of those nutrients enter waterways through fertilizers. Nitrogen fertilizer use in the Midwest has increased drastically since the 1960s, remaining at a steady increase of 2.4 kg ha^{-1} $year^{-1}$ until the late 1990s, when average use stabilized at around 147 kg N/ha [59]. Swoboda [232] reported that in Iowa alone, farmers spent approximately $100 million on excess nitrogen fertilizer applications while not accounting for residual nitrogen in the soil. Such widespread fertilizer consumption has severely degraded water quality, and some sort of solution is necessary going forward.

One such solution is the use of riparian buffers. Riparian buffers, such as grass strips, riparian forests, or a combination of both, can mitigate water quality degradation, attenuate flows running off fields, and reduce erosion [38, 185]. Buffers can be established anywhere alongside a riparian corridor between a stream and the agricultural field (Figure 3.2). Hydrologically, buffer strips improve infiltration of water through the propagation of macropores throughout the soil [37, 147] and enhance water holding capacity by adding organic matter and increasing porosity [183]. Buffers have been shown to retain or remove nitrate by 60-90 percent [211, 216, 250]. Denitrification in particular is efficient under suboxic conditions with plenty of available organic carbon. In the winter and early spring seasons, when susceptibility to erosion and nutrient leaching is highest, Burt et al. [32] showed that the higher water table during these months creates the necessary oxic/anoxic conditions that, combined with carbon-rich soils enhanced by vegetated buffer strips, prevents nitrates from leaching into streams through the process of denitrification. Buffer design determines the rate at which vegetation retains nitrates and enhances denitrification.

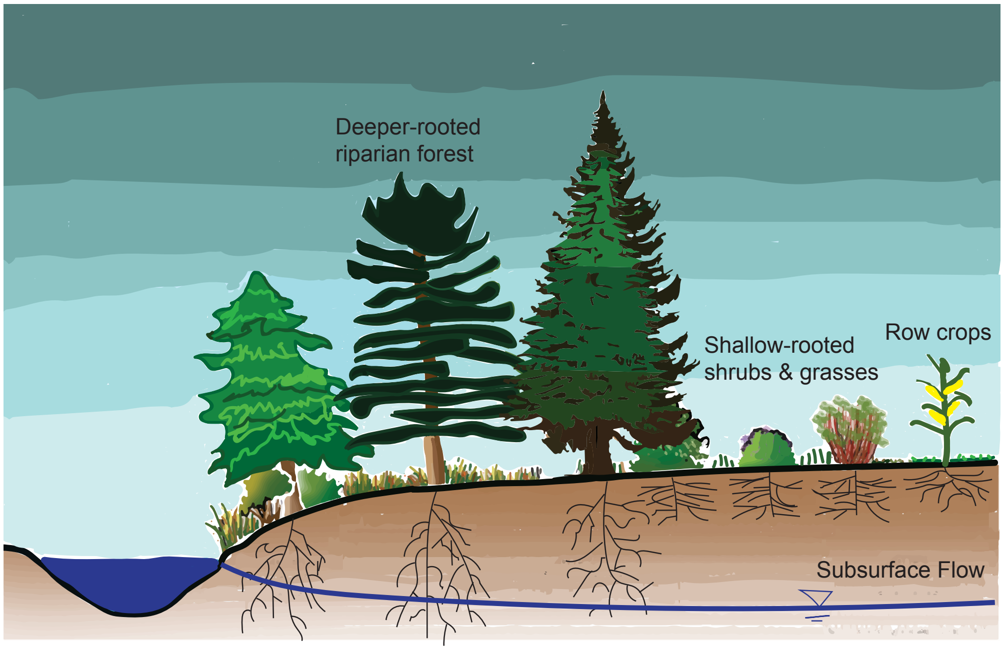


Figure 3.2: Multi species riparian buffers include trees, shrubs, and prairie grasses. Grasses attenuate surface runoff and filter out contaminants, while the deeper roots of the shrubs and trees uptake nutrients from groundwater levels.

Buffer strips are also designed to take up phosphorus. Phosphorus (P) is another contributor to eutrophic conditions that lead to hypoxia. P losses occur as particulates attached to soil sediments from field runoff and streambank erosion [57, 195]. Soluble P can also leach into groundwater [219]. Buffers prevent P losses through attenuating and infiltrating runoff, at which point particulate P is converted into biologically active P and is made available to plants and microorganisms [43]. In fact, the same conditions that are preferred for denitrification facilitate the process of converting particulate P to biologically active P [61]. Riparian buffers present a potential solution to P pollution, but every ditch and stream in agricultural watersheds cannot be lined with buffer strips. Terrain analyses are required to prioritize the location of buffer construction.

In the Midwestern United States, tile drainage exacerbates the problem of nutrient-laden water. Tile drains are located at some distance beneath the surface of a field and lower the water table by conveying excess water away from the field. However, after fertilizers are added to the soil and inorganic nitrogen undergoes nitrification, the resulting nitrates leach into groundwater and are transported via tile drains to surface waters. Tile drains contribute to water pollution, but solutions such as diverting flow to a riparian buffer can help decrease the amount of nutrients leaving fields [3, 101]. The 1992 National Resources Inventory database is the most recent and comprehensive report of tile drainage use at the U.S. county level [231], and for the 17 counties comprising the RRW, on average 51 percent of the land utilizes tile drainage [247]. It is difficult to assess the extent of tile drainage in the region, so the results of terrain analyses must be carefully inspected before deciding where riparian buffers are constructed.

3.1.1 Agricultural Green Infrastructure — Riparian Buffers

Riparian buffers are often most effective along first- and second-order streams [242], where a majority of the runoff in a watershed begins and can be targeted. Terrain analyses are utilized to determine where buffers might be most effective throughout the RRW. Terrain analyses determine topographically optimal locations with the most surface and subsurface flow where buffers will be most effective. Terrain analysis methods using geographic information systems (GIS) have proven effective at discerning fitting locations for buffers [45, 242]. Using the results from the analysis, a potential multi-species riparian buffer is designed that could be used throughout the RRW. Buffer performance is dependent upon factors in addition to topography, such as soil type, water table height, and vegetation type, and so the design can be amended as necessary. Using the principles behind water-quality trading (WQT), a partnership between DMWW and upstream agricultural operators is one possible solution to the immense nitrate problems the watershed faces. In total, three scenarios that DMWW (and Des Moines residents overall) could take to address nitrate pollution are investigated: S0) continuing business as usual, S1) constructing riparian buffers throughout the entire RRW, and S2) installing a new nitrate removal facility (Table 3.1). The new nitrate removal facility is labeled as a centralized, gray approach, and the riparian buffer solution as the distributed, green approach, using language similar to urban stormwater management. The economics of these potential solutions are analyzed with the additional goal of presenting an example for other communities throughout the Mississippi River Basin. Water quality is an increasingly important issue, and innovative solutions are required to create sustainable changes even while current policy cannot enforce such changes.

Table 3.1: Potential Des Moines Water Works Nitrate Control Solutions.

Scenario	**Description**
S0	Business as usual
S1	‘WQT’ Riparian Buffer Construction
S2	New Nitrate Removal Facility

Considering the plentiful benefits, the use of buffers is a viable option to manage nitrate levels throughout the United States. A buffer’s ability to uptake nutrients and protect water quality is dependent on its location within a watershed. Slope and soil conditions that lead to higher runoff and sediment generation represent better locations for buffers; these locations are typically alongside lower-order streams. Correll [45] found that most of the water moving into receiving water bodies enters through first- and second-order streams. Tomer et al. [242] further show that buffers function better by treating overland runoff and shallow groundwater along first- and second-order streams. These streams are often the most ideal for targeting water quality goals, and the analysis seeks to verify this finding through investigating first- and second-order streams in the RRW.

The RRW has been widely studied; Jha et al. [124] performed SWAT analyses (Soil and Water Assessment Tool, made available by United States Department of Agriculture (USDA) Agricultural Research Service), first calibrating the tool and then completing various sensitivity scenarios. The authors performed a sensitivity analysis in which they increased the amount of conservation land throughout the RRW, similar to the proposed installation of riparian buffers. Hatfield et al. [98] presented data from the past 70 years to show how cropping practices, fertilizer use, and land use changes for certain crops have led to changes in nitrate levels throughout the RRW. Schilling and Zhang [208] broke down nitrate concentration data in Iowa to determine the proportion of baseflow that contributes to nitrate levels. Tomer et al. [242] performed terrain analyses in Iowa, focusing on the Keg and Silver Creek watersheds. Thus, the gap in literature concerning terrain analyses of the RRW precipitates the necessity of the present analysis.

3.1.2 The Principles of Water-Quality Trading

Water-quality trading (WQT) represents an opportunity for municipalities, wastewater utilities, and other regulated dischargers to achieve regulatory water quality goals while allowing for flexibility in the level of technology employed. The U.S. Environmental Protection Agency (EPA) states that WQT is "an option for compliance with a water quality based effluent limitation in a National Pollution Discharge Elimination System (NPDES) permit" [66]. Regulated dischargers, most notably industrial and municipal wastewater treatment plants (WWTPs), need to meet water quality standards pertaining to the waterbody to which they discharge effluent. Total maximum daily loads (TMDLs) are assigned if a given waterbody continually fails to meet water quality standards. It can be difficult for WWTPs to meet TMDLs if agricultural runoff is a large contributor to the water quality problem. WQT, then, enables regulated dischargers to purchase agricultural nonpoint source credits. These agricultural nonpoint source credits often lead to adoption of best management practices (BMPs), such as cover crops, reduced tillage, and riparian buffers. Conceptually, regulated dischargers should be willing to purchase agricultural nonpoint source credits if the cost of implementing BMPs is less than treating the water on-site. Additionally, agricultural operators should be willing to sell credits if the price the regulated dischargers are willing to pay is greater than the credit production costs. Credit production costs include any verification costs, as well as BMP implementation costs [228]. It is important, however, to engage agricultural operators throughout the process concerning their positions on such environmental policies [206]. WQT has been cost-effective for WWTPs in agricultural watersheds [188], including the Chesapeake Bay [126, 132, 252]. WQT programs often focus on eutrophication issues associated with nutrients, specifically nitrogen and phosphorus [76, 89].

Using the WWTP as a precedent, the same concept is used to analyze the possibility of DMWW (a drinking water utility) "purchasing" agricultural nonpoint source credits through sponsoring installation of riparian buffers throughout the RRW. The proposal is that Des Moines residents, through DMWW, pay for buffer installations in appropriate locations upstream to reduce the cost of treating drinking water downstream. There is a similar precedent in water *quantity* trading; interbasin water transfers have occurred frequently throughout the history of the arid southwestern United States [24, 52]. Drinking water utilities, to the author's knowledge, have not previously utilized the ideas behind WQT to reduce the cost of removing nutrients from water. However, urban residents (through a drinking water utility) might encourage such a solution to reducing nutrient pollution in a river/lake/estuary, as their drinking water source. The economic feasibility of such an approach is explored in the RRW.

3.2 Materials and Methods

3.2.1 Terrain Analysis

Terrain analysis techniques put forth by Moore et al. [173] are used to determine factors such as upslope contributing area, slope, and catchment area that demonstrate portions of a watershed featuring water accumulation. Analysis of the RRW relies on two factors; slope (β, in degrees) and specific catchment area (A_s, units of m^2 m^{-1}). Slope determines the stream network and subbasins within the RRW, and slope also affects whether a location is fitting to readily remove nitrates from surface runoff or shallow groundwater flow. Specific catchment area is the upslope area that can potentially contribute surface runoff to a grid-cell location per width of flow. Calculating A_s for a raster of topography requires knowledge of the overland flow direction between adjacent cells. Thus, the D-∞ method put forth by Tarboton [236] using software published by D.G. Tarboton [235] is employed. Rather than the typical 8-direction model, the D-∞ method proportions flow as an angle in radians between 0 and 2π based on the steepest descent among eight triangular facets formed in a 3x3 cell block, with the cell of interest in the center (see Tarboton [236] for details).

After calculating β and A_s, compound hydrologic indices can be calculated to identify priority areas within the RRW. Topographic wetness index (TWI) is defined in Equation 3.1 and is the compound hydrologic index of interest.

$$TWI = ln(A_s/tan\beta) \tag{3.1}$$

TWI is used to demarcate areas most prone to soil saturation during precipitation events. The tanβ changes slope from degrees to more topographically useful units of rise over run (m/m). The natural log is used because the ratio (A_s/tanβ) spans many orders of magnitude over various landscapes. Areas with a large TWI are better situated for riparian buffers to remove nitrates. Specifically, flat areas with large upslope contributing areas potentially enable buffers to uptake nitrates from shallow groundwater flow and filter surface runoff. Flat areas at the bottom of hillslopes present opportunities for buffers to filter surface runoff, and flat riparian areas often feature shallow groundwater flow. Permanent vegetation can benefit water quality in all of these potential situations. Typically, riparian forest buffers function well for removing nutrients from shallow groundwater flow, while grass strips effectively filter contaminants from surface runoff [189, 209]. Similar terrain analyses have been applied to determine priority stream reaches [31] and riparian zones for field-level planning [241].

Raster datasets of the RRW were retrieved from the National Map. The National Map [246] is a comprehensive bare earth elevation dataset of North America at a 30-meter scale and provides freely available digital elevation models (DEMs). DEMs for the area surrounding the RRW were collected and clipped to the RRW boundaries using watershed delineations found through the National Hydrography Dataset. Two 8-digit Hydrologic Unit Code regions comprise the RRW: the North Raccoon (07100006) and the South Raccoon (07100007).

3.2.2 Buffer Design and Cost Analysis

A sustainable approach to nitrate pollution throughout the RRW takes into consideration the economic costs alongside the environmental benefits. A cost analysis is performed for the distributed approach of agricultural green infrastructure. Equation 3.2 presents the total cost (C_T) and includes vegetation planting (V), crop opportunity costs (O), and maintenance/labor costs (ML).

$$C_T = V + ML + O \tag{3.2}$$

The buffer design, summarized in Figure 3.3, determines vegetation costs. Figure 3.3 illustrates what is called a "buffer unit," i.e., what comprises a 20 m x 2 m area of buffer. There are two species of trees (Jack pine, *Pinus banksiana*, and eastern red cedar, *Juniperus virginiana*), two species of bushes (redosier dogwood, *Cornus stolonifera*, and ninebark, *Physocarpus opulifolius*), and one species of tall prairie grass (switchgrass, *Panicum virgatum*). All are native to Iowa, and the tree/shrub saplings can be purchased through the Iowa Department of Natural Resources state forest nursery [117]. Jack pine and eastern red cedar are chosen for tree species because both are evergreens and both grow at a rapid pace in full sun. Evergreen trees are preferable over deciduous trees because of the potential for deciduous trees to eventually contribute to water pollution after litterfall [152]. Redosier dogwood and ninebark are chosen because of their rapid growth rate and ability to grow in full sun. Buffer width is significant, as it determines nitrate removal efficiency. Based off a meta-analysis of riparian buffer studies [159], a 20 meter wide buffer system is chosen. A 20 meter wide multi-species riparian buffer can be expected to remove about 75 percent of nitrates coming off agricultural fields [159]. Nitrate levels as high as 30.5 mg/L have been recorded at USGS Gage 05482500, North Raccoon River near Jefferson, Iowa. Even at such high levels, assuming a fully effective riparian buffer, nitrate levels would be reduced to below the 10 mg/L federal drinking water limit set by the EPA [245]. Using a 20 meter wide buffer balances the tradeoff between nitrate removal and total cost.

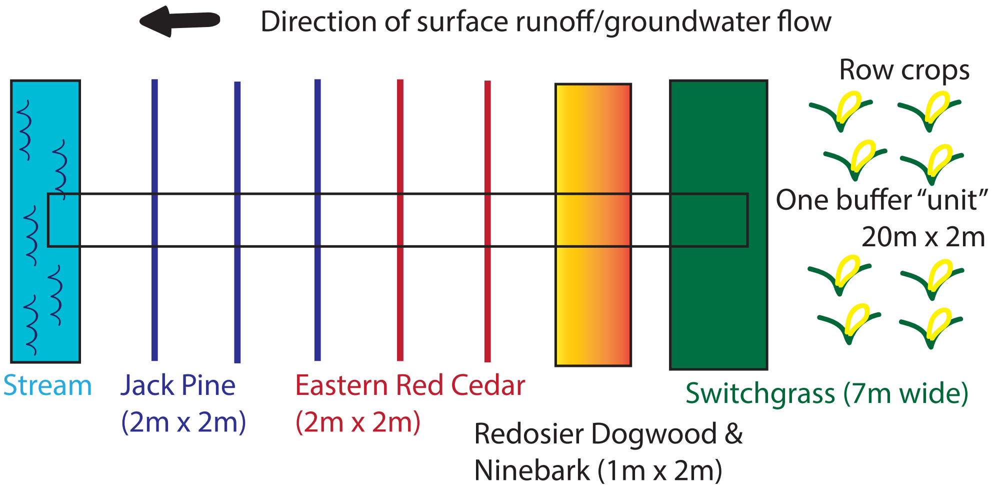


Figure 3.3: Multi-species riparian buffer design with spacing dimensions. One 'unit' represents three parallel Jack pines, two parallel eastern red cedars, one redosier dogwood bush, one ninebark bush, and a 7m x 2m plot of switchgrass.

Net present vegetation costs are calculated from three years' of planting data, discounted at a rate of 5 percent. The underlying assumption is a 60-70 percent tree/shrub survival rate from year to year, a conservative approach (see, e.g., [215]). The cost of trees and shrubs are sourced from the Iowa Department of Natural Resources seedling catalog. Trees require a shelter and accompanying stake upon planting, and thus these costs are included and based on a sample manufacturer, Forestry Suppliers [78]. Switchgrass costs are based on the cost of seed; Equation 3.3 presents the cost of one hectare of switchgrass, where 'PLS' indicates 'pure live seed,' the term used to describe switchgrass seed. The cost of switchgrass seed comes from the Osenbaugh Prairie Seed Farms in Lucas, Iowa [186], and the amount of seed comes from a study published by West and Kincer [254] that determined 6.72 kg PLS ha^{-1} grew a sufficient amount of switchgrass. Vegetation costs are presented in Table 3.2

$$\frac{\$7.50}{\text{PLS lb}} \cdot \frac{\text{lb}}{0.454\text{ kg}} \cdot \frac{6.72\text{ kg PLS}}{1\text{ hectare}} = \$111/ha \tag{3.3}$$

Table 3.2: Vegetation material costs per hectare for the first three years. Ranges represent the amount of replanting necessary from year to year.

Item	**Quantity**	**Initial Cost**	**Replanting, Years 2&3**	**Tree Shelter, 3 years**	**Net Present Cost, 3 years**
Eastern Red Cedar	500	\$500	\$150–\$200	\$1,129–\$1,270	\$1,908–\$2,142
Jack Pine	750	\$413	\$124–\$165	\$1,694–\$1,906	\$2,336–\$2,625
Redosier Dogwood	250	\$225	\$68–\$90	N/A	\$351–\$392
Ninebark	250	\$225	\$68–\$90	N/A	\$351–\$392
Switchgrass	1	\$111	N/A	N/A	\$111

Maintenance costs cover several activities and materials, including herbicide treatment and mowing between rows of vegetation. Mowing is necessary to reduce competition between inevitably-occurring weeds and the newly planted trees and bushes. Herbicides are also recommended to reduce such competition. These two maintenance techniques are recommended by the Iowa State University (ISU) Extension with corresponding maintenance cost data [119]. Accompanying such maintenance costs are the labor costs associated with preparing the land for buffer placement, planting vegetation, and maintaining the buffers. According to the Bureau of Labor and Statistics 2016 data for the state of Iowa, forest and conservation workers (occupation code 45-4001) earned a mean hourly salary of \$15.86. Using the ISU Extension labor data to estimate the time required for various activities, labor costs per hectare of buffer installation are calculated (Table 3.3).

Table 3.3: Maintenance activities and their associated labor costs for buffer installation, using the labor rate from the Iowa State University Extension, for an entire year. Ranges represent the amount of labor required based on replanting rates.

Site Prep (hr/ha)	5
Vegetation Planting (hr/ha)	40–50
Site Maintenance (hr/ha)	3.75–7.5
Mowing (hr/ha)	3.75
Labor Rate (\$/hr)	\$15.86
Herbicide Costs (\$/ha)	\$39–\$42
Total Labor (\$/ha)	**\$871–\$1,093**

The final category included in the riparian buffer cost analysis is labeled “opportunity costs.” Payments to farmers for use of their land as well as payments in recompense for lost crop growth (and thus, lost revenue) are included in opportunity costs. There are six counties that make up the RRW: Dallas, Guthrie, Greene, Carroll, Sac, and Calhoun. Yield, hectares planted, and market price data published by the USDA National Agriculture Statistics Service for the year 2016 are used to calculate the cost of taking corn/soybean crops out of production (Table 3.4). Upper and lower bounds represent the amount of land assumed to viably support crop growth. The conservative upper bound assumes all buffer land otherwise would have supported crops, and the realistic approach assumes 75 percent of buffer land would have supported crops (lower bound). Buffers would be placed on already marginal land, and thus, in practice, DMWW and the farmers could likely compromise on a payment between the two bounds. Final opportunity costs are an average between the six counties. Opportunity costs represent the largest portion of total costs for the riparian buffer scenario because of the fact that farmers are being asked to take land out of production, representing both an economic and cultural cost.

Table 3.4: Land lease and opportunity costs per hectare for RRW counties. Values are based on 2016 National Agriculture Statistics Service data, in addition to the land rent values.

County	Corn (ha)	Opportunity Cost—Corn	Soybean (ha)	Opportunity Cost—Soybean	2017 Rent	**Total Opportunity Cost ($/yr)**
Dallas	58,881	$1,825–$2,247	39,416	$1,586–$1,929	$558	**$1,729–$2,120**
Guthrie	44,919	$1,716–$2,126	36,381	$1,515–$1,857	$487	**$1,626–$2,018**
Greene	72,842	$1,840–$2,266	47,347	$1,676–$2,048	$561	**$1,775–$2,179**
Carroll	79,115	$1,873–$2,293	43,503	$1,641–$1,983	$613	**$1,791–$2,169**
Sac	68,795	$1,932–$2,374	52,001	$1,718–$2,090	$603	**$1,840–$2,260**
Calhoun	74,461	$1,866–$2,294	51,596	$1,678–$2,044	$581	**$1,789–$2,194**

3.2.3 Nitrate Removal Facility Costs

In the second proposed solution to the nitrate management issue, costs required for DMWW to operate its nitrate removal facility, as well as construct a new treatment facility altogether, are gathered. A recently-passed Five-Year Capital Plan [55, 165] includes $70 million earmarked specifically for nitrate removal measures, with an additional $10 million in the sixth year. Net present cost of such expenditures is calculated using a 5 percent discount rate, similar to the distributed, green approach. DMWW CEO Bill Stowe has stated that the utility company prefers to continually repair and maintain the current nitrate removal facility over constructing a new one [165]. A new facility could cost up to $184 million [46], a tremendous burden on Des Moines residents. However, such a project would include high costs initially but reduced maintenance costs over time. The $184 million estimate is used as the upper bound on the second scenario.

3.3 Results

Terrain analyses confirmed that first- and second-order streams are most suitable for buffer placement within the RRW. Figure 3.4 presents TWI for the extent of the RRW, and Table 3.5 gives the results for each stream order. A Student's t-test is performed to determine which stream orders are significantly different from one another, and the results indicate that first-order streams have significantly ($p < 0.05$) higher TWI values compared to higher-order streams. This finding agrees with other studies analyzing optimal buffer placement [9, 45]. Consequently, first-order streams in the RRW are focused as prime locations for buffer installation under the agricultural green infrastructure approach.

Table 3.5: Topographic wetness index results for the Raccoon River Watershed indicate that first-order streams are most suitable for buffer location based on topography.

Stream Order	Maximum	Range	Mean	Standard Deviation
1	17.9	16.1	5.41	1.93
2	16.3	14.8	5.37	2.00
3	17.5	15.6	5.27	1.98
4	19.2	17.5	5.08	1.92
5	20.4	18.8	4.94	1.82
6	20.8	19.1	4.69	1.72
7	22.7	21.2	4.71	1.67
8	23.2	21.6	4.59	1.71

Table 3.6 summarizes the total costs per hectare of riparian buffer installation, and Table 3.7 presents overall costs for each of the scenarios. The business as usual scenario, S0, is not feasible because the current insufficient nitrate removal facility will eventually become defunct without renovation or replacement; not meeting drinking water nitrate standards is not allowed by law. Final costs for the distributed, green approach are estimated between \$135-\$160 million. Again, the proposal is a partnership between DMWW and upstream agricultural operators, based off the principles of WQT, wherein Des Moines residents are paying for these costs over the period of time required to finance the buffer installation

project. These numbers represent a conservative approach in which agricultural green infrastructure in the form of riparian buffers are installed along every first-order stream. In total, there are about 10,700 hectares of riparian buffer, and the upper and lower bounds represent the amount of replanting required (and accompanying labor/maintenance costs). The centralized, gray approach, has wide monetary bounds due to the different approaches DMWW might take. The lower bound ($71 million) represents money already earmarked for updates and maintenance on the nitrate removal facility [55]. DMWW already anticipates passing these costs to ratepayers. The upper bound ($184 million) is the estimated cost of a completely new nitrate removal facility [46]. Constructing a new facility is the worst-case scenario economically, as DMWW would require financing, the cost of which would eventually be transferred to ratepayers.

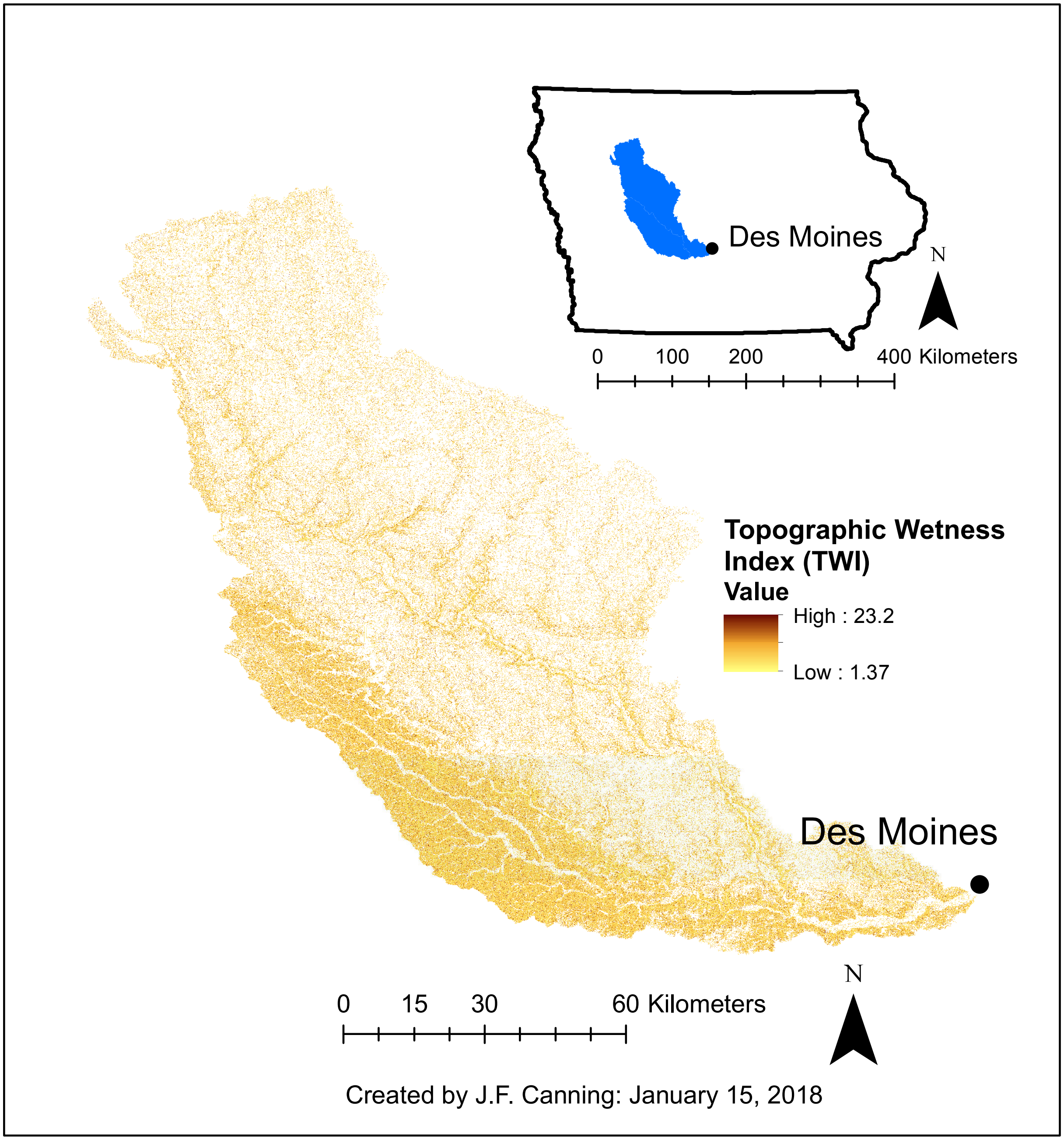


Figure 3.4: The resulting TWI map. A higher TWI represents land more suitable for riparian buffer placement.

Table 3.6: Total net present cost of the agricultural green infrastructure approach over three year period per hectare.

Vegetation Costs	$ 5,071	–	$ 5,678
Labor Costs	$ 2,491	–	$ 3,126
Opportunity Costs	$ 5,028	–	$ 6,167
Net Present Cost	**$ 12,590**	**–**	**$ 14,970**

Table 3.7: Scenario analysis results show that water-quality trading (WQT) can be a cost competitive approach to nitrate control through agricultural green infrastructure.

Scenario	**Description**	**Cost ($ million)**	**Satisfies EPA Requirements?**
S0	Business as usual	N/A	No — illegal
S1	WQT Riparian Buffers	$ 135–$ 160	Yes — nitrate levels below 10 mg/L
S2	Nitrate Removal Facility	$ 71–$ 184	Yes — nitrate levels below 10 mg/L

3.4 Discussion

3.4.1 Policy Implications — Current Policy and Shortcomings

Terrain analysis results can lead to improved water quality if riparian buffers are employed throughout the RRW; however, implementing buffers is currently left to farmers and other riparian property owners to do so voluntarily. Buffers are often supported financially and technologically by USDA conservation reserve programs, but there are no regulations requiring farmers to install buffers. Researchers have scrutinized whether voluntary conservation programs have environmental benefits that actually outweigh the cost of taxpayer dollars. Several studies investigated the cost-benefit relationship of agri-environment programs [41, 72]. Claassen et al. [41] found that both the Conservation Reserve Program (CRP) and Environmental Quality Incentive Program (EQIP), two of USDA's largest conservation programs, have much room for improvement; i.e., the deficit between benefits and costs can be eliminated and then reversed. The authors state that targeting payments toward performance of conservation measures (instead of mere practice) can increase benefits, and costs could be pushed down through encouraging more intense competition for payments [41]. Quantification of benefits is crucial for informing policy, and so studies that detail benefits are important for improving voluntary conservation programs.

Various authors have analyzed whether source water protection is a viable, cost-effective program. In 1986, Shortle and Dunn [220] compared policy measures that could be used to improve water quality conditions hampered by agricultural runoff. The authors found that economic incentives (e.g., a tax on soil loss) placed on farm management practices were the most practical and cost-effective measures that could be implemented. A small portfolio of incremental taxes for certain management practices would encourage farmers to adopt practices that maximize social environmental benefits. Three decades have passed since Shortle and Dunn [220], along with Harrington et al. [95], Gianessi and Peskin [81], and Libby [148], recommended *economic incentives* change behaviors and improve the quality of source water rather than voluntary participation.

In 2014, USDA introduced the Source Water Protection Program (SWPP) with a goal of helping communities, especially rural communities, develop a plan to decrease source water pollution risks. The SWPP does not 1) regulate agricultural runoff; 2) require that farmers comply with runoff standards; or 3) levy taxes on farmers for creating polluted waters. The Clean Water Act prevents SWPP, CRP, EQIP, and other programs from enforcing any sort of regulation restricting the runoff flowing off agricultural fields (as nonpoint sources), subsequently affecting rural communities. Despite long-standing research recommendations to create economic incentives to deter source water pollution, policymakers have been unable to improve upon voluntary conservation measures. Source water pollution became a serious issue for the city of Des Moines, Iowa, USA, and led to potential changes in how agricultural runoff might be addressed.

3.4.2 DMWW Lawsuit and Long-Term Policy Significance

Polluted drinking water has become an increasingly divisive issue in Des Moines. On March 16, 2015, DMWW filed a lawsuit (*Bd. of Water Works Trs. of Des Moines v. Sac Cty. Bd. of Supervisors as Trs. of Drainage Dists. 32, 42, 65, 79, 81, 83, 86, No. 5:15-cv-04020, 2015 WL 1191173*) against drainage districts in three Iowa counties (Buena Vista, Calhoun, and Sac). DMWW alleged that the drainage districts, located upstream of Des Moines and draining into the RRW, illegally discharge polluted water from a point source, i.e., tile drains. DMWW contends that the runoff is indeed a point source and thus liable to the full regulatory requirements of the Clean Water Act [197]. DMWW has been operating a nitrate removal facility since 1991, and costs to remove nitrates have risen to between \$4,000 – \$7,000 per day [46]. If elevated nitrate levels continue, the utility will need to construct a new nitrate removal facility costing taxpayers between \$76 million and \$183.5 million [46]. There are 77 entities that discharge into the Raccoon River and must hold National Pollution Discharge Elimination System permits, which require nitrate levels below 10 mg/L of effluent [198]. DMWW argued that the drainage districts must also remain below this nitrate threshold.

In January and March of 2017, after two years of deliberation, the final rulings in the DMWW lawsuit came from the Iowa Supreme Court and a federal judge, respectively. The former stated that the drainage districts were not responsible for costs incurred due to high nitrate levels, declaring the drainage districts would not pay the \$1 million DMWW was demanding. The federal judge argued that drainage districts lack the regulatory powers bestowed on other governmental entities, and furthermore, they were not depriving DMWW of the *water*, a common good, in the Raccoon River [23]. The judge abstained from ruling whether tile drains can be regulated as a point source. Thus, DMWW and other water quality stakeholders were forced to resort to encouraging voluntary conservation efforts and awaiting legislative action pertaining to nutrient pollution. Even though the courts decided the drainage districts are exempt from regulation and from paying for nitrate removal, the lawsuit could still have its desired effect. For example, the EPA could hand down a consent decree that forces a state to develop stricter regulations on agricultural runoff [46]. Already, there are 12 Midwestern states that have developed nutrient reduction strategies for this very purpose [e.g., 113, 118, 167], but such strategies do not have direct regulatory power. The recommended best management practices are left up to farmers to voluntarily adopt. However, voluntary conservation efforts can be ineffective, and thus another approach might be needed for water quality stakeholders. Water-quality trading is one such approach.

3.4.3 Comparing the Gray and Green Approaches

The RRW has an increasing water quality problem. DMWW unsuccessfully attempted to force farmers to pay to treat the water flowing off their fields, and the paltry effectiveness of USDA conservation programs indicates that novel physical and policy solutions are required. A distributed, green approach — widespread riparian buffer installation — would indeed cost DMWW (i.e., ratepayers) tens of millions of dollars more than continuing to maintain the nitrate removal facility. However, there are many benefits to the distributed, green approach compared to the centralized, gray approach. Construction and maintenance of the buffer network would require hundreds of hours of human labor, giving people living-wage and accessible employment. Such a social co-benefit pointedly addresses the triple-bottom line assessment of sustainability. Constructing a new nitrate removal facility would also require increased engineering and manufacturing jobs, an unquantified benefit of that scenario. However, such jobs might not be as accessible to unemployed/underemployed lower skilled laborers seeking work. Further benefits include carbon sequestration [187] and wildlife habitat due to the vegetation [51]. Once established, the trees would provide cooling effects throughout the stream network, providing further water quality benefits [174]. These benefits are not quantified in the present analysis, but installation of buffers offers more than reduction of nitrates in drinking water. Economically and socially, although buffer installation costs more than maintaining the present nitrate removal facility, the former offers various additional environmental and social benefits.

Agricultural green infrastructure is indeed more expensive than maintaining and updating the current nitrate removal facility, but building out riparian buffers could avoid the extravagant costs associated with building a new nitrate removal facility. DMWW already has committed \$71 million in maintaining the current facilities. At some point, increased investment will be necessary to at least renovate the facility, especially if nitrate levels continue to rise. Building a new facility might not be necessary in the near future given such investment in renovation; however, the facility cannot endure into perpetuity. A new facility will most likely be necessary at some point, at which time costs will almost certainly be higher. Furthermore, maintaining and/or building the current/new facility will accrue no environmental or social benefits. It might be more expensive at the current moment to install widespread riparian buffers, but such a project helps avoid the need to invest heavily in a new facility in the medium- to long-term future.

Both the centralized, gray approach and the distributed, green approach fulfill the requirement that DMWW reduce nitrate levels below the federal standard. However, the distributed, green approach might also help to alleviate an issue that appeared during the DMWW litigation: the rural-urban disconnect. The U.S. Water Alliance recently published seven ideas to solve America's water crises, and at the top of the list is accelerating "agriculture-utility partnerships to improve water quality" [70]. Water-quality trading through the construction of riparian buffers allows the wealthy and resource-intensive urban residents to sponsor projects in rural communities that provide clean drinking water. Rural residents are fairly compensated for use of their land, and there are other environmental and social/economic benefits for both communities. WQT enables a compromise that offers benefits to both urban and rural residents, benefits that the lawsuit could not offer to both parties.

Additional considerations beyond the present analysis include determining where tile drainage is used throughout the RRW and incorporating such drainage into the multi-species riparian buffer design. Buffers do not reduce nutrient pollution if tile drains are installed; however, there are potential solutions available. One such suggested solution is the use of constructed wetlands at certain intervals along a field with tile drainage [104, 192]. Constructed wetlands are similar to riparian buffers in that they feature vegetation that will uptake nutrients running off agricultural fields; their defining feature, however, is the anoxic conditions they create, allowing for denitrification to occur. Constructed wetlands are placed between the tile drain outlet and the recipient stream, thus acting to purify surface runoff and groundwater before it enters the stream. To reduce the problem of flow bypassing riparian buffers, Schultz et al. [215] recommended that one hectare of wetland be constructed at the field outlet for every 100 hectares of tile-drained agricultural field. Using the distributed, green approach in the RRW would require field-level analysis to determine whether tile drainage is present and, if so, where to install constructed wetlands.

CHAPTER 4

SOCIOECONOMIC ANALYSIS

As was evident in the agricultural green infrastructure analysis, comparing costs between various GSI projects and traditional gray infrastructure improves with the inclusion of economic and social impacts [258]. In an urban context, specifically quantifying co-benefits and encouraging municipalities to continually monitor and adjust GSI projects as an "urban experiment" [29, 36] adds to the store of data and information that investors demand. Providing decision makers with a framework that quantifies economic, environmental, and social costs and benefits can help alleviate potential investors' risks [129].

Analyzing social co-benefits can better equip decision makers in choosing how to design and construct GSI, depending not only on topographic or hydrologic factors but also human and social factors. The primary goals of GSI are typically environmental, i.e., hydrologic benefits, and economic/social co-benefits are an additional bonus. Many available stormwater calculators, featuring options to include GSI in their analyses, present the various hydrologic benefits that GSI offers without including details on potential economic or social co-benefits. This exclusion could partly be due to the nature of economic and social co-benefits — they are difficult to quantify and even more difficult to generalize across the various scenarios analyzed with stormwater calculators. However, municipal and regional decision makers strive to act as good stewards of the public's tax money, and thus these leaders seek out some sort of analysis pertaining to potential co-benefits of implementing GSI throughout their communities.

4.1 Stormwater Calculator Analysis

4.1.1 Methodology

Morales-Torres et al. [175] reviewed 11 decision support tools that analyze the costs and benefits of various forms of GSI, finding them to be lacking in presenting the entire breadth of benefits in the decision making process. The authors also created their own decision support tool, E^{2}STORMED. Throughout the publication, Morales-Torres et al. [175] focus on hydrologic benefits, energy consumption, and costs estimation; they do not hone in on the potential social benefits of the various decision support tools. Jayasooriya and Ng [122] compared 20 different GSI modeling tools, reviewing which GSI practices are supported (i.e., rain gardens, permeable pavement, and so on) and the purpose of such modeling tools. The authors make mention of the need for quantifying social benefits, but they do not analyze whether the tools themselves present social co-benefits. Available literature does not address whether or not stormwater calculators and tools avail social co-benefits to users. Value is added to the greater body of knowledge concerning GSI through the analysis of social co-benefits in these calculators and tools.

This portion of the analysis critically assesses 21 different stormwater calculators and modeling tools with the goal of determining to what extent each presents potential social co-benefits of GSI. When applicable, the calculator/tool was downloaded to discern any indication of social co-benefits. Otherwise, manuals, reports, and publications were analyzed for any presence of social co-benefits. A variety of potential social co-benefits was sought out, including improved mental health, recreation, decreased anxiety/stress, improved emotional health, reduced crime, and increased human engagement of the natural environment. Calculators and tools that include one or more social co-benefits are discussed in detail. A framework is then presented for amending the paradigm in GSI planning to incorporate a heightened awareness of social co-benefits.

Currently-available tools often highlight one particular region. These tools certainly offer valuable insight into the regions of origin (e.g., Tucson [115], New York [180], and St. Louis [166], among others) but are less applicable outside that region. Furthermore, current tools frequently cannot function across multiple spatial-temporal scales; i.e., they provide GSI benefits at the site-scale but not at a watershed-scale. The scale on which such calculators attempt to include social co-benefits is noted.

4.1.2 Results

Table 4.1 presents the results of the stormwater calculator analysis. Of the 21 GSI calculators, frameworks, and tools being analyzed, five have some sort of mention of social co-benefits, but only two attempt to quantify social co-benefits. The five tools acknowledging social co-benefits are the Center for Neighborhood Technology (CNT) Green Values National Stormwater Management Calculator (CNT Green Calculator), the Victoria Institute of Strategic Economic Studies (VISES) Green Paper, the CNT "The Value of Green Infrastructure" Framework (CNT GI Framework), the Stantec Evaluation of GI/LID Benefits in the Pima County Environment (Stantec Evaluation), and the Mersey Forest GI Valuation Toolkit (Mersey Toolkit). The CNT GI Framework and Mersey Toolkit go further and provide methods to quantify some social co-benefits. The other 16 calculators, frameworks, and tools all focus primarily on hydrologic benefits of GSI, including water quantity and quality improvements. While these tools are certainly useful for planning the hydrologic and environmental impacts of GSI installation, they would not be helpful in accounting for social co-benefits.

Table 4.1: GSI calculators and frameworks analyzed for the inclusion of social co-benefits.

Calculator/Framework	References and Case Studies	Social Co-Benefits Comments
1) Center for Neighborhood Technology (CNT) Green Values National Stormwater Management Calculator	Guo and Correa [93], Jaffe [120], Jaffe et al. [121], Kennedy et al. [135], Wise et al. [258]	Much greater focus on hydrologic and environmental benefits; mentions “aesthetic,” “public health,” and “recreation,” but does not provide any details
2) Victoria Institute of Strategic Economic Studies (VISES) Green Paper	Bowen and Lynch [27]	Describes physical, mental, and spiritual health benefits and community benefits and acknowledges difficulty in quantifying; discusses concept of social discounting
3) Environmental Valuation Reference Inventory (EVRI)	Bergstrom and Civita [20], McComb et al. [160], Villa et al. [251]	Contains broad database of publications concerning valuation of environmental assets; can search for “social benefits” and view publications, but does not provide comprehensive overview of benefits (up to user)
4) Environmental Protection Agency (EPA) National Stormwater Calculator	Beck et al. [16], Kertesz et al. [136]	Details hydrologic benefits to GSI implementation, but does not include social co-benefits
5) i-Tree Eco (v6)	Andrew and Slater [8], Cabaraban et al. [33], Rogers et al. [201]	Analyzes scenarios to produce hydrologic and climatic benefits, but does not contain any social co-benefits of tree planting
6) CNT ”The Value of Green Infrastructure” Framework	Gallet [80]	Outlines four social co-benefits; provides quantification methods for property value increases due to trees and recreational benefits
7) New York Department of Environmental Protection (NYDEP) Stormwater System Calculator	New York City Department of Environmental Protection [179]	Models GSI implementation and provides hydrologic benefits but no social co-benefits

Continued on next page

Table 4.1 **continued**

Calculator/Framework	References and Case Studies	Social Co-Benefits Comments
8) Metropolitan St. Louis Sewer District (MSD) Maximum Extent Practicable (MEP) Spreadsheets	Hoskins [108], Hoskins et al. [109]	Models runoff reductions from GSI installation but does not include any economic or social co-benefits
9) Stantec Evaluation of GI/LID Benefits in the Pima County Environment	Impact Infrastructure, LLC and Stantec [115]	Uses Institute of Sustainable Infrastructure's Envision program to outline potential "quality of life" improvements with purpose of earning Envision credits; no specific details on quantification of "quality of life" improvements
10) EPA Storm Water Management Model (SWMM)	Chui et al. [39], Huber et al. [112], Khader and Montalto [138], Rossman [203], Tsihrintzis and Hamid [243]	Extensively used to model runoff scenarios in urban areas with regard to reducing urban runoff; however, does not address social co-benefits
11) Virginia Runoff Reduction Method (VRRM)	Battiata et al. [15], Bork and Franklin [26]	Contains water quality and water quantity hydrologic/environmental benefits, but does not address social co-benefits
12) Water Environment Research Foundation (WERF) BMP Systems Effectiveness and Life Cycle Cost Evaluation (SELECT) Model	Reynolds et al. [199]	Provides hydrologic benefits and estimates cost of implementing GSI, but does not include social co-benefits
13) City of Chicago Department of Water Management Stormwater Spreadsheet Tool	Emanuel and Powers [63]	Only presents volume reduction due to GSI implementation; does not contain economic or social co-benefits
14) Delaware Urban Runoff Management Model (DURMM)	Balascio and Lucas [12], Lucas [153, 154]	No mention of social co-benefits, only hydrologic/environmental benefits

Continued on next page

Table 4.1 **continued**

Calculator/Framework	References and Case Studies	Social Co-Benefits Comments
15) Stormwater Investment Strategy Evaluator (StormWISE) Model	McGarity [162, 163, 164], Sebti et al. [217]	Quantifies water quality and quantity benefits of GSI but no social co-benefits
16) The Mersey Forest GI Valuation Toolkit	Finlay [73], Gill et al. [82], Horizons, Cambridgeshire [107]	Robust environmental, economic, and social co-benefit analysis; provides quantification of numerous social co-benefits
17) EPA System for Urban Stormwater Treatment and Analysis Integration (SUSTAIN)	Lai et al. [143], Lee et al. [144]	Estimates the costs associated with GSI and provides runoff reduction results, but does not provide social co-benefits
18) Wisconsin Department of Natural Resources RECARGA	Atchison et al. [11], Dietz [58]	Refined modeling of GSI and associated runoff and hydrologic impacts, but does not include social co-benefits
19) Model for Urban Stormwater Improvement Conceptualisation (MUSIC)	Cooperative Research Centre for Catchment Hydrology, Melbourne, Australia [44], Wong et al. [264, 265]	Proprietary modeling tool yielding hydrologic benefits of GSI and cost analyses; does not provide social co-benefits
20) Low Impact Development Rapid Assessment (LIDRA)	Aguayo et al. [4], Behr and Montalto [17], Montalto et al. [171], Yu et al. [267]	Quantifies runoff reduction and estimates costs, but contains no quantification of social co-benefits
21) Long-Term Hydrologic Impact Assessment (L-THIA)	Bhaduri et al. [22], Engel et al. [65], Liu et al. [149], Tang et al. [234]	Yields runoff reduction and nonpoint source pollution results, but does not make any mention of social co-benefits

The CNT Green Calculator was created with the purpose of evaluating the performance, cost, and benefits of GSI and comparing to conventional stormwater management. Many studies have utilized the CNT Green Calculator, especially for its capability to estimate the cost comparison between gray and green stormwater infrastructure [93, 120, 121, 135, 258]. The CNT Green Calculator includes several potential social co-benefits in its analysis, including aesthetic, public health, recreation, shelter, and sound absorption. However, no method of quantifying such social co-benefits is presented, and furthermore, the aesthetic and shelter categories do not cite any sources. Any proposed project evaluated with the Green Calculator will yield hydrologic, environmental, and economic benefits, but the tool will not quantify social co-benefits for the user. The CNT Green Calculator indeed makes users aware of potential social co-benefits but only if the user investigates the benefits fact sheet.

The VISES Green Paper and Stantec Evaluation likewise present potential social co-benefits without methods of quantifying those benefits. The VISES Green Paper goes into great detail about potential social co-benefits at the individual, community, and institutional levels. The authors further discuss "social discounting," or the practice of creating a present value for future long-term social benefits like recreation or improved physical and mental health. The VISES Green Paper does present quantified social co-benefits of several case studies; however, these numbers actually came from the Mersey Toolkit. The Stantec Evaluation uses a proprietary calculator called AutoCASE^{TM}, created by Impact Infrastructure, LLC [114], as well as the Envision^{TM} rating system [116]. AutoCASE^{TM} does produce detailed social co-benefits, such as reduced heat stress mortality for the Pima County case study. Because it is proprietary, however, the methodology is not presented, and current costs between \$1,500–\$25,000 prevent present analysis of the GSI tool. The Envision^{TM} rating system quantifies social co-benefits, but at the completion of a project, as is the case with the Stantec Evaluation. The report uses Envision^{TM} to corroborate how GSI can lead to social co-benefits. Overall, the Stantec Evaluation presents robust social co-benefits for GSI implementation in Pima County, Arizona, but the methodology is relatively opaque and not transferable to other projects.

The CNT GI Framework includes four social co-benefits: aesthetics, recreation, reduced noise pollution, and community cohesion [35]. Aesthetics are quantified as the increase in property value due to the presence of trees and green space. Recreation is quantified as an increase in "user days," or the additional days people gain access to the green space. The CNT GI Framework cites one Philadelphia study that quantified the value of increased user days for recreation [196]. Reduced noise pollution is also monetized as property values increase in response to decreased decibel levels. Lastly, community cohesion is qualitatively described as a benefit. The CNT GI Framework satisfactorily describes three quantified social co-benefits — such results would be helpful for planning GSI installation in the future. However, one issue is that property value increases can be highly variable and location-specific. The studies cited in the CNT GI Framework took place in Portland, OR; Philadelphia, PA; and King County, WA. Although the methods of valuation could apply elsewhere, GSI might not have the same magnitude of effect in other locations. Furthermore, no mention is made of environmental justice in relation to social co-benefits. Overall, the CNT GI Framework robustly presents several social co-benefits, given the difficulties of quantifying such benefits as property value increases.

Lastly, the Mersey Toolkit contains a thorough quantification of social co-benefits. In fact, the Mersey Toolkit focuses on social co-benefits: 6 of 11 subcategories of the calculator concern some measure of social co-benefits. These benefits categories include place and communities, health, land and property values, labor productivity, tourism, and recreation. The calculator's spreadsheets are detailed and require a deep knowledge of the potential GSI project, leading to robust results. Results include a cost-benefit analysis, a summary of economic outputs, and a summary of all benefits. The Mersey Forest created the Toolkit with the purpose of fostering a common understanding of all the potential GSI benefits between government, businesses, and civil society through an open-source platform.

The traditional approach to planning and implementing GSI focuses on hydrologic improvements as the primary goal. Economic and social co-benefits are not the focus, and the presence of such co-benefits might or might not add value to the GSI project sponsor. The 16 calculators that do not make mention of these co-benefits corroborates this traditional approach. A change to the traditional approach is suggested to incorporate consideration of the sustainability triple bottom line, wherein hydrologic, economic, and social benefits are analyzed and incorporated into a GSI project from the beginning (Figure 4.1). Utilizing such a tool as the Mersey Toolkit, GSI planners can better ensure that projects advance the ability of people to derive value from interaction with GSI.

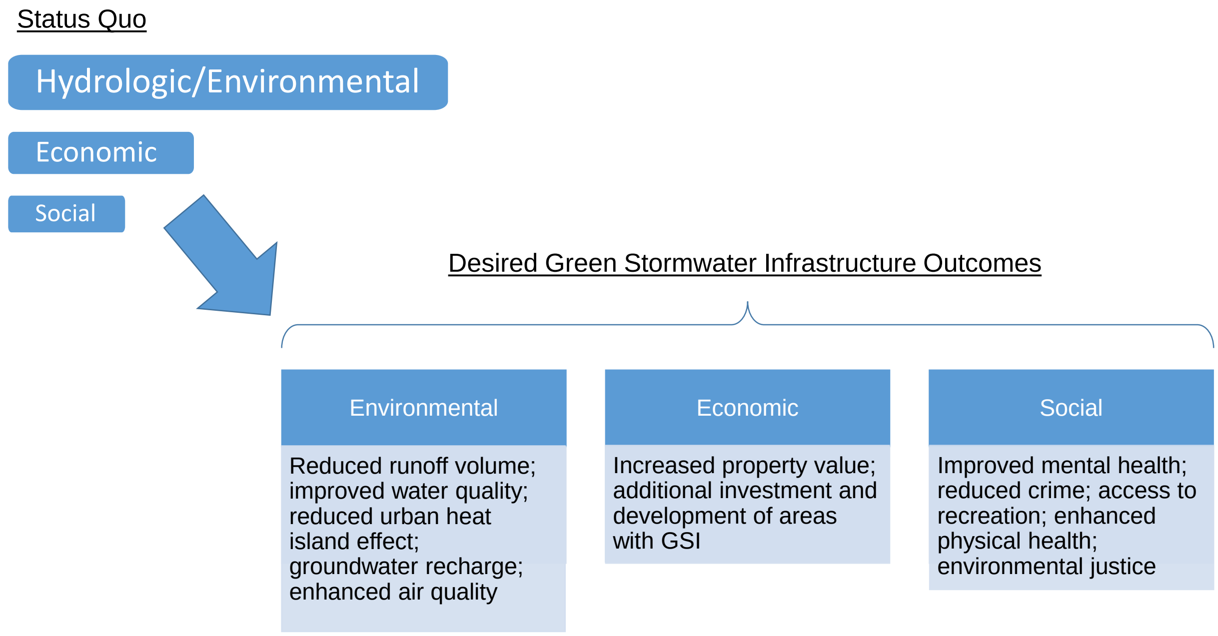


Figure 4.1: A shift in thinking is suggested, where green stormwater infrastructure designers and decision makers incorporate economic and social co-benefits at the planning stage and include those goals along with hydrologic benefits.

4.2 Green Stormwater Infrastructure and Environmental Justice Statistical Analyses

4.2.1 Chicago Data Description

GSI throughout the entirety of the City of Chicago is analyzed. Census tract-level socioeconomic data are first collected for the year 2015 for the 828 tracts within the city limits of Chicago. Race and income subsets of Census data are included. Income data consisted of the median household income. Race data include seven categories specified by the Census Bureau; white, black or African American, Asian, Native American and Alaska Native, Native Hawaiian and other Pacific Islander, some other race, and two or more races. These data are used to determine the percent white population present in each Census tract and use that percentage in the analysis. Note that there is inherent bias in these socioeconomic Census data, particularly regarding response bias. Furthermore, race categories can be confusing and even controversial [137]. The analysis proceeds fully acknowledging that these biases exist and might misrepresent the findings, yet U.S. Census data are the most complete socioeconomic data available.

GSI data were collected from the City of Chicago Data Portal [40] on neighborhood gardens, green roofs, and park spaces. As of 2015, there were 64 gardens, 172 green roofs, and 462 parks throughout Chicago. Data are downloaded as shapefiles and include the location of GSI and the areal coverage. Figure 4.2 presents an overview of the GSI within Chicago. The GSI data were combined with Census tract data and use ArcGIS to interpolate results for the frequency and areal coverage of GSI in each given Census tract. These results are used to perform statistical analyses on potential association of GSI with socioeconomic factors.

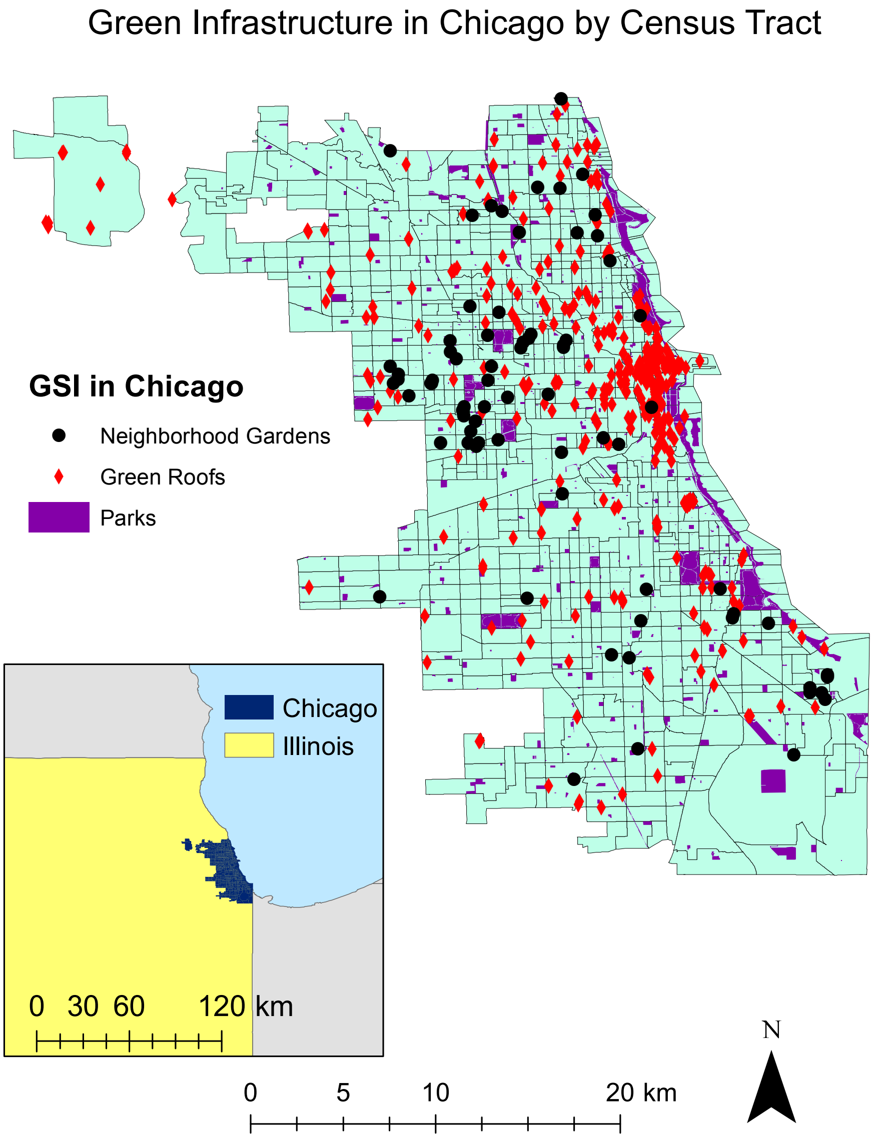


Figure 4.2: Green stormwater infrastructure (GSI) locations within the City of Chicago. The analysis investigates whether there is a relationship between the frequency and areal extent of GSI and socioeconomic factors for different Census tracts (data source: City of Chicago [40]).

4.2.2 Chicago Statistical Analysis

The aim in analyzing the City of Chicago is to determine whether both the number of GSI and areal coverage are associated with any trends in socioeconomic data. Linear regressions are employed to determine such associations. In total, there are five dependent variables and two independent variables — models are summarized in Table 4.2. Equations 4.1 through 4.5 represent the models, where P_{White} is the percent white population and Y is median household income. For all regression models, a significance level of $\alpha = 0.05$ is chosen.

$$Freq_{Garden} = \beta_0 + \beta_1 P_{White} + \beta_2 Y \tag{4.1}$$

$$Area_{Garden} = \beta_0 + \beta_1 P_{White} + \beta_2 Y \tag{4.2}$$

$$Freq_{Roof} = \beta_0 + \beta_1 P_{White} + \beta_2 Y \tag{4.3}$$

$$Area_{Roof} = \beta_0 + \beta_1 P_{White} + \beta_2 Y \tag{4.4}$$

$$Area_{Parks} = \beta_0 + \beta_1 P_{White} + \beta_2 Y \tag{4.5}$$

Table 4.2: The ten components as part of the Chicago analyses comprise racial makeup, income, and educational attainment.

Socioeconomic Indicator	Garden		Green Roof		Park
Race	Frequency	Area (sq ft)	Frequency	Area (sq ft)	Area (ac)
Income	Frequency	Area (sq ft)	Frequency	Area (sq ft)	Area (ac)

4.2.3 Chicago Results

Concerning the analysis of Chicago, the five regression models produced rather unconvincing results. Of the five, only one model was statistically significant: furthermore, only the median household income was a significant indicator variable (for both of the green roof models). Table 4.3 presents the results for all of the models. The green roof frequency model was significant, and median household income was a significant predictor in that model (correlated positively with GSI): however, the poor R^2 value of 0.04 indicates that more predictor variables would be necessary to fully predict variation in the frequency and area of green roofs in Chicago. Although the results are not conclusive regarding environmental justice in Chicago, it is worth establishing that data do exist and can be analyzed. The analysis in Chicago is a springboard for a deeper look into environmental justice and GSI in Philadelphia.

Table 4.3: Resulting significant linear regression coefficients based on household income.

GSI	Neighborhood Gardens		Green Roofs		Parks
Model	Count	Area	Count	Area	Area
R^2	0.02	0.01	0.04	0.03	0.003
Significant Variables			Median Household Income (+)	Median Household Income (+)	
p-value	0.58	0.73	0.03 *	0.09	0.52

4.2.4 Philadelphia Data Description

The analysis of GSI in Philadelphia (see Figure 4.3) involves a related yet different statistical approach. Multiple years of Census data are collected for Philadelphia, spanning from 2000 to 2016. Of particular interest are two subsets of the overall Census data: race and median household income, and the inherent bias mentioned previously is acknowledged.

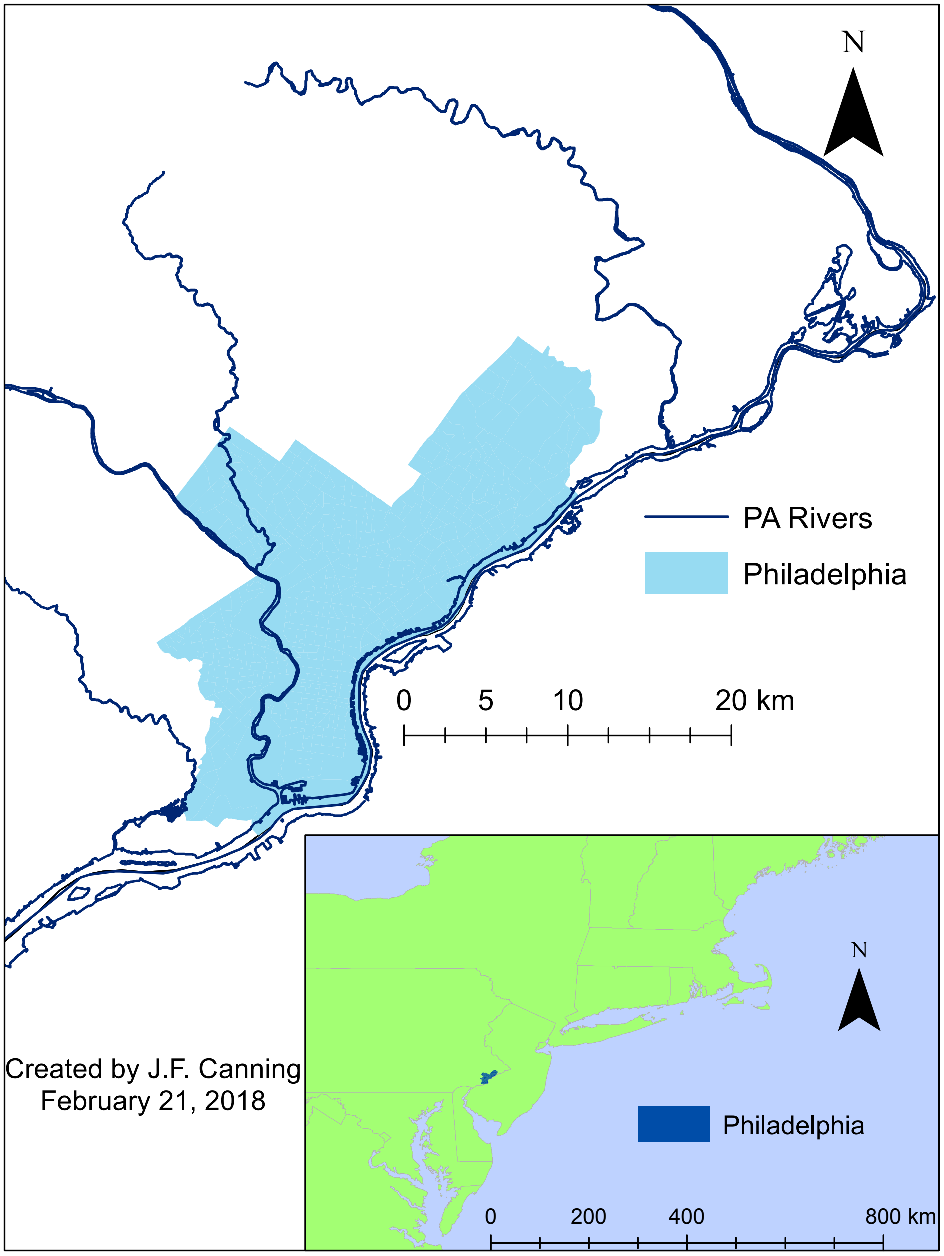


Figure 4.3: Philadelphia is situated along the Delaware River, and several tributaries run through the City (data source: The Pennsylvania Geospatial Data Clearinghouse [237]).

The United States Census Bureau has data available at the tract-level for the years 2000 and 2009–2016. Figure 4.4 presents the 2015 median household income and percent white residents at each tract, one example of the data in Philadelphia. The 2000 and 2010 data are the result of decennial efforts made by the Census Bureau. The 2009 and 2011–2016 data are the result of five-year American Community Survey (ACS) responses. The ACS began in 2005, and the first multi-year results were published in 2008. Initially, information was only collected in areas with a population greater than 20,000: however, beginning with results published in 2009, data were provided at the Census tract-level. Since the data are limited, non-parametric statistical tests are employed to accommodate the relatively small number of observations.

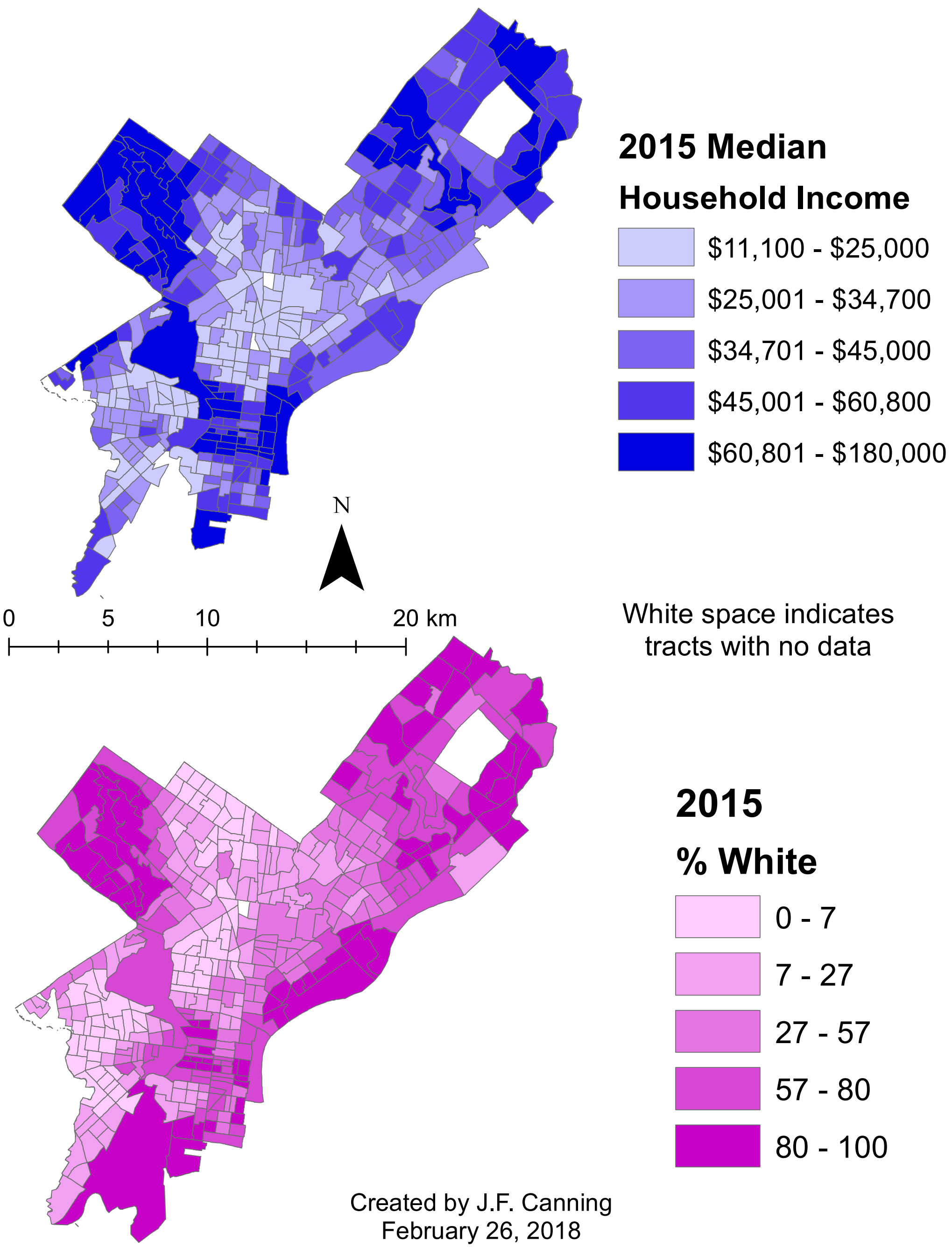


Figure 4.4: The median household income and racial makeup of Philadelphia in 2015 shows large differences between the city center and outlying areas (data source: U.S. Census Bureau).

Open-source GSI data from The Pennsylvania Geospatial Data Clearinghouse are then collected. The 'Philadelphia Green Stormwater Infrastructure Projects' [238] includes projects that began since the inception of the Philadelphia 'Green City, Clean Waters' program in 2011. This set of GSI data was chosen to analyze socioeconomic characteristics of Census tracts before and after GSI installation in 2011. In total, there were 277 different GSI projects, comprising of various types of GSI (stormwater tree trenches, rain gardens, infiltration trenches, and others). Figure 4.5 presents the location of GSI throughout Philadelphia.

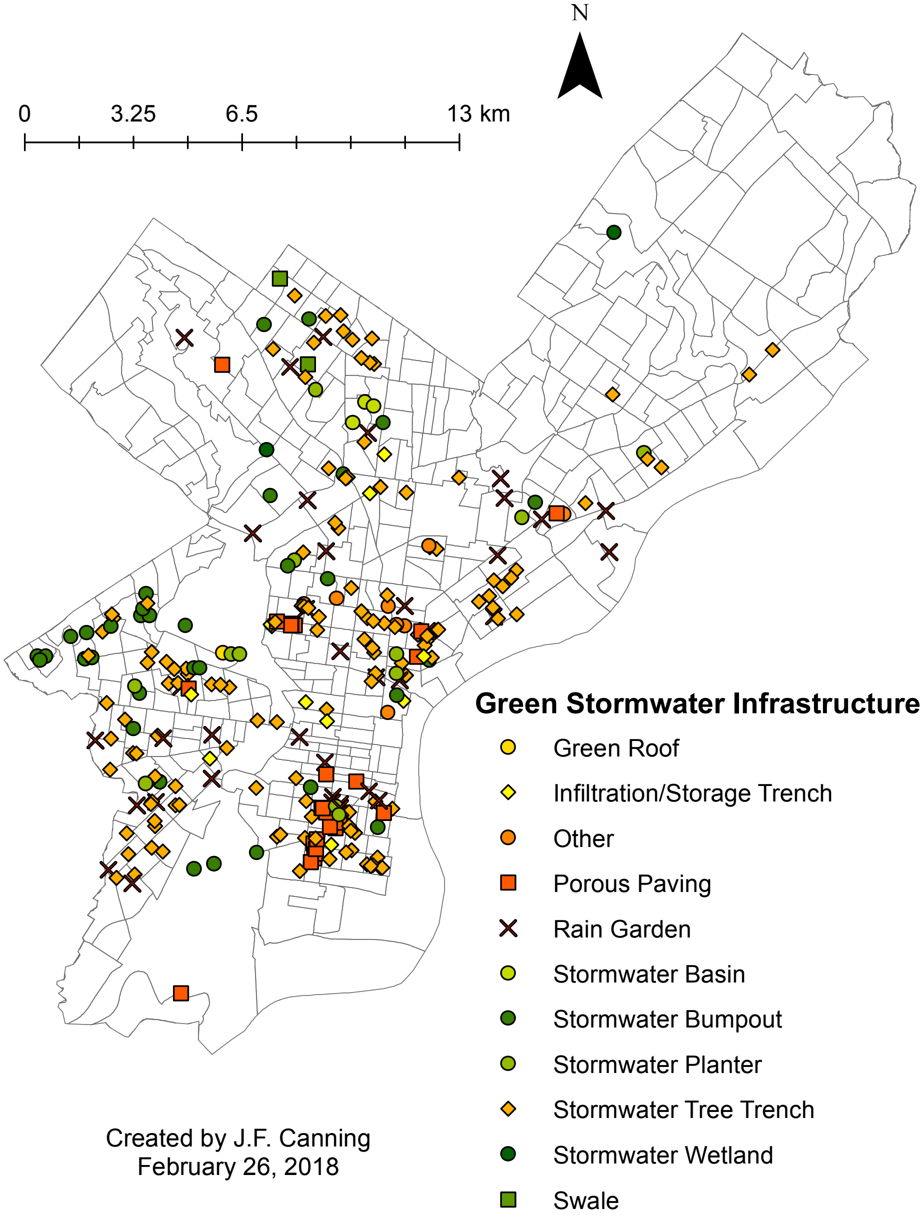


Figure 4.5: Green stormwater infrastructure locations within the City of Philadelphia (data source: The Pennsylvania Geospatial Data Clearinghouse [238]).

4.2.5 Philadelphia Statistical Analysis

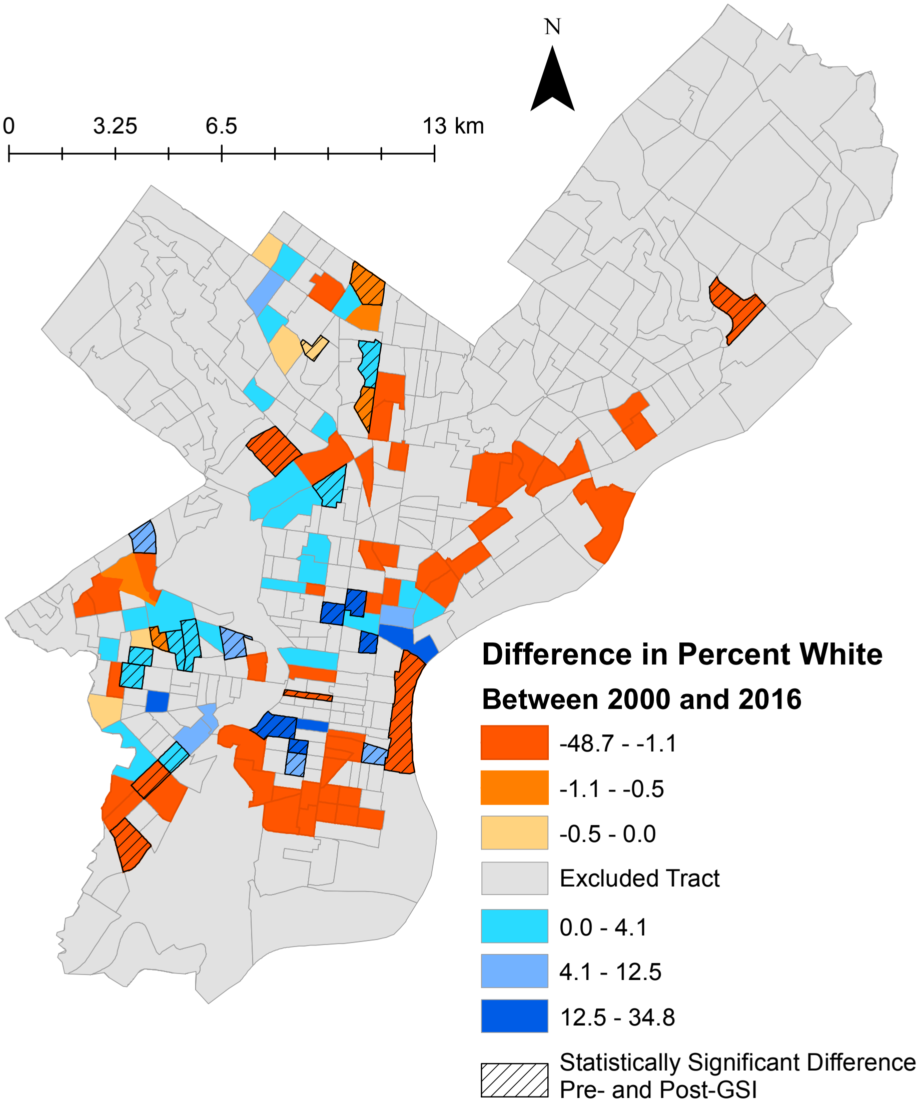
In total, there are nine years of socioeconomic data: three years before GSI installation and six years after. A non-parametric approach is then taken to analyze this relatively small dataset. Non-parametric testing assumes the data are independent and identically distributed, and is suitable for datasets of limited size. The Wilcoxon Rank Sum Test is utilized, a two-sample location problem that tests whether medians of the two samples are equal to one another [256]. The prediction is that median household income and percent white population will exhibit a statistically significant change after GSI installation efforts were initiated in 2011. Equation 4.6 lists the null and alternate hypotheses, where μ represents median household income or percent white population in a given tract. A significance threshold of $\alpha = 0.05$ is used to distinguish tracts that exhibit changes in income or percent white population before and after GSI installation.

$$\begin{aligned} H_0 &: \mu_{pre} = \mu_{post} \\ H_A &: \mu_{pre} \neq \mu_{post} \end{aligned} \tag{4.6}$$

The 277 GSI projects were spread throughout 142 Census tracts as of 2013. Of those 142 tracts, 36 were newly created at the 2010 Census, meaning that 106 existed at the 2000 Census and 2009 ACS data. There are a total of 384 tracts in Philadelphia as of 2016, and there are a total of 106 tracts with complete data. It is worth noting that numerous tracts are either not populated or have some extenuating circumstance rendering them non applicable to the analysis. For example, Tract 9891 encompasses a prison and thus has a population with no income; Tracts 50, 9807, and 9809 have no population, and Tract 9803 is the Northeast Philadelphia Airport.

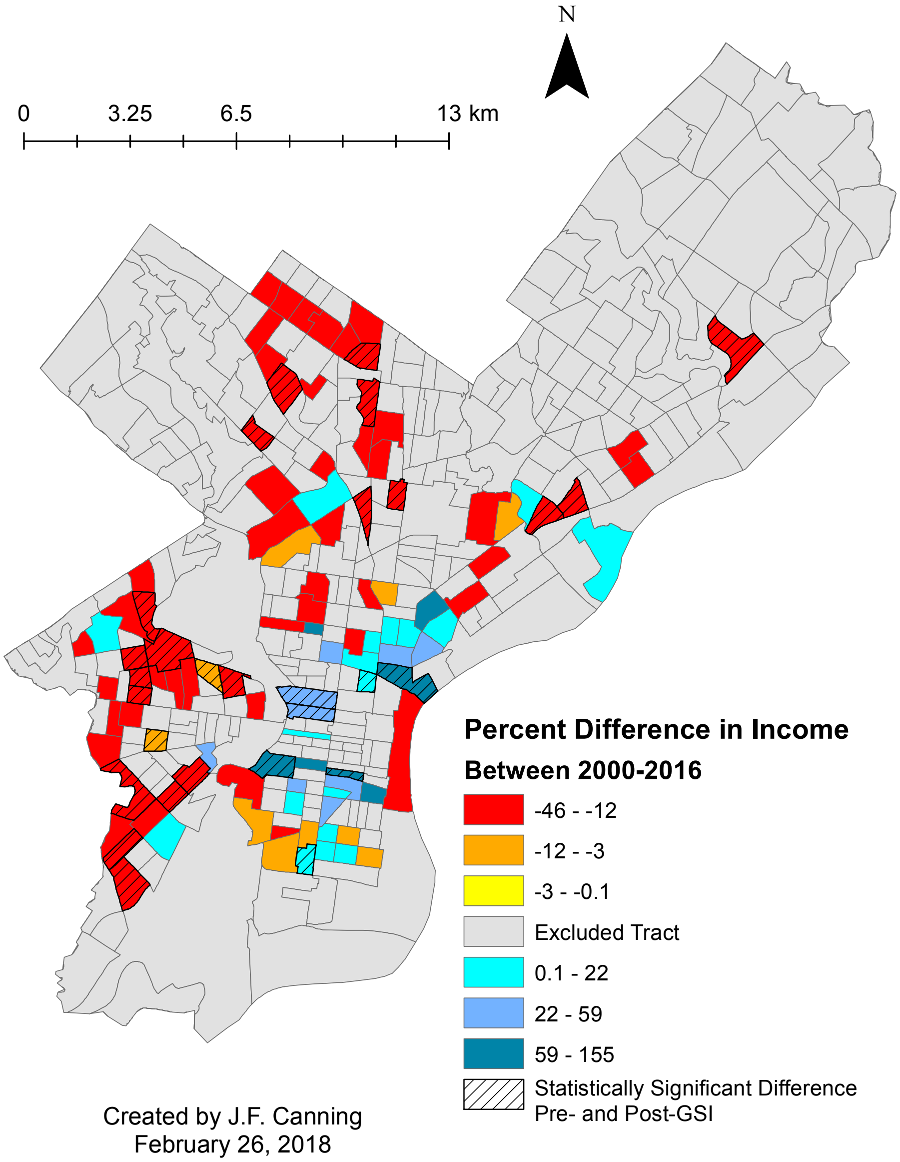
4.2.6 Philadelphia Results

After performing a Wilcoxon Rank Sum Test on the median household income and percent white population before and after Philadelphia's Green City Clean Waters program initiation in 2011, the analysis determined that 29 tracts exhibit a significant difference in median household income and 26 tracts exhibit a statistically significant difference in percent white population ($p < 0.05$). Of the 29 tracts that underwent a significant change in median household income before and after GSI installation, 7 increased in median household income between 2000 and 2016 (dollar amounts were adjusted for inflation). The mean 2000 household income in tracts that underwent a significant change was \$37,300, lower than the overall mean of \$38,800 for all tracts analyzed (in 2016 inflation-adjusted dollars). Of the 26 tracts with a significant change in percent white population, 16 had an increase in percent white population between 2000 and 2016. In 2000, the mean percentage of white population in tracts that underwent a significant change was 20.6 percent, lower than a mean value of 30.8 percent for all tracts analyzed. There were eight tracts that exhibited a significant change in both median household income and percent white population: two tracts had an increase in both factors; two tracts had a decrease in both factors, and four tracts had a decrease in median household income and simultaneous increase in percent white population. Figures 4.6 and 4.7 graphically show the analyzed tracts and their changes in percent white population and median household income, respectively.



Created by J.F. Canning
February 26, 2018

Figure 4.6: 106 tracts were analyzed for significant changes in racial makeup before and after green stormwater infrastructure (GSI) installation in 2011. Positive change indicates that the percentage of white residents increased between 2000 and 2016, while negative change indicates that racial diversity increased between 2000 and 2016. Hatching on an analyzed tract marks a statistically significant ($p < 0.05$) change in the percent of white residents.



Created by J.F. Canning
February 26, 2018

Figure 4.7: 106 tracts were analyzed for significant changes in median household income before and after green stormwater infrastructure (GSI) installation in 2011. Positive change indicates an increase in income between 2000 and 2016, and negative change indicates the opposite. Hatching on an analyzed tract marks a statistically significant ($p < 0.05$) change in the percent of white residents.

4.3 Statistical Analyses Discussion

These analyses first demonstrate that existing data can be analyzed using socioeconomic indicators to predict GSI installation in Chicago. Regression is used to show that median household income has a positive correlation with green roofs. That particular model is significant, yet ineffective with low predictive capability. However, the correlation between median household income and GSI still exists — as income increases, so does the likelihood that GSI is present near that Census tract. Further work could investigate more factors that could contribute to the presence of GSI throughout Chicago. Regression is a potential method of analyzing GSI for environmental justice implications. The Philadelphia analysis builds off these conclusions from the Chicago analysis.

Concerning Philadelphia, the analysis yielded some interesting and significant results. Multiple factors affect changes in median household income and the demographic makeup of each Census tract. Capturing all of these factors is outside the scope of this work; however, these results present possible correlations between GSI and income/demographics that have environmental justice implications. The 22 of 29 tracts that experienced a decrease in median household income between 2000 and 2016 could indicate that Philadelphia is intentionally installing GSI in economically depressed areas. Furthermore, the mean 2000 median household income was lower in these 22 tracts as compared to all tracts analyzed. In their "Green City, Clean Waters" Program Summary, the Philadelphia Water Department explicitly mentions striving to "restore urban waterways" in "areas not historically targeted by environmental outreach," i.e., environmental justice communities [191]. It would appear that Philadelphia is indeed focusing on communities that had been experiencing declines in median household income as locations for GSI installation. These results reflect this correlation between GSI and tracts that underwent a decrease in median household income.

This analysis of changes in demographics (i.e., changes in the percent white population living in a given tract) also yields results with implications for environmental justice. The 16 tracts that experienced a significant difference in demographic makeup had a larger minority population prior to GSI installation. Moreover, the 16 tracts consistently had a larger minority population throughout the entire time period as compared to the overall mean minority population in the 106 analyzed tracts. Environmental outreach has historically been neglected in minority communities, and thus once again, it appears that the Philadelphia Water Department is attaining its goal of furthering environmental justice as part of its Green City, Clean Waters program. However, the minority population is decreasing in those 16 tracts after GSI installation in 2011. Assessing causation between GSI installations and changing minority populations is beyond the capabilities of available data; however, it is worth mentioning that one undesirable effect of GSI implementation could be gentrification. Gentrification can loosely be defined as the "result of an alteration of preferences and/or a change in the constraints determining which preferences will or can be implemented" [223]. Gentrification can be thought of as a movement of capital and people back into a city center. Wolch et al. [260] showed that urban greening can have the unfortunate effect of displacing the same people whose livelihoods it strives to improve, and Dale and Newman [50] indicated that local sustainable development projects cannot guarantee maintaining existing social diversity and equity. This analysis does not imply that the same phenomenon is occurring in Philadelphia; the results merely illustrate that the decrease in racial diversity in these 16 tracts could possibly be an indication of impending gentrification. Further research would be necessary to fully substantiate the implications of these changes in racial demographics.

CHAPTER 5

CONCLUSION

As mentioned at the beginning of this document, stormwater management is necessary across all sorts of landscapes, both agricultural and urban. The immense growth of humanity and the demands placed on the planet have pushed stormwater management to the forefront of water-related issues in both agricultural and urban contexts. Green infrastructure mimics pre-development conditions and provides a more sustainable alternative to traditional gray infrastructure used to manage stormwater. This thesis addressed green infrastructure in both agricultural and urban contexts, seeking to maximize the triple-bottom line benefits offered by green infrastructure in both of the applications.

5.1 Agricultural Runoff and Water Quality Implications

Nitrates running off agricultural fields from fertilizer inputs are a major water-quality issue throughout the Midwestern United States. The Raccoon River Watershed (RRW) upstream of Des Moines, Iowa, USA, has some of the highest nitrate levels in the country, and Des Moines Water Works (DMWW) operates one of the world's largest nitrate removal facilities to reduce levels below the federal drinking water standard. In 2015, DMWW pursued litigation against upstream drainage districts, claiming that agricultural operators ought to be responsible for the high nitrate levels and treat the water leaving fields through tile drains. DMWW was unsuccessful in its litigation, and the nitrate problem continues to plague Des Moines residents. At the outset, several questions were proposed:

- What are the ramifications of using agricultural green infrastructure to nitrate reduction?
- Where might agricultural green infrastructure be placed, and how much will it cost?

Three possible response scenarios to the issue are analyzed: S0) business as usual, S1) initiating an extensive agricultural green infrastructure project, and S2) maintaining the current nitrate removal facility and/or building a new facility. The first scenario is not sustainable because eventually the current nitrate removal facility will become defunct and DMWW will not meet the federal drinking water standard. The second response is widespread installation of agricultural green infrastructure in the form of riparian buffers along first-order streams throughout the RRW; the distributed, green approach. Water-quality trading (WQT) encourages utility-agriculture partnerships to improve water quality, extended to apply to drinking water quality at DMWW. This proposition could cost Des Moines residents between \$135 and \$160 million, depending on the amount of vegetation required. This investment is admittedly a large sum of money; however, there are additional benefits beyond improved water quality, including increased living-wage jobs and carbon sequestration. Furthermore, the distributed, green approach helps bridge the rural-urban divide through providing a means for cooperation between the two parties. The third response includes renovations of the nitrate removal facility and potential construction of a brand new facility; the centralized, gray approach. In the present analysis, this proposition could cost Des Moines ratepayers anywhere between \$71 and \$184 million, the upper bound representing a completely new nitrate removal facility. Building a new nitrate removal facility certainly addresses nitrate pollution, but it does not advance cooperation between DMWW and upstream agricultural operators. Ultimately, infrastructure and management approaches are necessary to address water quality challenges, both in the RRW and throughout other agricultural watersheds in the Midwestern United States. Centralized treatment provides residents with clean drinking water, without the added co-benefits of a distributed, green approach. Agricultural green infrastructure is a potential solution to improving water quality with several co-benefits, and water-quality trading approaches can encourage such solutions.

5.2 Social Co-Benefits of Green Stormwater Infrastructure

Green stormwater infrastructure (GSI) is a distributed, natural means of infiltrating and treating stormwater runoff, akin to agricultural green infrastructure but in an urban context.

GSI is an alternative to traditional gray infrastructure and offers many co-benefits above and beyond achieving hydrologic improvements. These co-benefits follow a triple bottom-line approach and include environmental, economic, and social improvements. This thesis focuses on social co-benefits: GSI has been shown to improve physical health, reduce crime, decrease stress, improve mental health, and provide recreation. Often, achieving hydrologic goals is the primary motivation behind GSI projects, and social co-benefits are an afterthought. However, results suggest that more emphasis could be placed on attaining social co-benefits at the planning stages of GSI implementation. Synergistic planning between engineers, landscape architects, policy makers, and neighborhood leaders can foment projects that achieve hydrologic and social benefits for the local community. A question was posed investigating whether such synergistic planning is possible using currently available GSI calculators:

- To what extent are social co-benefits included in planning for GSI projects, as reflected by available calculators?

Currently available GSI calculators, tools, and frameworks are examined to assess whether they include social co-benefits as part of their methodologies/results. Several similar studies have analyzed multiple calculators for their hydrologic methodologies, but none, to the author's knowledge, have focused on the social aspects of GSI. A total of 21 different calculators, tools, and frameworks were assessed, and only five mention social co-benefits. Of these five, two attempt to quantify social co-benefits in a meaningful way: the Center for Neighborhood Technology "The Value of Green Infrastructure" Framework and the Mersey Forest Green Infrastructure Valuation Toolkit. The latter is an extensive set of spreadsheets that focuses more on quantifying social co-benefits than on quantifying hydrologic/environmental benefits. Admittedly, GSI is typically designed for hydrologic functions, but more of a concerted effort can be made to achieve social co-benefits, especially environmental justice.

5.3 Environmental Justice

Environmental injustice has burdened a disproportionate amount of environmental ills on disadvantaged communities, such as low-income and minority communities, and allowed a disproportionate amount of environmental benefits to befall affluent communities. Theoretically, decision makers could utilize the information provided in GSI calculators to enact social change and benefits as they pertain to investing in low-income and minority communities. The following question was addressed:

- How might existing socioeconomic data be used to analyze GSI projects and their implications on environmental justice?

Two different statistical approaches are conducted to analyze two cities leading in GSI implementation in the United States: Chicago, IL and Philadelphia, PA. Maps indicating where GSI are located at the Census tract-level are utilized and socioeconomic data at the tract-level are gathered to determine whether potential relationships exist. In Chicago, a regression is used to analyze potential correlations between median household income and race and the frequency and size of GSI in 2015. Findings indicate that median household income significantly ($p < 0.05$) correlates positively with green roof frequency and area. However, using these indicators does not explain much of the variation in GSI frequency and area (R^2 values were less than 0.1). The analysis shows how freely-available data can be used to investigate GSI implications on environmental justice.

Socioeconomic data are further used at the Census tract-level to analyze the City of Philadelphia and its Green City, Clean Waters program. Using 2011 as the crux point, the differences in median household income and race before and after GSI installation are analyzed using non-parametric testing to find significant changes at the tract-level. In total, 106 tracts are analyzed: of these tracts, 29 had a significant change in median household income and 26 had a significant change in percent of white residents ($p < 0.05$). Philadelphia Water Department included in their goals as part of Green City, Clean Waters to invest in areas not traditionally targeted for environmental outreach. The program is still rather young at about seven years, but findings indicate that Philadelphia Water Department is indeed investing in low-income neighborhoods. The analysis shows that several areas have experienced statistically significant increases in income from the period before GSI installation to after. There are many possible reasons such an increase could occur; however, the results suggest that Philadelphia Water Department is moving toward its environmental justice goals. The analysis in Philadelphia, like the analysis in Chicago, shows that these socioeconomic data can be utilized to plan and monitor environmental justice efforts as part of GSI installation.

5.4 Concluding Remarks

Policy makers, engineers, landscape architects, and community leaders have an opportunity to streamline hydrologic and social improvements in agricultural areas through the use of agricultural green infrastructure and in urban areas through the use of GSI. Adjusting the GSI paradigm can achieve the necessary hydrologic improvements while simultaneously improving local residents' mental and physical health, providing recreation and aesthetic, and advancing environmental justice in low-income communities. Encouraging widespread use of riparian buffers in agricultural watersheds can reduce water quality degradation locally and further downstream, besides providing social co-benefits such as increased access to living-wage jobs. Overall, distributed, green systems of stormwater runoff management yield plentiful benefits over centralized, gray systems of treatment and management.

REFERENCES

[1] 33rd U.S. Congress. U.S. Federal Water Pollution Control Act. Section 402(q)(1), 2002.

[2] Lauren C Abercrombie, James F Sallis, Terry L Conway, Lawrence D Frank, Brian E Saelens, and James E Chapman. Income and racial disparities in access to public parks and private recreation facilities. *American Journal of Preventive Medicine*, 34(1):9–15, 2008.

[3] K. Addy, A.J. Gold, L.E. Christianson, M.B. David, L.A. Schipper, and N.A. Ratigan. Denitrifying bioreactors for nitrate removal: a meta-analysis. *Journal of Environmental Quality*, 45:873–881, 2016.

[4] Miguel Aguayo, Ziwen Yu, Michael Piasecki, and Franco Montalto. Development of a web application for low impact development rapid assessment (LIDRA). *Journal of Hydroinformatics*, 15(4):1276–1295, 2013.

[5] A Osman Akan. Modified rational method for sizing infiltration structures. *Canadian Journal of Civil Engineering*, 29(4):539–542, 2002.

[6] Hashem Akbari. Shade trees reduce building energy use and CO2 emissions from power plants. *Environmental Pollution*, 116:S119–S126, 2002.

[7] Amy W Ando and Luiz PC Freitas. Consumer demand for green stormwater management technology in an urban setting: The case of Chicago rain barrels. *Water Resources Research*, 47(12), 2011.

[8] Callum Andrew and Duncan Slater. Why some UK homeowners reduce the size of their front garden trees and the consequences for urban forest benefits as assessed by i-Tree ECO. *Arboricultural Journal: The International Journal of Urban Forestry*, 36 (4):197–215, 2014.

[9] Jonathan T Angier and Gregory W McCarty. Variations in base-flow nitrate flux in a first-order stream and riparian zone. *Journal of the American Water Resources Association*, 44(2):367–380, 2008.

[10] Yeganeh Asadian, Markus Weiler, et al. A new approach in measuring rainfall interception by urban trees in coastal British Columbia. *Water Quality Research Journal of Canada*, 44(1):16, 2009.

[11] Dustin Atchison, Kenneth W Potter, and Linda Severson. *Design guidelines for stormwater bioretention facilities*. Univesity of Wisconsin Water Resources Institute, 2006.

[12] Carmine C Balascio and William C Lucas. A survey of storm-water management water quality regulations in four Mid-Atlantic States. *Journal of Environmental Management*, 90(1):1–7, 2009.

[13] April Karen Baptiste, Catherine Foley, and Richard Smardon. Understanding urban neighborhood differences in willingness to implement green infrastructure measures: a case study of Syracuse, NY. *Landscape and Urban Planning*, 136:1–12, 2015.

[14] Julia Bartens, Susan D Day, J Roger Harris, Theresa M Wynn, and Joseph E Dove. Transpiration and root development of urban trees in structural soil stormwater reservoirs. *Environmental Management*, 44(4):646–657, 2009.

[15] Joseph Battiata, Kelly Collins, David Hirschman, and Greg Hoffmann. The runoff reduction method. *Journal of Contemporary Water Research & Education*, 146(1): 11–21, 2010.

[16] Nicole G Beck, Gary Conley, Lisa Kanner, and Margaret Mathias. An urban runoff model designed to inform stormwater management decisions. *Journal of Environmental Management*, 193:257–269, 2017.

[17] Chris Behr and Franco Montalto. Risk Analysis Application for assessing the cost-effectiveness of low impact development for CSO control using LIDRA. In *Low Impact Development for Urban Ecosystem and Habitat Protection*, pages 1–10. 2009.

[18] Janice F Bell, Jeffrey S Wilson, and Gilbert C Liu. Neighborhood greenness and 2-year changes in body mass index of children and youth. *American Journal of Preventive Medicine*, 35(6):547–553, 2008.

[19] Mark A Benedict and Edward T McMahon. Green infrastructure: smart conservation for the 21st century. *Renewable Resources Journal*, 20(3):12–17, 2002.

[20] John C Bergstrom and Paul Civita. Status of benefits transfer in the United States and Canada: a review. *Canadian Journal of Agricultural Economics*, 47(1):79–87, 1999.

[21] Justyna Czemiel Berndtsson. Green roof performance towards management of runoff water quantity and quality: A review. *Ecological Engineering*, 36(4):351–360, 2010.

[22] Budhendra Bhaduri, Marie Minner, Susan Tatalovich, and Jon Harbor. Long-term hydrologic impact of urbanization: a tale of two models. *Journal of Water Resources Planning and Management*, 127(1):13–19, 2001.

[23] 42 65 et al. Board of Water Works Trustees of the City of Des Moines, Iowa v. Sac County Board of Supervisors as Trustees of Drainage Districts 32. United States District Court for the Northern District of Iowa Western Division. decided March 17, 2017, 2017.

[24] James F Booker and Robert A Young. Modeling intrastate and interstate markets for Colorado River water resources. *Journal of Environmental Economics and Management*, 26(1):66–87, 1994.

[25] Christopher G Boone, Geoffrey L Buckley, J Morgan Grove, and Chona Sister. Parks and people: An environmental justice inquiry in Baltimore, Maryland. *Annals of the Association of American Geographers*, 99(4):767–787, 2009.

[26] Dean R Bork and Josh Franklin. Revitalizing Urbanized Watersheds through Smart Growth: The Fairfax Boulevard Case Study. 2010.

[27] Kathryn J Bowen and Yvonne Lynch. The public health benefits of green infrastructure: the potential of economic framing for enhanced decision-making. *Current Opinion in Environmental Sustainability*, 2017.

[28] David Briggs, Juan J Abellan, and Daniela Fecht. Environmental inequity in england: small area associations between socio-economic status and environmental pollution. *Social Science & Medicine*, 67(10):1612–1629, 2008.

[29] Darrell Brown, Jesse F Dillard, and Scott Marshall. Triple bottom line: a business metaphor for a social construct. 2006.

[30] W.D. Browning, C. Garvin, C. Ryan, B. Fox, R. Cook, L. Labruto, N. Kallianpurkar, S. Watson, and T. Knop. The Economics of Biophilia: Why designing with nature in mind makes financial sense. *Terrapin Bright Green*, 2012.

[31] M.R. Burkart, O.E. James, and M.D. Tomer. Hydrologic and terrain variables to aid strategic location of riparian buffers. *Journal of Soil and Water Conservation*, 59(5): 216–223, 2004.

[32] T.P. Burt, G. Pinay, F.E. Matheson, N.E. Haycock, A. Butturini, J.C. Clement, S. Danielescu, D.J. Dowrick, M.M. Hefting, A. Hillbricht-Ilkowska, and V. Maitre. Water table fluctuations in the riparian zone: comparative results from a pan-European experiment. *Journal of Hydrology*, 265(1):129–148, 2002.

[33] Maria Theresa I Cabaraban, Charles N Kroll, Satoshi Hirabayashi, and David J Nowak. Modeling of air pollutant removal by dry deposition to urban trees using a WRF/CMAQ/i-Tree Eco coupled system. *Environmental Pollution*, 176:123–133, 2013.

[34] Jacob Carstensen, Jesper H Andersen, Bo G Gustafsson, and Daniel J Conley. Deoxygenation of the Baltic Sea during the last century. *Proceedings of the National Academy of Sciences*, 111(15):5628–5633, 2014.

[35] Center for Neighborhood Technology. The Value of Green Infrastructure: A Guide to Recognizing Its Economic, Environmental and Social Benefits, 2010.

[36] Christopher M Chini, James F Canning, Kelsey L Schreiber, Joshua M Peschel, and Ashlynn S Stillwell. The green experiment: Cities, green stormwater infrastructure, and sustainability. *Sustainability*, 9(1):105, 2017.

[37] B. Christen. *Tree root influences on soil physical properties under shelterbelts on pasture: design and evaluation of an approach using dye staining.* dissertation, University of Wales, Bangor, 2007.

[38] B. Christen and T. Dalgaard. Buffers for biomass production in temperate European agriculture: A review and synthesis on function, ecosystem services and implementation. *Biomass and Bioenergy*, 55:53–67, 2013.

[39] Ting Fong May Chui, Xin Liu, and Wenting Zhan. Assessing cost-effectiveness of specific LID practice designs in response to large storm events. *Journal of Hydrology*, 533:353–364, 2016.

[40] City of Chicago. Chicago Data Portal. Accessed at `https://data.cityofchicago.org/browse`, 2018.

[41] Roger Claassen, Andrea Cattaneo, and Robert Johansson. Cost-effective design of agri-environmental payment programs: U.S. experience in theory and practice. *Ecological Economics*, 65(4):737–752, 2008.

[42] James F Coles, Gerard McMahon, Amanda H Bell, Larry R Brown, Faith A Fitzpatrick, Barbara C Scudder Eikenberry, Michael D Woodside, Thomas F Cuffney, Wade L Bryant, and Karen Cappiella. Effects of urban development on stream ecosystems in nine metropolitan study areas across the United States. Technical report, US Geological Survey, 2012.

[43] A.B. Cooper, C.M. Smith, and M.J. Smith. Effects of riparian set-aside on soil characteristics in an agricultural landscape: implications for nutrient transport and retention. *Agriculture Ecosystems and Environment*, 55(1):61–67, 1995.

[44] Cooperative Research Centre for Catchment Hydrology, Melbourne, Australia. Model for Urban Stormwater Improvement Conceptualisation (MUSIC) (Version 3.0.1), 2005.

[45] D.L. Correll. Principles of planning and establishment of riparian buffer zones. *Ecological Engineering*, 24:433–439, 2005.

[46] Amanda L. Crawford. Nutrient Pollution and the Gulf of Mexico Dead Zone: Will Des Moines Water Works Be a Turning Point? *Tulane Law Review*, 91:157, 2016.

[47] Bethany B Cutts, Kate J Darby, Christopher G Boone, and Alexandra Brewis. City structure, obesity, and environmental justice: an integrated analysis of physical and social barriers to walkable streets and park access. *Social Science & Medicine*, 69(9):1314–1322, 2009.

[48] JM Dabrowski, K Murray, PJ Ashton, and JJ Leaner. Agricultural impacts on water quality and implications for virtual water trading decisions. *Ecological Economics*, 68 (4):1074–1082, 2009.

[49] Michael J Dagg and Greg A Breed. Biological effects of Mississippi River nitrogen on the northern Gulf of Mexico: A review and synthesis. *Journal of Marine Systems*, 43 (3-4):133–152, 2003.

[50] Ann Dale and Lenore L Newman. Sustainable development for some: green urban development and affordability. *Local Environment*, 14(7):669–681, 2009.

[51] Marcel Darveau, Patrick Beauchesne, Louis Belanger, Jean Huot, and Pierre Larue. Riparian forest strips as habitat for breeding birds in boreal forest. *The Journal of Wildlife Management*, pages 67–78, 1995.

[52] Bryan R Davies, Martin Thoms, and Michael Meador. An assessment of the ecological impacts of inter-basin water transfers, and their threats to river basin integrity and conservation. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 2(4):325–349, 1992.

[53] Allen P Davis. Field performance of bioretention: Hydrology impacts. *Journal of Hydrologic Engineering*, 13(2):90–95, 2008.

[54] Sjerp De Vries, Robert A Verheij, Peter P Groenewegen, and Peter Spreeuwenberg. Natural environments — healthy environments? An exploratory analysis of the relationship between greenspace and health. *Environment and Planning A*, 35(10):1717–1731, 2003.

[55] Des Moines Water Works. Five-year capital improvement plan, June 27, 2017.

[56] Giovanna Di Chiro. Nature as community: The convergence of environment and social justice. *Uncommon ground: Rethinking the Human Place in Nature*, pages 298–320, 1996.

[57] M.W. Diebel, J.T. Maxted, D.M. Robertson, S. Han, and M.J.V. Handen. Landscape planning for agricultural nonpoint source pollution reduction III: assessing phosphorus and sediment reduction potential. *Environmental Management*, 43(1):69–83, 2009.

[58] Michael E Dietz. Low impact development practices: A review of current research and recommendations for future directions. *Water, Air, and Soil Pollution*, 186(1-4): 351–363, 2007.

[59] D.L. Dinnes, D.L. Karlen, D.B. Jaynes, T.C. Kaspar, J.L. Hatfield, T.S. Colvin, and C.A. Cambardella. Nitrogen management strategies to reduce nitrate leaching in tile-drained Midwestern soils. *Agronomy Journal*, 94:153–171, 2002.

[60] O Doering, JN Galloway, TL Theis, V Aneja, E Boyer, KG Cassman, EB Cowling, RR Dickerson, W Herz, DL Hey, et al. Reactive Nitrogen in the United States: an analysis of inputs, flows, consequences, and management options. *United States Environmental Protection Agency*, 2011.

[61] J.M. Dorioz, D. Wang, J. Poulenard, and D. Trévisan. The effect of grass buffer strips on phosphorus dynamics — a critical review and synthesis as a basis for application in agricultural landscapes in France. *Agriculture Ecosystems and Environment*, 117(1): 4–21, 2006.

[62] Alexandra Dapolito Dunn. Siting green infrastructure: legal and policy solutions to alleviate urban poverty and promote healthy communities. *Boston College Environmental Affairs Law Review*, 37:41, 2010.

[63] Rahm Emanuel and TH Powers. City of Chicago Stormwater Management Ordinance Manual. *City*, 2014.

[64] Clay H Emerson, Bridget M Wadzuk, and Robert G Traver. Hydraulic evolution and total suspended solids capture of an infiltration trench. *Hydrological Processes*, 24(8): 1008–1014, 2010.

[65] Bernard A Engel, Jin-Yong Choi, Jon Harbor, and Shilpam Pandey. Web-based DSS for hydrologic impact evaluation of small watershed land use changes. *Computers and Electronics in Agriculture*, 39(3):241–249, 2003.

[66] Environmental Protection Agency. Water quality trading. Accessed at `https://www.epa.gov/npdes/water-quality-trading`, 2017.

[67] Henrik Ernstson. The social production of ecosystem services: A framework for studying environmental justice and ecological complexity in urbanized landscapes. *Landscape and Urban Planning*, 109(1):7–17, 2013.

[68] Henrik Ernstson and Sverker Sörlin. Weaving protective stories: connective practices to articulate holistic values in the Stockholm National Urban Park. *Environment and Planning A*, 41(6):1460–1479, 2009.

[69] Ekaterini Eumorfopoulou and Dimitris Aravantinos. The contribution of a planted roof to the thermal protection of buildings in Greece. *Energy and Buildings*, 27(1): 29–36, 1998.

[70] Ian Evans. Seven ideas for fixing water in the United States. Accessed at `https://www.newsdeeply.com/water/community/2018/01/17/seven-ideas-for-fixing-water-in-the-united-states`, January 17, 2018.

[71] Andrea Faber Taylor and Frances E Kuo. Children with attention deficits concentrate better after walk in the park. *Journal of Attention Disorders*, 12(5):402–409, 2009.

[72] P. Feather, D. Hellerstein, and L. Hansen. Economic valuation of environmental, benefits and the targeting of conservation programs: the case of the CRP. *Agricultural Economic Report*, 778:1–56, 1999.

[73] E Finlay. Green infrastructure to combat climate change. *Northwest Climate Change Action Plan and GRaBS project. Edinburgh: Community Forest Northwest*, 2010.

[74] Roberto Fioretti, Anna Palla, Luca G Lanza, and Paolo Principi. Green roof energy and water related performance in the Mediterranean climate. *Building and Environment*, 45(8):1890–1904, 2010.

[75] Joshua B Fisher, Maggi Kelly, and Jeff Romm. Scales of environmental justice: Combining gis and spatial analysis for air toxics in west oakland, california. *Health & Place*, 12(4):701–714, 2006.

[76] Karen Fisher-Vanden and Sheila Olmstead. Moving pollution trading from air to water: potential, problems, and prognosis. *The Journal of Economic Perspectives*, 27 (1):147–171, 2013.

[77] Myron F Floyd, Carlos J Crespo, and James F Sallis. Active living research in diverse and disadvantaged communities: stimulating dialogue and policy solutions. *American Journal of Preventive Medicine*, 34(4):271–274, 2008.

[78] Forestry Suppliers. Blue-X Tree Shelters. Accessed at `http://www.forestry-suppliers.com/product_pages/products.asp?mi=8138`, 2018.

[79] Stuart R Gaffin, Cynthia Rosenzweig, and Angela YY Kong. Adapting to climate change through urban green infrastructure. *Nature Climate Change*, 2(10):704, 2012.

[80] Danielle Gallet. The value of green infrastructure: a guide to recognizing its economic, environmental and social benefits. *Proceedings of the Water Environment Federation*, 2011(17):924–928, 2011.

[81] Leonard P. Gianessi and Henry M. Peskin. Analysis of national water pollution control policies: 2. Agricultural sediment control. *Water Resources Research*, 17(4):803–821, 1981.

[82] Susannah Gill, Paul Nolan, Tom Butlin, Tom Ferguson, and Clare Olver. Planning green infrastructure at a strategic level: experience from The Mersey Forest. *Handbook on Green Infrastructure: Planning, Design and Implementation. Cheltenham, UK: Edward Elgar Publishing*, pages 124–144, 2015.

[83] JW Gilliam, JE Parsons, and RL Mikkelsen. Nitrogen dynamics and buffer zones. *Buffer Zones*, page 54, 1996.

[84] Brendan Gleeson and Nicholas Low. *Justice, society and nature: An exploration of political ecology.* Routledge, 2002.

[85] H.C.J. Godfray, J.R. Beddington, I.R. Crute, L. Haddad, D. Lawrence, J.F. Muir, J. Pretty, S. Robinson, S.M. Thomas, and C. Toulmin. Food security: the challenge of feeding 9 billion people. *Science*, 327(5967):812–818, 2010.

[86] Nivedita G Gogate, Pradip P Kalbar, and Pratap M Raval. Assessment of stormwater management options in urban contexts using Multiple Attribute Decision-Making. *Journal of Cleaner Production*, 142:2046–2059, 2017.

[87] Donald A Goolsby and William A Battaglin. Long-term changes in concentrations and flux of nitrogen in the Mississippi River basin, USA. *Hydrological Processes*, 15 (7):1209–1226, 2001.

[88] Ashantha Goonetilleke, Evan Thomas, Simon Ginn, and Dale Gilbert. Understanding the role of land use in urban stormwater quality management. *Journal of Environmental Management*, 74(1):31–42, 2005.

[89] Suzie Greenhalgh and Mindy Selman. Comparing water quality trading programs: what lessons are there to learn? *Journal of Regional Analysis & Policy*, 42(2):104, 2012.

[90] Frank R Greer and Michael Shannon. Infant methemoglobinemia: the role of dietary nitrate in food and water. *Pediatrics*, 116(3):784–786, 2005.

[91] Sara Grineski, Timothy W Collins, Jayajit Chakraborty, and Marilyn Montgomery. Hazardous air pollutants and flooding: A comparative interurban study of environmental injustice. *GeoJournal*, 80(1):145–158, 2015.

[92] Sara E Grineski. Incorporating health outcomes into environmental justice research: The case of children's asthma and air pollution in Phoenix, Arizona. *Environmental Hazards*, 7(4):360–371, 2007.

[93] Qizhong Guo and Carlos A Correa. The impacts of green infrastructure on flood level reduction for the Raritan River: Modeling assessment. In *World Environmental and Water Resources Congress 2013: Showcasing the Future*, pages 367–376, 2013.

[94] H.J. Hamlin. Nitrate toxicity in Siberian sturgeon (*Acipenser baeri*). *Aquaculture*, 253: 688–693, 2005.

[95] Whinston Harrington, Alan J. Krupnick, and Henry M. Peskin. Policies for Nonpoint-Source Water Pollution Control. *Journal of Soil and Water Conservation*, 40:27–32, 1985.

[96] Terry Hartig, Gary W Evans, Larry D Jamner, Deborah S Davis, and Tommy Gärling. Tracking restoration in natural and urban field settings. *Journal of Environmental Psychology*, 23(2):109–123, 2003.

[97] David Harvey and Bruce Braun. *Justice, nature and the geography of difference*, volume 468. Wiley Online Library, 1996.

[98] J.L. Hatfield, L.D. McMullen, and C.S. Jones. Nitrate-nitrogen patterns in the Raccoon River Basin related to agricultural practices. *Journal of Soil and Water Conservation*, 64(3):190–199, 2009.

[99] Sabrina Havard, Séverine Deguen, Denis Zmirou-Navier, Charles Schillinger, and Denis Bard. Traffic-related air pollution and socioeconomic status: a spatial autocorrelation study to assess environmental equity on a small-area scale. *Epidemiology*, 20(2):223–230, 2009.

[100] John Heisler, Patricia M Glibert, JoAnn M Burkholder, Donald M Anderson, William Cochlan, William C Dennison, Quay Dortch, Christopher J Gobler, Cynthia A Heil, and Edythe Humphries. Eutrophication and harmful algal blooms: a scientific consensus. *Harmful Algae*, 8(1):3–13, 2008.

[101] M.E. Hernandez and W.J. Mitsch. Denitrification in created riverine wetlands: Influence of hydrology and season. *Ecological Engineering*, 30:78–88, 2007.

[102] Will Hewes. Creating Jobs and Stimulating the Economy through Investment in Green Water Infrastructure. *American Rivers and Alliance for Water Efficiency*, 2008.

[103] Nik Heynen, Harold A Perkins, and Parama Roy. The political ecology of uneven urban green space: the impact of political economy on race and ethnicity in producing environmental inequality in milwaukee. *Urban Affairs Review*, 42(1):3–25, 2006.

[104] MJ Higgins, CA Rock, R Bouchard, and B Wengrezynek. Controlling agricultural runoff by use of constructed wetlands. *Constructed wetlands for water quality improvement. CRC Press, Boca Raton*, pages 359–367, 1993.

[105] Alan R Hill. Nitrate removal in stream riparian zones. *Journal of Environmental Quality*, 25(4):743–755, 1996.

[106] Jennifer K Holman-Dodds, A Allen Bradley, and Kenneth W Potter. Evaluation of hydrologic benefits of infiltration based urban storm water management. *Journal of the American Water Resources Association*, 39(1):205–215, 2003.

[107] Horizons, Cambridgeshire. Cambridgeshire green infrastructure strategy. *Histon: Cambridgeshire Horizons*, 2011.

[108] Jay Hoskins. MEP Spreadsheet and User Instructions. Accessed at `https://www.stlmsd.com/sites/default/files/engineering/442676.PDF`, 2012.

[109] Jay Hoskins, Trent Stober, David Carani, and John Christiansen. Strategy to Yield Sound Bacteria TMDLs and Implementable Load Reductions in St. Louis, Missouri. *Proceedings of the Water Environment Federation*, 2017(6):4783–4786, 2017.

[110] Robert W Howarth, Andrew Sharpley, and Dan Walker. Sources of nutrient pollution to coastal waters in the United States: Implications for achieving coastal water quality goals. *Estuaries*, 25(4):656–676, 2002.

[111] T.C. Hrubec. Nitrate toxicity: a potential problem of recirculating systems. *Proceedings of the Aquaculture Engineering Society*, pages 41–48, 1996.

[112] Wayne C Huber, Robert E Dickinson, Thomas O Barnwell Jr, and Assessment Branch. Storm Water Management Model; Version 4. *Environmental Protection Agency, United States*, 1988.

[113] Illinois Dept of Agriculture, Illinois Environmental Protection Agency, and University of Illinois at Urbana-Champaign. Illinois Nutrient Loss Reduction Strategy, 2015.

[114] Impact Infrastructure, LLC. Autocase. Accessed at `https://www.autocase.com/`, 2017.

[115] Impact Infrastructure, LLC and Stantec. AutoCASE Beta Testing Project Evaluation of GI/LID Benefits in the Pima County Environment Final Report. Prepared for The Pima County Regional Flood Control District and Pima Association of Governments with the Cooperation of the City of Tucson, 2014.

[116] Institute for Sustainable Infrastructure. Envision. Accessed at `https://sustainableinfrastructure.org/`, 2018.

[117] Iowa Department of Natural Resources. State forest nursery. Accessed at `http://www.iowadnr.gov/Conservation/Forestry/State-Forest-Nursery`, 2018.

[118] Iowa Dept of Agriculture & Land Stewardship, Iowa Dept of Natural Resources, and Iowa State University College of Agriculture and Life Sciences. Iowa Nutrient Reduction Strategy A science and technology-based framework to assess and reduce nutrients to Iowa waters and the Gulf of Mexico, September 2016.

[119] Iowa State University Extenstion. Estimated costs of crop production in Iowa - 2017. Accessed at `https://store.extension.iastate.edu/Product/Estimated-Costs-of-Crop-Production-in-Iowa-2017`, 2017.

[120] Martin Jaffe. Environmental reviews & case studies: reflections on Green Infrastructure economics. *Environmental Practice*, 12(4):357–365, 2010.

[121] Martin Jaffe, Moira Zellner, Emily Minor, Miquel Gonzalez-Meler, Lisa Cotner, Dean Massey, Hala Ahmed, Megan Elberts, Hal Sprague, and Steve Wise. Using green infrastructure to manage urban stormwater quality: a review of selected practices and state programs. *Illinois Environmental Protection Agency*, page 1, 2010.

[122] VM Jayasooriya and AWM Ng. Tools for modeling of stormwater management and economics of green infrastructure practices: a review. *Water, Air, & Soil Pollution*, 225(8):2055, 2014.

[123] Viniece Jennings, Cassandra Johnson Gaither, and Richard Schulterbrandt Gragg. Promoting environmental justice through urban green space access: A synopsis. *Environmental Justice*, 5(1):1–7, 2012.

[124] M.K. Jha, P.W. Gassman, and J.G. Arnold. Water quality modeling for the Raccoon River Watershed using SWAT. *Transactions of the ASABE*, 50(2):479–493, 2007.

[125] Christopher S Jones, Caroline A Davis, Chad W Drake, Keith E Schilling, Samuel HP Debionne, Daniel W Gilles, Ibrahim Demir, and Larry J Weber. Iowa statewide stream nitrate load calculated using in situ sensor network. *Journal of the American Water Resources Association*, pages 1–16, 2017.

[126] Cy Jones, Evan Branosky, Mindy Selman, and Michelle Perez. How nutrient trading could help restore the Chesapeake Bay. *World Resources Institute Working Paper*, 2010.

[127] Matthew P Jones and William F Hunt. Bioretention impact on runoff temperature in trout sensitive waters. *Journal of Environmental Engineering*, 135(8):577–585, 2009.

[128] Carol Kambites and Stephen Owen. Renewed prospects for green infrastructure planning in the UK. *Planning, Practice & Research*, 21(4):483–496, 2006.

[129] Christopher Kaminker, Osamu Kawanishi, Fiona Stewart, Ben Caldecott, and Nicholas Howarth. Institutional investors and green infrastructure investments: selected case studies. *OECD Working Papers on Finance, Insurance and Private Pensions*, (35):1, 2013.

[130] Rachel Kaplan. The role of nature in the context of the workplace. *Landscape and Urban Planning*, 26(1-4):193–201, 1993.

[131] Andrew Karvonen. *Politics of urban runoff: nature, technology, and the sustainable city.* MIT Press, 2011.

[132] Zach Kaufman, David Abler, James Shortle, Jayson Harper, James Hamlett, and Peter Feather. Agricultural costs of the Chesapeake Bay total maximum daily load. *Environmental Science & Technology*, 48(24):14131–14138, 2014.

[133] Aleksandra Kazmierczak and Jeremy Carter. Adaptation to climate change using green and blue infrastructure. a database of case studies. 2010.

[134] Melissa Keeley. Using individual parcel assessments to improve stormwater management. *Journal of the American Planning Association*, 73(2):149–160, 2007.

[135] Julia Kennedy, Peter Haas, and Bill Eyring. Measuring the economic impacts of greening: The Center for Neighborhood Technology Green Values Calculator. *Growing Greener Cities: Urban Sustainability in the Twenty-First Century*, pages 326–345, 2008.

[136] Ruben Kertesz, Olivia Odom Green, and William D Shuster. Modeling the hydrologic and economic efficacy of stormwater utility credit programs for US single family residences. *Water Science and Technology*, 70(11):1746–1754, 2014.

[137] David I Kertzer and Dominique Arel. *Census and identity: The politics of race, ethnicity, and language in national censuses*, volume 1. Cambridge University Press, 2002.

[138] Omar Khader and Franco A Montalto. Development and calibration of a high resolution SWMM model for simulating the effects of LID retrofits on the outflow hydrograph of a dense urban watershed. In *Low Impact Development for Urban Ecosystem and Habitat Protection*, pages 1–9. 2009.

[139] Myung Hee Kim, Chan Yong Sung, Ming-Han Li, and Kung-Hui Chu. Bioretention for stormwater quality improvement in Texas: Removal effectiveness of Escherichia coli. *Separation and Purification Technology*, 84:120–124, 2012.

[140] Michelle C Kondo, Sarah C Low, Jason Henning, and Charles C Branas. The impact of green stormwater infrastructure installation on surrounding health and safety. *American Journal of Public Health*, 2015.

[141] Frances E Kuo. Social aspects of urban forestry: the role of arboriculture in a healthy social ecology. *Journal of Arboriculture*, 29(3):148–155, 2003.

[142] Raffaele Lafortezza, Giuseppe Carrus, Giovanni Sanesi, and Clive Davies. Benefits and well-being perceived by people visiting green spaces in periods of heat stress. *Urban Forestry & Urban Greening*, 8(2):97–108, 2009.

[143] Fu-hsiung Lai, Ting Dai, Jenny Zhen, John Riverson, Khalid Alvi, and Leslie Shoemaker. SUSTAIN-An EPA BMP process and placement tool for urban watersheds. *Proceedings of the Water Environment Federation*, 2007(5):946–968, 2007.

[144] Joong Gwang Lee, Ariamalar Selvakumar, Khalid Alvi, John Riverson, Jenny X Zhen, Leslie Shoemaker, and Fu-hsiung Lai. A watershed-scale design optimization model for stormwater best management practices. *Environmental Modelling & Software*, 37: 6–18, 2012.

[145] Raul P Lejano and Hiroyuki Iseki. Environmental justice: spatial distribution of hazardous waste treatment, storage and disposal facilities in Los Angeles. *Journal of Urban Planning and Development*, 127(2):51–62, 2001.

[146] Conor Lewellyn, Cara E Lyons, Robert G Traver, and Bridget M Wadzuk. Evaluation of seasonal and large storm runoff volume capture of an infiltration green infrastructure system. *Journal of Hydrologic Engineering*, 21(1):04015047, 2015.

[147] W.L. Liang, K. Kosugi, and T. Mizuyama. A three-dimensional model of the effect of stemflow on soil water dynamics around a tree on a hillslope. *Journal of Hydrology*, 366(1–4):62–75, 2009.

[148] Lawrence W. Libby. Paying the nonpoint pollution control bill. *Journal of Soil and Water Conservation*, 40(1):33–36, 1985.

[149] Yaoze Liu, Laurent M Ahiablame, Vincent F Bralts, and Bernard A Engel. Enhancing a rainfall-runoff model to assess the impacts of BMPs and LID practices on storm runoff. *Journal of Environmental Management*, 147:12–23, 2015.

[150] Louisiana Universities Marine Consortium. What is hypoxia? url-http://www.gulfhypoxia.net/Overview/, 2017.

[151] Louisiana Universities Marine Consortium. Press Release Summary. eds. Nancy N. Rabalais and R. Eugene Turner, August 4, 2015.

[152] Richard Lowrance, Ralph Leonard, and Joseph Sheridan. Managing riparian ecosystems to control nonpoint pollution. *Journal of Soil and Water Conservation*, 40(1): 87–91, 1985.

[153] William C Lucas. Delaware Urban Runoff Management Model: Hydrology and Hydraulics. In *World Water & Environmental Resources Congress 2003*, pages 1–10, 2003.

[154] William C Lucas. Developing an effective urban runoff management approach. In *Impacts of Global Climate Change*, pages 1–12. 2005.

[155] Robb Lukes and Christopher Kloss. Managing wet weather with green infrastructure, Municipal handbook, Green streets, Low impact development center. Technical report, EPA-833-F-08-009. http://water. epa. gov/infrastructure/greeninfrastructure/upload/gi_munichandbook_green_streets. pdf (accessed 16 October 2015), 2010.

[156] D. Lutz. Water quality studies: Red Rock and Saylorville reservoirs, Des Moines River, Iowa. Annual Report. Ames, Iowa: Iowa State University, Department of Civil, Construction, and Environmental Engineering, Engineering Research Institute. Accessed at `www.ccee.iastate.edu/reserach/lutz/homepage.html`, 2005.

[157] Floyd J Malveaux and Sheryl A Fletcher-Vincent. Environmental risk factors of childhood asthma in urban centers. *Environmental Health Perspectives*, 103(Suppl 6):59, 1995.

[158] Andrew R Maroko, Juliana A Maantay, Nancy L Sohler, Kristen L Grady, and Peter S Arno. The complexities of measuring access to parks and physical activity sites in New York City: a quantitative and qualitative approach. *International Journal of Health Geographics*, 8(1):34, 2009.

[159] Paul M Mayer, Steven K Reynolds, Marshall D McCutchen, and Timothy J Canfield. Meta-analysis of nitrogen removal in riparian buffers. *Journal of Environmental Quality*, 36(4):1172–1180, 2007.

[160] Greg McComb, Van Lantz, Katrina Nash, and Robyn Rittmaster. International valuation databases: overview, methods and operational issues. *Ecological Economics*, 60 (2):461–472, 2006.

[161] Richard H McCuen and Glenn E Moglen. Multicriterion stormwater management methods. *Journal of Water Resources Planning and Management*, 114(4):414–431, 1988.

[162] Arthur E McGarity. Screening optimization model for watershed-based management of urban runoff nonpoint pollution. *US Environmental Protection Agency, Office of Wetlands, Oceans, and Watersheds, final report for project AW-83238401-0*, 2006.

[163] Arthur E McGarity. Watershed-based optimal stormwater management: part 1application of StormWISE to Little Crum Creek in Suburban Philadelphia. In *World Environmental and Water Resources Congress 2010: Challenges of Change*, pages 2502–2512, 2010.

[164] Arthur E McGarity. Storm-water investment strategy evaluation model for impaired urban watersheds. *Journal of Water Resources Planning and Management*, 138(2): 111–124, 2011.

[165] Timothy Meinch. Water Works to end direct dumping of nitrates into river. Accessed at `https://www.desmoinesregister.com/story/news/2016/02/24/water-works-end-direct-dumping-nitrates-into-river/80811354/`, 2016.

[166] Metropolitan St. Louis Sewer District. Maximum Extent Practicable Tool. Accessed at `http://www.stlmsd.com/what-we-do/stormwater-management/bmp-toolbox/calculation-and-report-preparation-tools`, 2016.

[167] Minnesota Pollution Control Agency, Minnesota Dept of Agriculture, Minnesota Board of Water & Soil Resources, and University of Minnesota. The Minnesota Nutrient Reduction Strategy, September 2014.

[168] Bruce C Mitchell and Jayajit Chakraborty. Landscapes of thermal inequity: disproportionate exposure to urban heat in the three largest US cities. *Environmental Research Letters*, 10(11):115005, 2015.

[169] Bruce Coffyn Mitchell and Jayajit Chakraborty. Urban heat and climate justice: a landscape of thermal inequity in Pinellas County, Florida. *Geographical Review*, 104 (4):459–480, 2014.

[170] Richard Mitchell and Frank Popham. Effect of exposure to natural environment on health inequalities: an observational population study. *The Lancet*, 372(9650):1655–1660, 2008.

[171] Franco Montalto, Christopher Behr, Katherine Alfredo, Max Wolf, Matvey Arye, and Mary Walsh. Rapid assessment of the cost-effectiveness of low impact development for CSO control. *Landscape and Urban Planning*, 82(3):117–131, 2007.

[172] Franco A Montalto, Timothy A Bartrand, Alexander M Waldman, Katharine A Travaline, Charles H Loomis, Chariss McAfee, Juliet M Geldi, Gavin J Riggall, and Laureen M Boles. Decentralised green infrastructure: the importance of stakeholder behaviour in determining spatial and temporal outcomes. *Structure and Infrastructure Engineering*, 9(12):1187–1205, 2013.

[173] I.D. Moore, R.B. Grayson, and A.R. Larson. Digital terrain modeling: a review of hydrological, geomorphological, and biological applications. *Hydrologic Processes*, 5: 3–30, 1991.

[174] R Moore, DL Spittlehouse, and Anthony Story. Riparian microclimate and stream temperature response to forest harvesting: a review. *Journal of the American Water Resources Association*, 41(4):813–834, 2005.

[175] Adrián Morales-Torres, Ignacio Escuder-Bueno, Ignacio Andrés-Doménech, and Sara Perales-Momparler. Decision Support Tool for energy-efficient, sustainable and integrated urban stormwater management. *Environmental Modelling & Software*, 84: 518–528, 2016.

[176] Rachel Morello-Frosch, Manuel Pastor, and James Sadd. Environmental justice and Southern Californias riskscape the distribution of air toxics exposures and health risks among diverse communities. *Urban Affairs Review*, 36(4):551–578, 2001.

[177] National Research Council. *Urban stormwater management in the United States*. National Academies Press, 2009.

[178] Kären C Nelson and Margaret A Palmer. Stream temperature surges under urbanization and climate change: data, models, and responses. *Journal of the American Water Resources Association*, 43(2):440–452, 2007.

[179] New York City Department of Environmental Protection. Guidelines for the Design and Construction of Stormwater Management Systems. Accessed at `http://www.nyc.gov/html/dep/pdf/green_infrastructure/stormwater_guidelines_2012_final.pdf`, 2012.

[180] New York City Department of Environmental Protection. Stormwater Management System Calculator. Accessed at `http://www.nyc.gov/html/dep/html/stormwater/stormwater_management_construction.shtml`, 2012.

[181] David J Nowak, Daniel E Crane, and Jack C Stevens. Air pollution removal by urban trees and shrubs in the United States. *Urban Forestry & Urban Greening*, 4(3-4): 115–123, 2006.

[182] Ocean Studies Board and National Research Council. *Clean coastal waters: understanding and reducing the effects of nutrient pollution*. National Academies Press, 2000.

[183] M. Olszewska and H. Smal. The effect of afforestation with scots pine (*Pinus sylvestris L.*) of sandy post-arable soils on their selected properties. I. Physical and sorptive properties. *Plant and Soil*, 305(1–2):157–169, 2008.

[184] Marie S O'Neill, Michael Jerrett, Ichiro Kawachi, Jonathan I Levy, Aaron J Cohen, Nelson Gouveia, Paul Wilkinson, Tony Fletcher, Luis Cifuentes, and Joel Schwartz. Health, wealth, and air pollution: advancing theory and methods. *Environmental Health Perspectives*, 111(16):1861, 2003.

[185] L.L. Osborne and D.A. Kovacic. Riparian vegetated buffer strips in water-quality restoration and stream management. *Freshwater Biology*, 29(2):243–258, 1993.

[186] Osenbaugh's Prairie Seed Farms. Prairie Seed Farms - Prairie Grasses. Accessed at `http://prairieseedfarms.com/prairie-seed-farms-prairie-grass-seed/prairie-seed-farms-prairie-grass-seed.html`, 2018.

[187] Peter J Parks and Ian W Hardie. Least-cost forest carbon reserves: cost-effective subsidies to convert marginal agricultural land to forests. *Land Economics*, pages 122–136, 1995.

[188] Michelle Perez, Sara Walker, and Cy Jones. Nutrient trading in the MRB: A feasibility study using large-scale interstate nutrient trading in the Mississippi River Basin to help address Hypoxia in the Gulf of Mexico. *World Resources Institute Report*, 2013.

[189] W.T. Peterjohn and D.L. Correll. Nutrient dynamics in an agricultural watershed: observations on the role of a riparian forest. *Ecology*, 65(5):1466–1475, 1984.

[190] Philadelphia City Planning Commission. Design Guidelines for Off-Street Parking (Zoning Code §14-1400), 2010.

[191] Philadelphia Water Department. Green City, Clean Waters: The City of Philadelphia's Program for Combined Sewer Overflow Control. Accessed at `http://www.phillywatersheds.org/doc/GCCW_AmendedJune2011_LOWRES-web.pdf`, 2011.

[192] Amy C Poe, Michael F Piehler, Suzanne P Thompson, and Hans W Paerl. Denitrification in a constructed wetland receiving agricultural runoff. *Wetlands*, 23(4):817–826, 2003.

[193] D.S. Powlson, T.M. Addiscott, N. Benjamin, K.G. Cassman, T.M. de Kok, H. van Grinsven, J. L'hirondel, A.A. Avery, and C. van Kessel. When does nitrate become a risk for humans? *Journal of Environmental Quality*, 37:291–295, 2008.

[194] S. Rabotyagov, T. Campbell, M. Jha, P.W. Gassman, J. Arnold, L. Kurkalova, S. Secchi, H. Feng, and C.L. Kling. Least-cost control of agricultural nutrient contributions to the Gulf of Mexico hypoxic zone. *Ecological Applications*, 20(6):1542–1555, 2010.

[195] N.S. Rao, Z.M. Easton, E.M. Schneiderman, M.S. Zion, D.R. Lee, and T.S. Steenhuis. Modelling watershed-scale effectiveness of agricultural best management practices to reduce phosphorus loading. *Journal of Environmental Management*, 90(3):1385–1395, 2009.

[196] Robert Raucher and Janet Clements. A triple bottom line assessment of traditional and green infrastructure options for controlling CSO events in Philadelphia's watersheds. *Proceedings of the Water Environment Federation*, 2010(9):6776–6804, 2010.

[197] Daryll E. Ray and Harwood D. Schaffer. Des Moines' water-quality suit is based on drainage-tile water not stormwater. Accessed at `http://www.farmandranchguide.com/news/opinion/des-moines-water-quality-suit-is-based-on-drainagetile/article.html`, August 5, 2015.

[198] T.P. Redick and C. Brooks. WOTUS Wars and Endangered Species: Where Will Farmers Find Their Legal High Ground? *Natural Resources and Environment*, 31(2): 20, 2016.

[199] Shannon K Reynolds, Christine A Pomeroy, A Charles Rowney, and Chris M Rowney. Linking stormwater BMP systems water quality and quantity performance to whole life cycle cost to improve BMP selection and design. In *World Environmental and Water Resources Congress 2012: Crossing Boundaries*, pages 2320–2328, 2012.

[200] Jenny J Roe, Catharine Ward Thompson, Peter A Aspinall, Mark J Brewer, Elizabeth I Duff, David Miller, Richard Mitchell, and Angela Clow. Green space and stress: evidence from cortisol measures in deprived urban communities. *International Journal of Environmental Research and Public Health*, 10(9):4086–4103, 2013.

[201] Kenton Rogers, Keith Sacre, Jessica Goodenough, and Kieron Doick. Valuing London's urban forest: results of the London i-Tree eco project. 2015.

[202] Cynthia Rosenzweig, William D Solecki, Lily Parshall, Barry Lynn, Jennifer Cox, Richard Goldberg, Sara Hodges, Stuart Gaffin, Ronald B Slosberg, Peter Savio, et al. Mitigating New York City's heat island: Integrating stakeholder perspectives and scientific evaluation. *Bulletin of the American Meteorological Society*, 90(9):1297–1312, 2009.

[203] Lewis A Rossman. *Storm Water Management Model User's Manual, Version 5.0.* National Risk Management Research Laboratory, Office of Research and Development, US Environmental Protection Agency Cincinnati, 2010.

[204] Allison H Roy, Lee K Rhea, Audrey L Mayer, William D Shuster, Jake J Beaulieu, Matthew E Hopton, Matthew A Morrison, and Ann St Amand. How much is enough? Minimal responses of water quality and stream biota to partial retrofit stormwater management in a suburban neighborhood. *Public Library of Science*, 9(1):e85011, 2014.

[205] F Sabouri, B Gharabaghi, AA Mahboubi, and EA McBean. Impervious surfaces and sewer pipe effects on stormwater runoff temperature. *Journal of Hydrology*, 502:10–17, 2013.

[206] Oshadhi Samarasinghe, Adam Daigneault, Suzie Greenhalgh, Oscar Montes de Oca Munguia, and Jill Walcroft. Impacts of Farmer Attitude on the Design of a Nutrient Reduction Policy–a New Zealand Catchment Case Study. In *Proceedings: 56th Australian Agricultural and Resource Economics Society (AARES) Conference, Fremantle, Western Australia*, page 62, 2012.

[207] Alexis Schäffler and Mark Swilling. Valuing green infrastructure in an urban environment under pressure — The Johannesburg Case. *Ecological Economics*, 86:246–257, 2013.

[208] K.E. Schilling and Y.K. Zhang. Baseflow contribution to nitrate-nitrogen export from a large, agricultural watershed. *Journal of Hydrology*, 295:305–316, 2004.

[209] TJ Schmitt, MG Dosskey, and KD Hoagland. Filter strip performance and processes for different vegetation, widths, and contaminants. *Journal of Environmental Quality*, 28(5):1479–1489, 1999.

[210] Miklas Scholz and Sara Kazemi Yazdi. Treatment of road runoff by a combined storm water treatment, detention and infiltration system. *Water, Air, and Soil Pollution*, 198(1-4):55–64, 2009.

[211] Jon E Schoonover and Karl WJ Williard. Ground water nitrate reduction in giant cane and forest riparian buffer zones. *Journal of the American Water Resources Association*, 39(2):347–354, 2003.

[212] Jon E Schoonover, B Graeme Lockaby, and Brian S Helms. Impacts of land cover on stream hydrology in the west Georgia piedmont, USA. *Journal of Environmental Quality*, 35(6):2123–2131, 2006.

[213] E. Schram, J.A.C. Roques, W. Abbink, Y. Yokohama, T. Spanings, P. de vries, S. Bierman, H. van de Vis, and G. Flik. The impact of elevated water nitrate concentration on physiology, growth and feed intake of African catfish *Clarias gariepinus* (Burchell 1822). *Aquaculture Research*, 2012.

[214] Thomas Schueler. Impacts of impervious cover on aquatic systems. *Center for Watershed Protection. Ellicott City, MD*, 2003.

[215] RC Schultz, JP Collettil, TM Isenhart, WW Simpkins, CW Mize, and ML Thompson. Design and placement of a multi-species riparian buffer strip system. *Agroforestry Systems*, 29(3):201–226, 1995.

[216] R.C. Schultz, T.M. Isenhart, W.W. Simpkins, and J.P. Colletti. Riparian forest buffers in agroecosystems – lessons learned from the Bear Creek watershed, central Iowa, USA. *Agroforestry Systems*, 61(1):35–50, 2004.

[217] Anas Sebti, Musandji Fuamba, and Saad Bennis. Optimization model for BMP selection and placement in a combined sewer. *Journal of Water Resources Planning and Management*, 142(3):04015068, 2015.

[218] Vivek Shandas, Anne Nelson, and Carine Arendes. Integrating Education, Evaluation & Partnerships into Large-Scale Sustainable Stormwater Management Programs. 2010.

[219] A.N. Sharpley, R.W. McDowell, and P.J.A. Kleinmann. Phosphorus loss from land to water: integrating agricultural and environmental management. *Plant and Soil*, 237 (2):287–307, 2001.

[220] James S. Shortle and James W. Dunn. The Relative Efficiency of Agricultural Source Water Pollution Control Policies. *American Journal of Agricultural Economics*, 68(3): 668–677, 1986.

[221] William D Shuster, Matthew A Morrison, and Rachel Webb. Front-loading urban stormwater management for success–a perspective incorporating current studies on the implementation of retrofit low-impact development. *Cities and the Environment*, 1(2):8, 2008.

[222] Chona Sister, Jennifer Wolch, and John Wilson. Got green? Addressing environmental justice in park provision. *GeoJournal*, 75(3):229–248, 2010.

[223] Neil Smith. Toward a theory of gentrification a back to the city movement by capital, not people. *Journal of the American Planning Association*, 45(4):538–548, 1979.

[224] Sverker Sörlin. Monument and memory: landscape imagery and the articulation of territory. *Worldviews: Global Religions, Culture, and Ecology*, 2(3):269–279, 1998.

[225] N.E. Spahr, N.M. Dubrovsky, J.M. Gronberg, O.L. Franke, and D.M. Wolock. Nitrate loads and concentrations in surface-water base flow and shallow groundwater for selected basins in the United States, water years 19902006: U.S. Geological Survey scientific investigations report 20105098, 2010.

[226] Sabrina Spatari, Ziwen Yu, and Franco A Montalto. Life cycle implications of urban green infrastructure. *Environmental Pollution*, 159(8-9):2174–2179, 2011.

[227] Mai Stafford, David Gimeno, and Michael G Marmot. Neighbourhood characteristics and trajectories of health functioning: a multilevel prospective analysis. *The European Journal of Public Health*, 18(6):604–610, 2008.

[228] Kurt Stephenson and Leonard Shabman. Where did the agricultural nonpoint source trades go? Lessons from Virginia water quality trading programs. *Journal of the American Water Resources Association*, 53(5):1178–1194, 2017.

[229] Paul C Stern. Information, incentives, and proenvironmental consumer behavior. *Journal of Consumer Policy*, 22(4):461–478, 1999.

[230] Paul C Stern. Towards a coherent theory of environmentally significant behavior. *Journal of Social Issues*, (56), 2000.

[231] Z. Sugg. Assessing U.S. farm drainage: Can GIS lead to better estimates of subsurface drainage extent? World Resources Institute Report, 2007.

[232] R Swoboda. Are you applying too much N? *Wallaces Farmer*, 1011, 1990.

[233] Hideki Takebayashi and Masakazu Moriyama. Surface heat budget on green roof and high reflection roof for mitigation of urban heat island. *Building and Environment*, 42 (8):2971–2979, 2007.

[234] Z Tang, BA Engel, BC Pijanowski, and KJ Lim. Forecasting land use change and its environmental impact at a watershed scale. *Journal of Environmental Management*, 76(1):35–45, 2005.

[235] David Tarboton. Taudem version 5. Accessed at `http://hydrology.usu.edu/taudem/taudem5/downloads.html`, 2017.

[236] D.G. Tarboton. A new method for the determination of flow directions and upslope areas in grid digital elevation models. *Water Resources Research*, 33:309–319, 1997.

[237] The Pennsylvania Geospatial Data Clearinghouse. Pennsylvania Major Rivers. Accessed at `http://www.pasda.psu.edu/uci/DataSummary.aspx?dataset=94`, 1996.

[238] The Pennsylvania Geospatial Data Clearinghouse. Philadelphia Green Stormwater Infrastructure Projects. Accessed at `http://www.pasda.psu.edu/uci/DataSummary.aspx?dataset=1574`, 2013.

[239] Theodore G Theodosiou. Summer period analysis of the performance of a planted roof as a passive cooling technique. *Energy and Buildings*, 35(9):909–917, 2003.

[240] D. Tilman, C. Balzer, J. Hill, and B.L. Befort. Global food demand and the sustainable intensification of agriculture. *Proceedings of the National Academy of Sciences*, 108 (50):20260–20264, 2011.

[241] M.D. Tomer, D.E. James, and T.M. Isenhart. Optimizing the placement of riparian practices in a watershed using terrain analysis. *Journal of Soil and Water Conservation*, 58(4):198–206, 2003.

[242] M.D. Tomer, M.G. Dosskey, M.R. Burkart, D.E. James, M.J. Helmers, and D.E. Eisenhauer. Methods to prioritize placement of riparian buffers for improved water quality. *Agroforestry Systems*, 75:17–25, 2009.

[243] Vassilios A Tsihrintzis and Rizwan Hamid. Runoff quality prediction from small urban catchments using SWMM. *Hydrological Processes*, 12(2):311–329, 1998.

[244] Konstantinos Tzoulas, Kalevi Korpela, Stephen Venn, Vesa Yli-Pelkonen, Aleksandra Kaźmierczak, Jari Niemela, and Philip James. Promoting ecosystem and human health in urban areas using Green Infrastructure: A literature review. *Landscape and Urban Planning*, 81(3):167–178, 2007.

[245] United States Environmental Protection Agency. National Primary Drinking Water Standards. Accessed at `https://www.epa.gov/ground-water-and-drinking-water/national-primary-drinking-water-regulations`, 2018.

[246] United States Geological Survey. The national map. Accessed at `https://viewer.nationalmap.gov/launch/`, 2018.

[247] USDA-NRCS. National resources inventory: A statistical survey of land use and natural resource conditions and trends on u.s. non-federal lands. Washington, D.C.: USDA Natural Resources Conservation Service. Available at: `https://water.usgs.gov/GIS/metadata/usgswrd/XML/`, 2006.

[248] Agnes E Van den Berg, Terry Hartig, and Henk Staats. Preference for nature in urbanized societies: Stress, restoration, and the pursuit of sustainability. *Journal of Social Issues*, 63(1):79–96, 2007.

[249] Ma D Velarde, Gareth Fry, and Mari Tveit. Health effects of viewing landscapes–Landscape types in environmental psychology. *Urban Forestry & Urban Greening*, 6 (4):199–212, 2007.

[250] George Vellidis, Robert Lowrance, P Gay, and RK Hubbard. Nutrient transport in a restored riparian wetland. *Journal of Environmental Quality*, 32(2):711–726, 2003.

[251] Ferdinando Villa, Matthew A Wilson, Rudolf de Groot, Steven Farber, Robert Costanza, and Roelof MJ Boumans. Designing an integrated knowledge base to support ecosystem services valuation. *Ecological Economics*, 41(3):445–456, 2002.

[252] Lisa A Wainger, George Van Houtven, Ross Loomis, Jay Messer, Robert Beach, and Marion Deerhake. Tradeoffs among ecosystem services, performance certainty, and cost-efficiency in implementation of the Chesapeake Bay total maximum daily load. *Agricultural and Resource Economics Review*, 42(1):196–224, 2013.

[253] Brad J Wardynski, Ryan J Winston, and William F Hunt. Internal water storage enhances exfiltration and thermal load reduction from permeable pavement in the North Carolina mountains. *Journal of Environmental Engineering*, 139(2):187–195, 2012.

[254] DR West and DR Kincer. Yield of switchgrass as affected by seeding rates and dates. *Biomass and Bioenergy*, 35(9):4057–4059, 2011.

[255] Victoria Whitford, A Roland Ennos, and John F Handley. City form and natural processindicators for the ecological performance of urban areas and their application to Merseyside, UK. *Landscape and Urban Planning*, 57(2):91–103, 2001.

[256] Frank Wilcoxon, SK Katti, and Roberta A Wilcox. Critical values and probability levels for the Wilcoxon rank sum test and the Wilcoxon signed rank test. *Selected Tables in Mathematical Statistics*, 1:171–259, 1970.

[257] RJ Winston, WF Hunt, and WG Lord. Thermal mitigation of urban storm water by level spreader–vegetative filter strips. *Journal of Environmental Engineering*, 137(8): 707–716, 2011.

[258] Steven Wise, Joseph Braden, Danielle Ghalayini, Joseph Grant, Chris Kloss, Edward MacMullan, Stephanie Morse, Franco Montalto, Dan Nees, and David Nowak. Integrating valuation methods to recognize green infrastructure's multiple benefits. In *Low Impact Development 2010: Redefining Water in the City*, pages 1123–1143. 2010.

[259] Jennifer Wolch, John P Wilson, and Jed Fehrenbach. Parks and park funding in Los Angeles: An equity-mapping analysis. *Urban Geography*, 26(1):4–35, 2005.

[260] Jennifer R Wolch, Jason Byrne, and Joshua P Newell. Urban green space, public health, and environmental justice: The challenge of making cities just green enough. *Landscape and Urban Planning*, 125:234–244, 2014.

[261] Kathleen L Wolf, Marcus K Measells, Stephen C Grado, and Alicia ST Robbins. Economic values of metro nature health benefits: A life course approach. *Urban Forestry & Urban Greening*, 14(3):694–701, 2015.

[262] Nyuk Hien Wong, Yu Chen, Chui Leng Ong, and Angelia Sia. Investigation of thermal benefits of rooftop garden in the tropical environment. *Building and Environment*, 38 (2):261–270, 2003.

[263] Nyuk Hien Wong, DK W Cheong, H Yan, J Soh, CL Ong, and A Sia. The effects of rooftop garden on energy consumption of a commercial building in Singapore. *Energy and Buildings*, 35(4):353–364, 2003.

[264] Tony HF Wong, Tim D Fletcher, Hugh P Duncan, John R Coleman, and Graham A Jenkins. A Model for Urban Stormwater Improvement: Conceptualization. In *Global Solutions for Urban Drainage*, pages 1–14. 2002.

[265] Tony HF Wong, Tim D Fletcher, Hugh P Duncan, and Graham A Jenkins. Modelling urban stormwater treatment — A unified approach. *Ecological Engineering*, 27(1): 58–70, 2006.

[266] Qingfu Xiao, E Gregory McPherson, James R Simpson, Susan L Ustin, et al. Rainfall interception by Sacramento's urban forest. *Journal of Arboriculture*, 24:235–244, 1998.

[267] Ziwen Yu, Miguel Aguayo, Franco Montalto, Michael Piasecki, and Christopher Behr. Developments in LIDRA 2.0: a planning level assessment of the cost-effectiveness of low impact development. In *World Environmental and Water Resources Congress 2010: Challenges of Change*, pages 3261–3270, 2010.