ASSESSMENT OF URBANIZATION IMPACTS ON SURFACE RUNOFF AND EFFECTS OF GREEN INFRASTRUCTURE ON HYDROLOGY AND WATER QUALITY

by

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ABSTRACT

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Urbanization is one of the most important anthropogenic modifications of the global environment. It has significant impacts on hydrologic processes and water quantity and quality. Research related to urbanization impacts on surface runoff has focused on changes up to the watershed scale. However, quantitative assessment at a national scale is scarce. Parallel to urbanization impacts, climate is also one of the greatest challenges facing humanity today and its effects are already being felt from strengthened storms and rising sea levels to changing temperature and weather patterns. The adverse impacts of urbanization and climate can converge in synergistic ways, which may render hundreds of millions of urban residents increasingly vulnerable to floods, landslides and other natural disasters. The challenge is in finding mitigating solutions.

The specific objectives of this study were to: 1) assess urbanization impacts on surface runoff of the contiguous United States using the Long-Term Hydrologic Impact Assessment (L-THIA) model (version of L-THIA Tabular Tool) based on the 2001, 2006, and 2011 National Land Cover Databases; 2) assess simulated precipitation based on an updated CLIGEN database and associated impacts on the surface runoff using the L-THIA Tabular Tool in the five states of the Great Lakes Region (WI, IL, IN, MI, and OH); and 3) evaluate the effectiveness of green infrastructure practices on hydrology and water quality in a Combined Sewer Overflows (CSO) community.

National analysis results showed that: 1) urbanization occurred non-uniformly across the nation from 2001 to 2011; 2) urban expansion and intensification were the main driving forces altering surface runoff; 3) the majority of counties had long-term (50 years) normalized average annual runoff depth (NAARD) from urban land less than 17.8 mm; 4) the states with the largest NAARD values had both high precipitation and increases in urban land, while the ten states with the largest NAARD change percentages were mainly in the western U.S. with low precipitation and the NAARD values were mainly influenced by large increases in urban land; 5) nationally, average annual runoff increased by 10% (approximately 3.3 billion m^3) due to urbanization from 2001 to 2011.

Weather generators rely on historical meteorological records to simulate time series of synthetic weather inputs, the quality of which has direct influence on model applications. The weather generator CLIGEN's database has recently been updated to comprise consistent historical records from 1974 to 2013 (updated CLIGEN database, UCD) compared to the current database in which records are of different lengths. In the second objective, CLIGEN's performance in estimating precipitation using UCD and the subsequent impact on urban runoff simulations were evaluated. Generally, UCD-based precipitation could replicate observed daily precipitation up to the 99.5th percentile, but maximum precipitation was underestimated. Results from the Long-Term Hydrologic Impact Assessment model using UCD-based precipitation showed about 0.57 billion cubic meters more (14.9%) average annual runoff being simulated compared with simulations based on the current CLIGEN database. Overall, CLIGEN with the updated database was found suitable for providing precipitation estimates and for use with modeling urban runoff or urbanization effects.

From Objective 3, the enhanced L-THIA-LID 2.2 model was able to simulate more detailed impervious surfaces including sidewalks, roads, driveways, and parking lots, to conduct cost calculations for these more detailed impervious surfaces, and to consider the actual suitable area for bioretention in the study area. The effectiveness of green infrastructure (GI) practices on hydrology and water quality at a combined sewer overflow urban watershed was examined in 10 simulation scenarios using 8 practices including rain barrels/cisterns, green roofs, green roofs plus rain barrels/cisterns, bioretention, porous roads, porous parking lots, porous sidewalks, and porous driveways. The annual cost and the cost effectiveness for each scenario considering a 20-year GI practices lifetime were examined. Main findings included: (1) combined implementation of GI practices performed better than applying individual practices alone; (2) the various adoption levels and combinations of GI practices could potentially reduce runoff volume by 0.2-23.5%, TSS by 0.18-30.8%, TN by 0.2-27.9%, and TP by 0.20 to 28.1%; (3) based on site characteristics, adding more GI practices did not necessarily mean that substantial runoff and pollutant reduction would be achieved; (4) the most cost-effective scenario had an associated cost of \$9.21 to achieve 1 m^3 runoff reduction per year and \$119 to achieve 1 kg TSS reduction per year. This, however, assumes cooperation from the general public in implementing GI practices on their properties; and, (5) adoption of GI practices on all possible areas could potentially achieve the greatest runoff and pollutant load reduction, but would not be the most cost-effective option. The enhanced model from this research can be applied to various locations to support assessing the beneficial uses of GI practices.

Overall, results of this research provide important information on the negative impacts of urbanization and climate, as well as the importance of green infrastructure as a sustainable development or re-development approach. This research also demonstrated the technical capability of the L-THIA Tabular Tool and the L-THIA-LID 2.2 model. The outcomes of this research will assist urban planners and decision makers to make sustainable development or re-development strategies.

CHAPTER 1. INTRODUCTION

1.1 Problem Statement

Global urban population increased from 220 million to 2.8 billion during the 20th century, yet unprecedented urban growth is anticipated over the next few decades with urban populations expected to swell to almost 5 billion by 2030 (UNFPA, 2007). Parallel to this, climate change is a major challenge facing humanity today and its effects are already being felt in terms of strengthened storms and rising sea levels as well as changing temperature and weather patterns (UNFPA, 2016). The adverse impacts of urbanization and climate change can converge in synergistic ways (Kacyira, 2012); although no country has ever achieved significant economic growth without urbanization, changes associated with modern civilization have resulted in serious environmental problems (UNFPA, 2007). When compounded with climate, these problems render hundreds of millions of urban residents increasingly vulnerable to floods, landslides, extreme weather and other natural disasters (Kacyira, 2012). Regardless, the potential value of cities to long-term sustainability is well recognized (UNFPA, 2007); the challenge is in finding solutions that balance trade-offs between the advantages and possible disadvantages of urbanization in relation to climate. This study aims to assess urbanization and climate impacts on surface runoff and then to evaluate the effects of green infrastructure on hydrology and water quality with a focus on the U.S. and working progressively from national to regional and local scales.

1.1.1 Urbanization and surface runoff

Urbanization is one of the most important anthropogenic modifications of the global environment (Antrop, 2004; DeFries & Eshleman, 2004; Eshleman, 2004; Foley et al., 2005; Wu, 2014). Every urban area in the United States has expanded substantially in area in recent decades (USEPA, 2013). This increase is projected to continue; for example, developed land in the U.S. is projected to increase from 5.2% to 9.2% of the total land base in the next 25 years (Alig et al., 2004); the National Land Cover Database showed that developed land in the contiguous U.S. increased in area by 20,296 km^2 from 2001 to 2011.

Land surface modifications occur during the process of urbanization, including vegetation reduction, soil compaction, and change from pervious surfaces to impervious surfaces such as roofs, roads, and parking lots (Arnold Jr & Gibbons, 1996; Booth & Jackson, 1997; Schueler, 1994). The consequences of urbanization include but are not limited to: changes in water supply from altered hydrologic processes of infiltration, groundwater recharge, and runoff; changes in water quality degradation from urban runoff and combined sewer overflows (CSO); and changes in water demands (Burns et al., 2012; DeFries & Eshleman, 2004; Passerat et al., 2011; Semadeni-Davies et al., 2008; Vietz et al., 2016).

Surface runoff and river discharge increase when natural vegetation, especially forests, decrease (Costa et al., 2003; Foley et al., 2005; Sahin & Hall, 1996). Impervious surfaces from urbanization contribute to more surface runoff due to decreased infiltration (Arnold Jr & Gibbons, 1996; Schueler, 1994; Schueler et al., 2009). Reduced infiltration leads to higher peak flows, even for short duration low intensity rainfall, and increases the risk of flooding (Bhaduri et al., 2001; Suriya & Mudgal, 2012). Urban runoff also carries non-point source pollutants, such as oil, grease, metals, and pesticides, which sometimes are not captured in a storm sewer system and then directly enter into streams and rivers during rainfall events (Arnold Jr & Gibbons, 1996; Blair et al., 2014; Schueler, 1994). Even if urban runoff is captured by the sewer system and can be conveyed to wastewater treatment plants, CSO occur in some highly urbanized areas during storm events that may cause serious water pollution problems (Bhaduri et al., 2001; Passerat et al., 2011; Semadeni-Davies et al., 2008).

1.1.2 Climate and surface runoff

Anthropogenically driven enhanced climate change is expected to continue throughout the 21st century (Karl et al., 1996; Pruski & Nearing, 2002). Karl et al.'s analysis of historical weather records indicate that precipitation increases from 1900 to 1994 were widespread within the contiguous U.S. and local increases of nearly 20% were common (Karl et al., 1996). Increased precipitation typically results in more surface runoff.

Climate inputs, including precipitation, temperature, solar radiation, and wind speed are frequently required for many environmental analyses, such as to evaluate the impacts of watershed changes on hydrology, water quality, or erosion, to design hydraulic structures, to conduct land use planning, or to assess alternative crop or range management strategies (Hanson et al., 2002; Richardson & Wright, 1984). Typical approaches to get climate inputs include accessing historical climate records and using synthetic weather generators. Weather generators are computer models with mathematical algorithms that use historical meteorological records to simulate time series of synthetic weather data that, ideally, have similar statistical properties as observed data for different spatial and temporal resolutions (Chen et al. 2010, Peleg et al., 2017). Compared to historical climate records that may be too short or may contain considerable amounts of missing data due to equipment servicing or malfunction, the output of weather generators can provide complete meteorological data for any desired period of time that can enhance continuous hydrological models (Baffaut et al., 1996; Zhang & Garbrecht, 2003). By spatially interpolating the model parameters from adjacent gauged locations, weather generators can be used to simulate daily weather records for ungauged areas (Baffaut et al., 1996; Semenov & Barrow, 1997; Zhang & Garbrecht, 2003). Moreover, weather generators can be combined with general circulation models (GCMs) for developing climate change scenarios to assess the impact of future climate change by modifying weather generator parameters (Flanagan et al., 2014; Li et al., 2011; Mullan, 2013; Pruski & Nearing, 2002; Semenov & Barrow, 1997; Wilks, 1992; Zhang, 2005; Zhang & Liu, 2005).

Several studies have concluded that the Climate Generator (CLIGEN) is a reliable weather generator (Chen & Brissette, 2014; Johnson et al., 1996). Many enhancements and quality control techniques have been applied to CLIGEN for improved reproducibility of climatic parameters (Flanagan et al., 2001; Scheele et al., 2001; Meyer et al., 2002; Meyer et al., 2008; Yu, 2000). CLIGEN has also been tested in many locations around the world, such as the U.S. (Flanagan et al., 2001; Hoomehr et al., 2016; Lim & Engel, 2003; Thomas et al., 2007; Zhang, 2013), Australia (Vaghefi & Yu, 2016), Uganda (Elliot & Arnold, 2001), Turkey (Yüksel et al., 2008), Korea (Kim et al., 2009), Germany (Al-Mukhtar et al., 2014), Chile (Lobo et al., 2015), and China (Chen et al., 2009; Fan et al., 2013).

Statistical parameterization of weather station datasets for use is a time-consuming exercise and requires access to a significant amount of historical data for any one station. Weather data parameterization has largely been limited to selected stations. Recent updates to the CLIGEN database (http://brenton.nserl.purdue.edu/cligenupdate/) provide up-to-date statistical parameterization of a majority of weather stations across the U.S. Assessment of surface runoff based on simulated precipitation from the updated CLIGEN database can also provide useful information to the public and planning agencies to assist sustainable development.

1.1.3 Developing solutions to runoff challenges related to urbanization and climate

Increased urbanization can cause several adverse impacts including but not limited to: changes in water supply from altered hydrologic processes of infiltration, groundwater recharge, and runoff; changes in water quality degradation from urban runoff and combined sewer overflows (CSO); and changes in water demands (Burns et al., 2012; DeFries & Eshleman, 2004; Passerat et al., 2011; Semadeni-Davies et al., 2008; Vietz et al., 2016). Increased precipitation typically results in more surface runoff. Dual impacts from increased urban area and increased precipitation can amplify single impacts of each one and cause serious environmental problems, such as flooding, water quality degradation, and CSO.

According to the U.S. EPA (USEPA, 2004), there are 746 communities in the U.S. with combined sewer systems with a total of 9,348 identified CSO outfalls; the EPA estimates that about 3.2 billion cubic meters (850 billion gallons) of untreated wastewater and stormwater are released from CSO each year in the U.S.; combined sewer systems are found in 32 states and CSO communities are regionally concentrated in older communities in the Northeast and Great Lakes regions. Among those 746 CSO communities, more than half of them are found in the five states of Illinois, Indiana, Ohio, Wisconsin, and Michigan (USEPA, 2008). There is an urgent need to develop solutions to mitigate runoff challenges related to adverse impacts of urbanization and climate.

Green infrastructure (GI) practices are on-site stormwater management approaches that can reduce the negative impacts of urbanization and climate on hydrology and water quality by increasing infiltration and storage, delaying runoff peak, reducing runoff rate and volume, and controlling the movement of pollutants (Benedict & McMahon, 2012; Gill et al., 2007; Tiwary & Kumar, 2014; Tzoulas et al., 2007). Large scale green infrastructure practices, also known as best management practices (BMPs) including retention ponds and detention basins, are space-intensive and usually collect and treat runoff at the drainage outlets (USEPA, 1999; USEPA, 2009). Small scale green infrastructure practices, also termed low impact development (LID) practices including porous pavements, bioretention, and green roofs, seek to control the timing and volume of stormwater discharges from impervious surfaces as well as the volume of wastewater generated by residential, commercial, and industrial land (USEPA, 2004; Di Vittorio & Ahiablame, 2015). CSO communities are usually located in dense development areas, where large scale GI practices are not always suitable. Small scale GI practices or LIDs have shown promise in increasing the infiltration and storage of stormwater and contributing to inflow control to sewer systems (USEPA, 2004).

Planning strategies for implementing GI practices, including BMPs and LIDs, occur at the sewershed/watershed scale. Sewersheds are similar to watersheds except that they occur within a city and are defined based on the sewer network. The strategies consider environmental and economic objectives (Liu et al., 2016a, 2016b). Due to site-specific complexities, the types of GI, locations where they would be implemented, and the extent of implementation would vary by sewershed. Budgetary limitations and environmental objectives necessitate a systematic approach for selecting and placing GI practices– a task often best accomplished through optimization (Gitau et al., 2004; Guoshun et al., 2013; Lee et al., 2012; Maringanti et al., 2011; Maringanti et al., 2009; McGarity, 2011). Used in combination with hydrologic/water quality models, spatial optimization algorithms can be used to generate potential GI solutions based on pre-defined objective functions (Maringanti et al., 2009).

1.1.4 Modeling approaches for assessing runoff responses to urbanization, climate, and GI implementation

Computer-based hydrological models can simulate temporal and spatial effects of hydrologic processes and management activities on water quantity and water quality (Moriasi et al., 2007). The Long-Term Hydrologic Impact Assessment (L-THIA) model is an easy, quick, and user-friendly tool, which only requires land use, hydrologic soil group (HSG), and long-term precipitation data to assess location-specific surface runoff as influenced by past, present, and future land management practices (Bhaduri et al., 2001; Grove et al., 2001; Harbor, 1994; Tang et al., 2005). The L-THIA model has been successfully used in numerous studies to assess the impact of land use change on surface hydrology and water quality (Bhaduri et al., 2000; Choi, 2007; Davis et al., 2010; Grove et al., 2001; Gunn et al., 2012; Kim et al., 2002; Lim et al., 2006a, 2006b; Lim et al., 2010; Muthukrishnan et al., 2006; Pandey et al., 2000; Tang et al., 2005; Wilson & Weng, 2010).

Although research evaluating urbanization impacts on surface runoff using L-THIA has been conducted in many studies, these mainly focused on watershed-scale analysis. White et al. (2015) conducted a national-scale study in the U.S. that focused on regional blue and green water balances and use by four crops: corn, wheat, sorghum, and soybean. However, quantitative assessment of urbanization impacts on surface runoff at a national scale is limited. The release of the National Land Cover Database (NLCD) 2011 edition datasets for 2001, 2006, and 2011 across the nation makes the assessment of urban land use change impacts on surface hydrology at the national level feasible (Homer et al., 2015). The NLCD extends land use/land cover coverage over larger areas at more frequent time intervals allowing large scale investigations to be conducted (DeFries & Eshleman, 2004). Explicit understanding of urbanization impacts on surface runoff will provide necessary input information to urban planners and decision makers who need to balance trade-offs between the advantages and possible unintended consequences of urbanization.

The Long-Term Hydrologic Impact Assessment-Low Impact Development 2.1 (The L-THIA-LID 2.1) model, developed by Liu et al. (2015b), has demonstrated its ability regarding evaluation of the performance of BMPs and LIDs at the watershed scale (Liu et al., 2015a, 2015b; Liu et al., 2016a, 2016b). A total of 12 BMPs or LIDs are represented in this model, including bioretention systems, rain barrels/cisterns, green roofs, open wooded spaces, porous pavements, permeable patios, detention basins, retention ponds, wetland basins, biofilter-grass swales, wetland channels, and biofilter-grass strips (Liu et al., 2015b). The model also represents the annual cost and cost effectiveness of implementing BMPs and LIDs (Liu et al., 2015b).

1.2 Research Objectives

Overall, this study aims to assess urbanization and climate impacts on surface runoff from national and regional scopes and then to evaluate the effects of green infrastructure on hydrology and water quality as mitigation approaches at local scales.

The specific research questions include: (1) how has urbanization impacted surface runoff on a national scale? (2) How does simulated precipitation from the updated CLIGEN database impact surface runoff in the Great Lakes Region? (3) How effective are green infrastructure practices on hydrology and water quality as sustainable development or re-development approaches at a CSO community?

Corresponding to these research questions, the specific objectives of this study are to:

(1) Assess urbanization impacts on surface runoff of the contiguous United States using the L-THIA Tabular Tool based on 2001, 2006, and 2011 National Land Cover Databases.

(2) Assess simulated precipitation from the updated CLIGEN database impacts on the surface runoff using the L-THIA Tabular Tool in the five states of the Great Lakes Region (WI, IL, IN, MI, and OH).

(3) Evaluate the effectiveness of green infrastructure practices as sustainable development or re-development tools on hydrology and water quality in a CSO community.

A snapshot of these three objectives and how they are related to each other is presented in Figure 1.1.

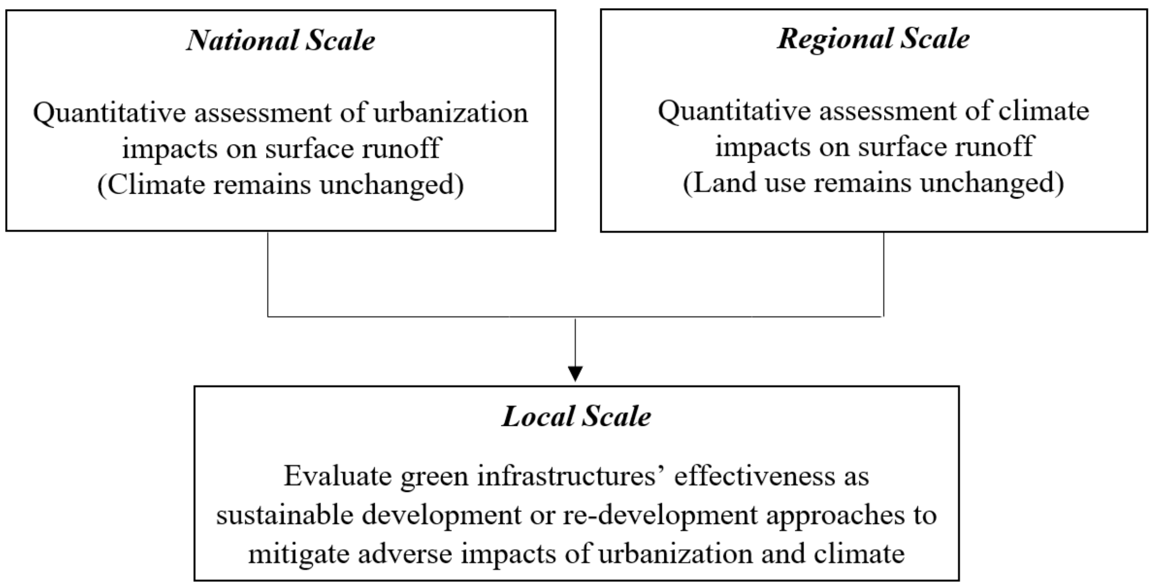

Figure 1.1 A flowchart of the research plan

1.3 Dissertation Organization

This dissertation has five chapters. Chapter 1 provides the problem statement and research objectives. Chapter 2 assessed urbanization impacts on surface runoff of the contiguous United States. Chapter 3 evaluated the suitability of CLIGEN precipitation estimates based on an updated database and their impacts on urban runoff at the Great Lakes Region. In Chapter 4, an evaluation of the effectiveness of green infrastructure practices on hydrology and water quality in a combined sewer overflow community is presented. The major research findings are summarized in Chapter 5. Recommendations for future research emanating from this work are also summarized in this chapter. The dissertation is presented in paper format; at the time of this submission, the work in Chapters 2 has already been published in a peer-reviewed journal, while the work in Chapter 3 has been accepted and is in press. The work in Chapter 4 is scheduled for submission.

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CHAPTER 2. URBANIZATION IMPACTS ON SURFACE RUNOFF OF THE CONTIGUOUS UNITED STATES[1]

Abstract

Urbanization has significant impacts on hydrologic processes and water quantity and quality. Research related to urbanization impacts on surface runoff has focused on changes up to the watershed scale. However, quantitative assessment at a national scale is scarce. This study applied a newly developed version of the Long-Term Hydrologic Impact Assessment (L-THIA) model, the L-THIA Tabular Tool, to assess urbanization impacts on average annual runoff of the contiguous U.S. based on the National Land Cover Database for 2001, 2006 and 2011. The results show that: 1) urbanization occurred non-uniformly across the nation from 2001 to 2011; 2) urban expansion and intensification were main driving forces altering surface runoff; 3) the majority of counties had low runoff with long-term (50 years) normalized average annual runoff depth (NAARD) from urban land less than 17.8 mm; 4) the states with the largest NAARD values had both high precipitation and increases in urban land, while the ten states with the largest NAARD change percentages were mainly in the western U.S. with low precipitation and the NAARD values were mainly influenced by large increases in urban land; 5) nationally, average annual runoff increased approximately 3.3 billion m^3 (10%) due to urbanization from 2001 to 2011. These results provide insight for decision makers who need to balance trade-offs between the advantages and possible disadvantages of urbanization.

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2.1 Introduction

Urbanization is one of the most important anthropogenic modifications of the global environment (Antrop, 2004; DeFries and Eshleman, 2004; Eshleman, 2004; Foley et al., 2005; Weng, 2002; Wu, 2014). Every urban region in the United States has expanded substantially in area in recent decades (USEPA, 2013). Urbanization presents humans with a dilemma (Foley et al., 2005). On one hand, urban development is essential because it provides convenience of infrastructure, goods and services needed by people, government, economic development, industry, and trade (Foley et al., 2005; Lowry, 1990); on the other hand, land surface modifications occur during the process of urbanization including vegetation reduction, soil compaction, and change from pervious surfaces to impervious surfaces such as roofs, roads, and parking lots (Arnold Jr and Gibbons, 1996; Booth and Jackson, 1997; Schueler, 1994). The consequences of these land surface modifications include but are not limited to: changes in water supply from altered hydrologic processes of infiltration, groundwater recharge, and runoff; water quality degradation from urban runoff and combined sewer overflows (CSO); and changes in water demand (Burns et al., 2012; DeFries and Eshleman, 2004; Gitau et al., 2016; Passerat et al., 2011; Semadeni-Davies et al., 2008; Tong and Chen, 2002; Vietz et al., 2016).

In general, surface runoff and river discharge increase when natural vegetation, especially forests, decrease (Costa et al., 2003; Foley et al., 2005; Sahin and Hall, 1996). Impervious surfaces developed during urbanization contribute more surface runoff due to decreased infiltration (Arnold Jr and Gibbons, 1996; Schueler, 1994; Schueler et al., 2009). Reduced infiltration leads to higher peak flows, even for short duration low intensity rainfall, and increases the risk of flooding (Bhaduri et al., 2001; Suriya and Mudgal, 2012). Urban runoff also carries non-point source pollutants, such as oil, grease, metals, and pesticides, into streams and rivers during rainfall events (Arnold Jr and Gibbons, 1996; Blair et al., 2014; Schueler, 1994). Even if urban runoff is captured by a sewer system and can be conveyed to wastewater treatment plants, Combined Sewer Overflows (CSO) in some highly urbanized areas continue to cause serious water pollution problems (Bhaduri et al., 2001; Passerat et al., 2011; Semadeni-Davies et al., 2008).

Computer-based hydrological models can save time and money because of their ability to perform temporal and spatial simulation of the effects of hydrologic processes and management activities on water quantity and water quality (Moriasi et al., 2007). The United States Department of Agriculture (USDA) Soil Conservation Services (SCS) Curve Number (CN) (NRCS, 1986) approach is widely used in several hydrological models that are used to assess the impact of land use change on surface hydrology, including sophisticated models such as the Soil and Water Assessment Tool (SWAT) (Arnold et al., 1998; Srinivasan et al., 1998), as well as models adopting the philosophy of simplicity such as the Long-Term Hydrologic Impact Assessment (L-THIA) (Bhaduri et al., 2000; Harbor, 1994). Sophisticated models usually require more parameters to set up the model before simulation, which may create limitations due to lack of data or time (Bhaduri et al., 2001). L-THIA is an easy, quick, and user-friendly tool, which only requires land use, hydrologic soil group (HSG), and long-term precipitation data to assess surface runoff influenced by past, present, as well as future land management practices for specific locations (Bhaduri et al., 2001; Grove et al., 2001; Tang et al., 2005). The L-THIA model has been successfully used in numerous studies to assess the impact of land use change on surface hydrology and water quality (e.g., Bhaduri et al., 2000; Choi, 2007; Davis et al., 2010; Grove et al., 2001; Gunn et al., 2012; Kim et al., 2002; Lim et al., 2006a, 2006b, 2010; Muthukrishnan et al., 2006; Pandey et al., 2000; Tang et al., 2005; Wilson and Weng, 2010). The L-THIA model has also been incorporated and integrated into Web-based and GIS-based decision support systems (Choi et al., 2003, 2005a, 2005b; Choi and Engel, 2003; Engel, 2001; Engel et al., 2003; Engel and Hunter, 2009; Pandey et al., 2000; Shi et al., 2004; Tang et al., 2004), as well as a low impact development model (Ahiablame et al., 2012a, 2012b, 2013; Engel and Ahiablame, 2011; Hunter et al., 2010; Liu et al., 2015a, 2015b, 2016a, 2016b; Martin et al., 2015; Wright et al., 2016).

Research evaluating urbanization impacts on surface runoff using L-THIA has focused primarily on watershed-scale analysis, and quantitative assessment of urbanization impacts on surface runoff at a national scale is limited. The release of the National Land Cover Database (NLCD) 2011 edition datasets for 2001, 2006, and 2011 across the nation makes the assessment of urban land use change impacts on surface hydrology at the national level feasible (Homer et al., 2015). The NLCD extends land use/land cover coverage over larger areas at more frequent time intervals allowing large scale investigations to be conducted (DeFries and Eshleman, 2004). Explicit understanding of urbanization impacts on surface runoff will provide information for decision makers who need to balance trade-offs between the advantages and possible unintended consequences of urbanization. This study assessed urbanization impacts on surface runoff for 2001, 2006, and 2011 in the contiguous United States using the L-THIA Tabular Tool, a derivative of the L-THIA model. We first present the assessment of the normalized average annual runoff depths (NAARD) for 2001, 2006, and 2011 by contiguous U.S. counties and evaluate whether population change is a consistent indicator for urban development; then we present an assessment of NAARD for 2001, 2006 and 2011 by U.S. states; finally, we present the assessment of the national average annual runoff volume increase due to urbanization.

2.2 Methods and Materials

2.2.1 L-THIA tabular tool

The L-THIA Tabular Tool, which shares the same philosophy as the original L-THIA model (Harbor, 1994), uses the SCS CN method (NRCS, 1986) to calculate average annual runoff for land use and soil combinations based on long-term climate data for that area (Bhaduri et al., 2001). The runoff depth is estimated when rainfall exceeds initial abstraction based on (NRCS, 1986):

$$Q = \frac{(P - I_a)^2}{(P - I_a) + S} \text{ for } P > I_a = 0.2S$$
$$Q = 0 \text{ for } P \le I_a = 0.2S \tag{1}$$

$$S = \frac{25400}{CN} - 254 \tag{2}$$

$$I_a = 0.2S \tag{3}$$

where Q represents direct runoff depth in mm; P represents rainfall depth in mm; S represents potential maximum retention after runoff begins in mm; CN represents curve number; and I_a represents initial abstraction in mm.

The L-THIA Tabular Tool was developed using the Python programming language and is run as a toolbox in ArcGIS version 10.2.2. It is designed to expedite tabulating calculations over diverse geographical areas. An external component (a Java program) calculates daily runoff for all curve numbers using the equations above. The calculations are performed for each precipitation gauge in a region using daily rainfall and all curve numbers. The calculations are made for each year in the database (typically between 30 to 100 years). The user selects how many of the available years of annual runoff to average in order to create average annual runoff, and the same number of years is used from every gauge.

The ArcGIS implementation (a Python script tool) of L-THIA Tabular consists of three main components. The first component aims to generate a CN raster using the land use and HSG combination. A six-digit combination raster layer including HSG (first digit), land use (second to third digits), and corresponding CN (fourth to sixth digits) was created for each pixel in the NLCD data, as shown in Table 2.1. The second component prepares the study area boundary for the runoff calculation. This operation determines which Thiessen polygons (Thiessen, 1911) representing rain gauges intersect the study area. The tool determines how many pixels of each CN raster is within each Thiessen polygon. The third component is used to tabulate the selected years of average annual runoff, under antecedent moisture condition (AMC) II condition (normal antecedent soil moisture), using the CN raster information generated from the first component inside the study outline generated from the second component, as well as rainfall data. The tabulation consists of accumulating the area of each CN present in each precipitation polygon in the region, and fetching the appropriate runoff total for each CN from the web-based results of the runoff calculator, and averaging the requested number of years to obtain average annual runoff for each CN used in the study area. Runoff depth is converted to volume through multiplying by pixel area for each CN. The final outputs of the L-THIA Tabular Tool include average annual runoff volume and depth within a given boundary. The units of the output average annual runoff volume and depth can be represented using both U.S. and metric units. The third component also computes non-point source pollution estimates using the runoff volume and the CN-land use pairs to choose the appropriate pollution coefficient (event mean concentration, EMC) if desired.

Table 2.1 Six-digit combination raster layer for urban land in the national land cover database (NLCD)

Land use index in NLCD	Urban land types in NLCD	Six-digit combinations			
21	Developed Open Space	121039	221061	321074	421080
22	Developed Low Intensity	122056	222071	322081	422086
23	Developed Medium Intensity	123069	223081	323087	423090
24	Developed High Intensity	124089	224092	324094	424095

Note: For six-digit combination raster layer, the first digit represents hydrologic soil group, A=1, B=2, C=3, and D=4; the second to third digits represent the urban land index in NLCD, 21 = Developed open space, 22 = Developed low intensity, 23 = Developed Medium Intensity, and 24 = Developed high intensity; the last three digits represent SCS CN value, those values are based on USDA NRCS (NRCS, 1986).

The study area is the contiguous United States, which includes 48 adjoining states and the District of Columbia (DC). The L-THIA Tabular Tool was set up to create analyses based on U.S. county boundaries. A total of 3,109 counties were modeled. Calibration and validation would be challenging for a national analysis due to the scarcity of observed surface runoff data from county based urban areas. However, the SCS CN method is a well-known and widely used runoff estimation approach, and a substantial number of studies have reported its usefulness and credibility; for example, many L-THIA studies have validated the SCS CN method with little or no calibration (Ahiablame et al., 2013; Bhaduri et al., 2001; Grove et al., 2001; Kim et al., 2002; Liu et al., 2015b; You et al., 2012). Thus, this approach can be applied to assess surface runoff at a national scale.

2.2.2 Input data

In the L-THIA Tabular Tool, the basic input data include daily precipitation, land use/ land cover data, and HSG data. In this study, CLIGEN (Climate Generator, version 5.3) - a stochastic weather generator employing quality control techniques for improved reproducibility of climatic parameters - was used to produce daily precipitation based on monthly statistical values derived from historic measurements at each particular site (Meyer et al., 2002, 2008; Zhang and Garbrecht, 2003). Quality control techniques in CLIGEN involve computing the probability that the mean and standard deviation of the random numbers driving CLIGEN were standard normal as assumed and reject those that were not. For the contiguous United States, 50-year daily precipitation was generated for a total of 2,600 CLIGEN stations nationwide using station specific parameter files. The Thiessen method (Thiessen, 1911) was applied to generate Thiessen polygons covering the contiguous United States based on CLIGEN station points. Areas within the same polygon shared the same daily precipitation dataset.

With the recent release of the NLCD 2011 product, a decade of consistently produced land cover datasets became available (Homer et al., 2015). Homer et al. (2015) suggested that NLCD 2001 (2011 Edition) and NLCD 2006 (2011 Edition) must be used if conducting direct change comparison. In this study, NLCD 2001 (2011 version), NLCD 2006 (2011 version), and NLCD 2011 (Fry et al., 2011; Homer et al., 2007, 2015), which were created by the Multi-Resolution Land Characteristics (MRLC) Consortium (Wickham et al., 2014), were used as sources of land use data. The three NLCD datasets adopted the same 16 class land cover classification scheme based on a decision-tree classification of Landsat satellite data circa that year at a spatial resolution of 30 m (Fry et al., 2011; Homer et al., 2007, 2015).

The State Soil Geographic (STATSGO) dataset (Wolock, 1997) was applied as a source of HSG. In this study, any complex HSG such as A/D, B/D, and C/D, was assumed to be D soil for urban areas due to construction impacts (Lim et al., 2006a); other single type HSG were assumed to be the same as the original HSG.

2.2.3 Preliminary assessment of land use/land cover change impacts on surface runoff

The initial motivation of this study was to estimate the land use/land cover change impacts on surface runoff nationally including all 16 land use types from the NLCDs. Precipitation and HSG data were applied uniformly to the three NLCDs within a ten-year time period. The L-THIA Tabular Tool was set up based on U.S. county boundaries, and the 50-year average annual runoff depth (AARD) was calculated for each pixel with land use and HSG combination within each specific U.S. county. The final output was AARD and volume for each U.S. county. The percentage change (from 2001 to 2006, from 2006 to 2011, and from 2001 to 2011) in AARD was calculated for each U.S. county. Counties classified as mild outliers (between 1.5 interquartile range (IQR) and 3.0 IQR) and extreme outliers (greater than 3.0 IQR) for both increasing and decreasing percent change AARD were mapped.

As shown in fig. 2.1, outliers with respect to percentage change in AARD showed no pattern, and some were reversed between the two five-year periods. For example, numerous counties in the western U.S. showed an increasing trend in AARD from 2001 to 2006, while from 2006 to 2011 they had decreasing AARD, which is counterintuitive. Land use/land cover changes were examined further for counties that had outliers of both increasing and decreasing AARD since the land use/land cover served as the only variable. As shown in fig. 2.2 a through d, some land use/land cover types such as wetlands, cultivated crops, forests, herbaceous, barren land, and open water, changed inconsistently between two five-year periods, which indicates likely misclassification issues. For example, in sub-figure a, classified barren land in the counties belonging to outliers with increasing AARD increased in area by 1915 km^2 from 2001 to 2006, but then decreased in area by 670 km^2 from 2006 to 2011; classified open water in the counties belonging to outliers with increasing AARD decreased in area by 2384 km^2 from 2001 to 2006, while it increased in area by 1348 km^2 from 2006 to 2011. Homer et al. (2015) stated that even though extensive efforts had been put into image pre-processing, spectral change detection, and change labeling work during NLCD creation, some misclassifications occurred. Our results of this preliminary modeling trial also indicate that apparent misclassification issues in some land use/land cover types in NLCDs may limit the analysis at a national scale. However, according to Homer et al. (2015), during the classification processes, urban class pixels in the NLCDs had top priority, with any change related to newly developed lands always being included in the final land cover change map. Since urban land classification has more accuracy than other categories, only urban land was considered in further analysis to assess the urbanization impact on surface runoff at a national scale for 2001, 2006, and 2011.

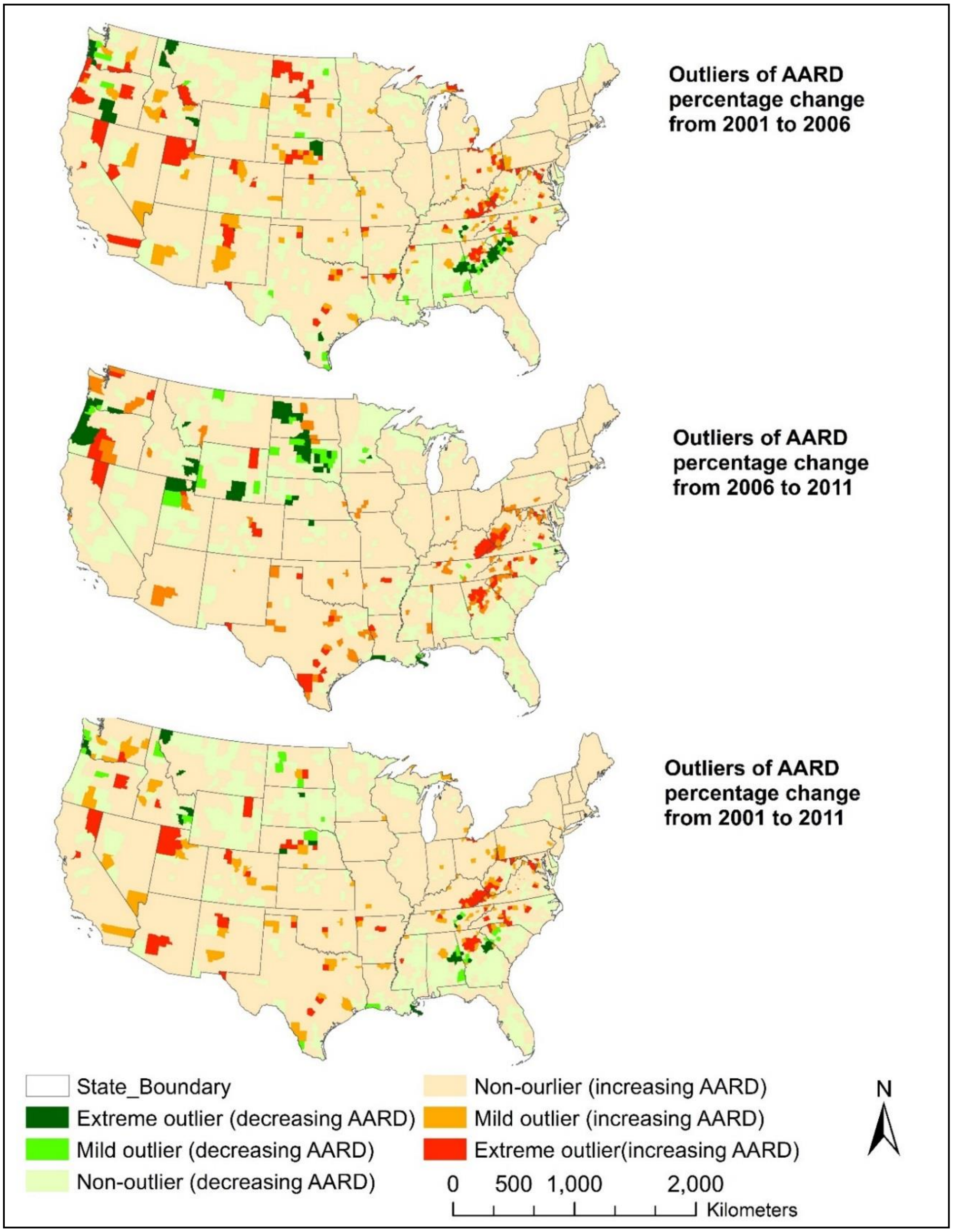
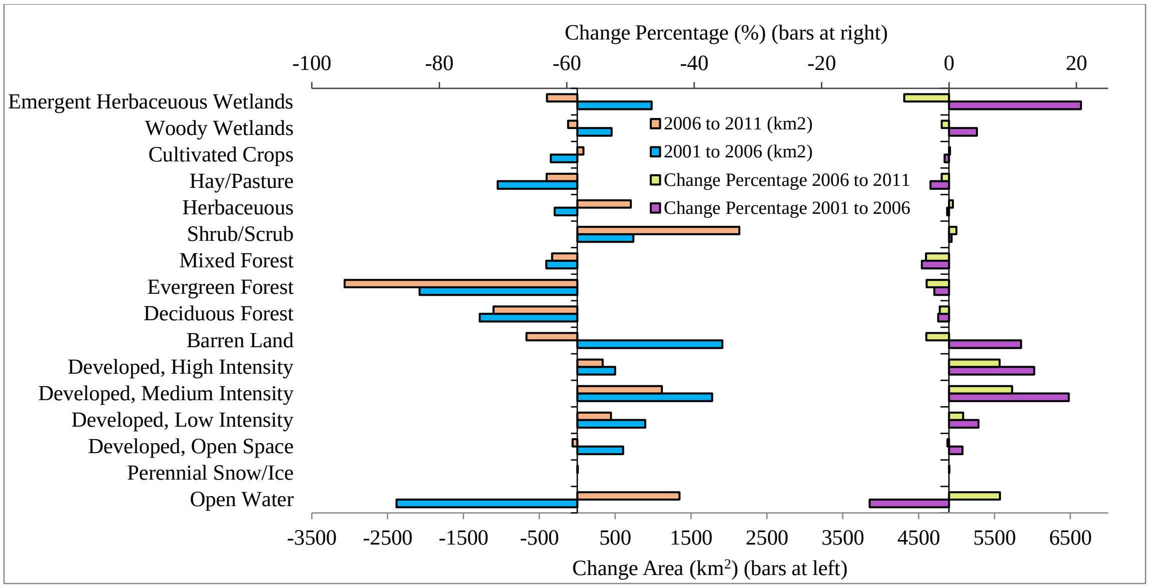
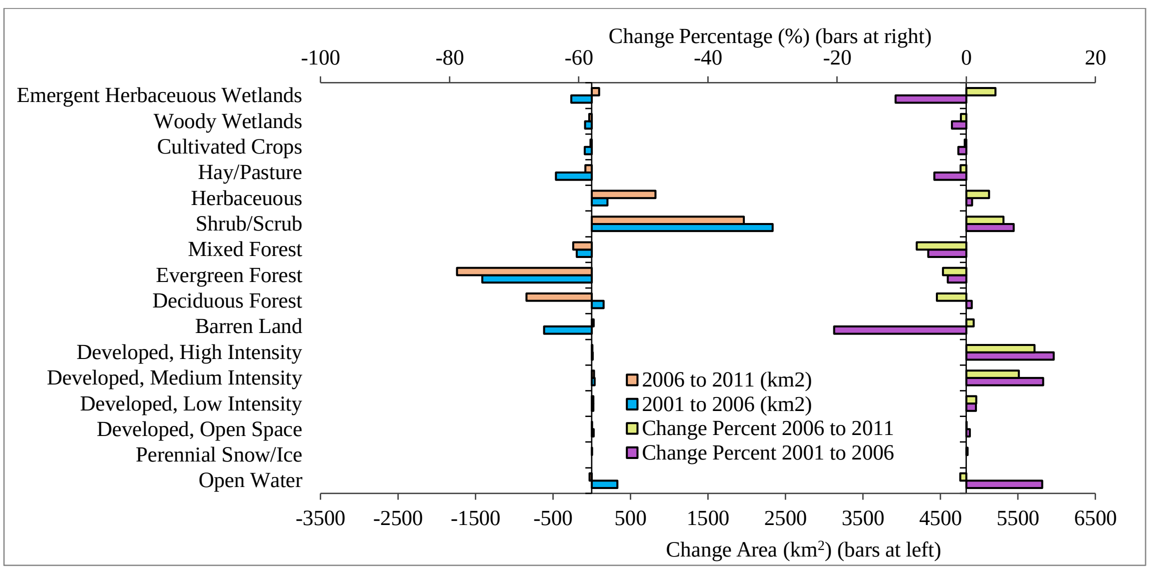


Figure 2.1 County based outlier of AARD change percentage



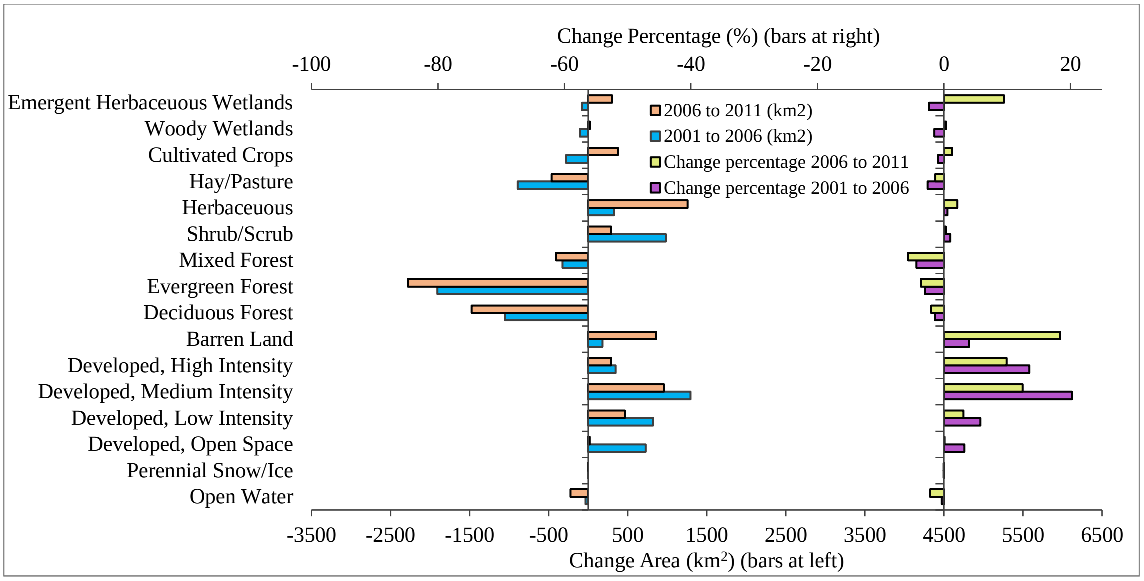
a. Land use/land cover change of counties belonging to outliers with increasing AARD from 2001 to 2006



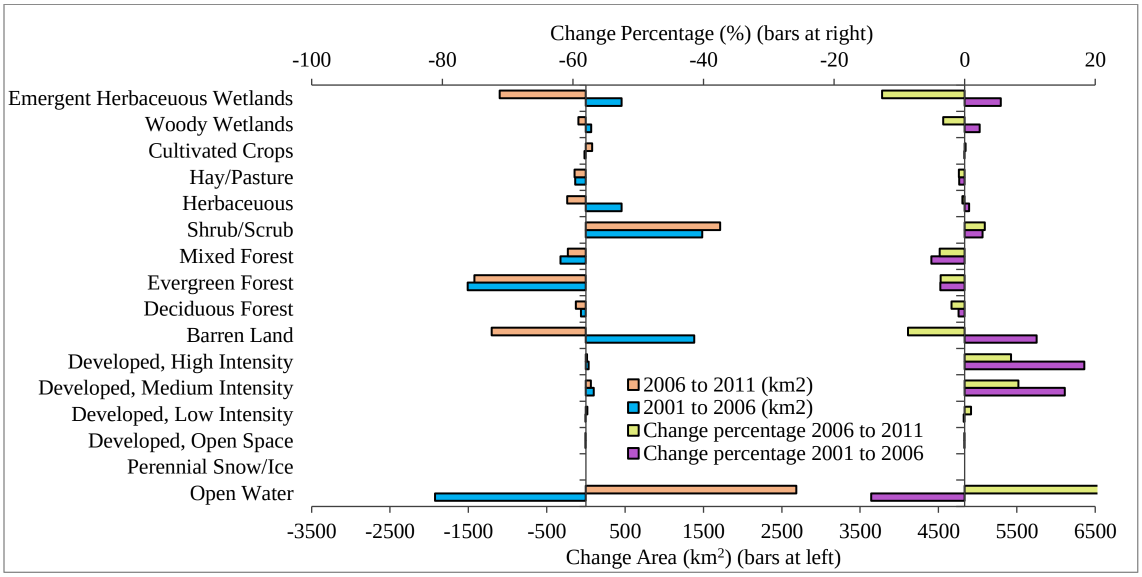
b. Land use/land cover change of counties belonging to outliers with decreasing AARD from 2001 to 2006

Figure 2.2 Land use/land cover change of outlier counties

Figure 2.2 continued



c. Land use/land cover change of counties belonging to outliers with increasing AARD from 2006 to 2011



d. Land use/land cover change of counties belonging to outliers with decreasing AARD from 2006 to 2011

The urban classes in NLCDs include developed open space, developed low intensity, developed medium intensity, and developed high intensity lands (Fry et al., 2011; Homer et al., 2007, 2015). Fig. 2.3 illustrates changes in urban land between 2001, 2006 and 2011. Within the ten-year time period from 2001 to 2011, urban development occurred and increased in area by 20,296 km^2. Developed medium intensity ranked at the top of net gain in area (9,049 km^2), developed low intensity ranked second with an increased area of 4,437 km^2, and developed high intensity and open space ranked third and fourth with similar net area gains (3,427 km^2 and 3,383 km^2, respectively). In the first five-year time period from 2001 to 2006, the ranks of net gain in area were: developed medium intensity (5,441 km^2) > developed low intensity (2,689 km^2) > developed open space (2,563 km^2) > developed high intensity (1,975 km^2). In the latter five-year period from 2006 to 2011, the ranks of net gain in area were: developed medium intensity (3,609 km^2) > developed low intensity (1,748 km^2) > developed high intensity (1,453 km^2) > developed open space (821 km^2). Comparing urban land change between the two five-year time periods, all four urban land types have higher net gain in area during the first five-year time period from 2001 to 2006 than in the latter five-year period from 2006 to 2011. One possible reason for this trend might be slower economic development in the latter five years due to the financial crisis beginning in 2008 (Erkens et al., 2012; Ivashina and Scharfstein, 2010; Peters et al., 2012; Reinhart and Rogoff, 2008).

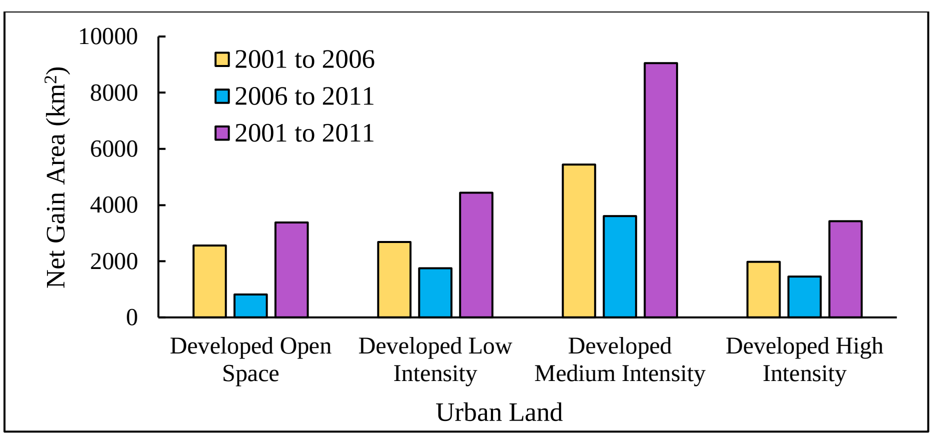


Figure 2.3 Urban land change between 2001, 2006, and 2011 for the contiguous U.S. (Data source: Homer et al., 2015)

2.2.4 Simulation of urban land impact on surface runoff

The L-THIA Tabular Tool was set up based on U.S. county boundaries, and the AARD was calculated only from pixels belonging to developed land as indicated by categories 21, 22, 23, and 24. The final output was AARD and volume from urban land for each U.S. county. In order to conduct direct comparison between different U.S. counties, normalized average annual runoff depth (NAARD) was calculated using the following equation:

$$NAARD_{County} = \frac{RV_{Urban}}{A_{County}} \times 1000 \tag{4}$$

where $NAARD_{county}$ represents NAARD for each U.S. county in mm, RV_{urban} represents the average annual runoff volume from urban land within each U.S. county in m^3, A_{county} represents the U.S. county area in m^2, and multiplying by 1000 represents the unit conversion from meters to millimeters.

The NAARD of each U.S. state was also calculated using the following equation:

$$NAARD_{State} = \frac{\sum_{i=1}^{n} RV_{County\_i}}{\sum_{i=1}^{n} A_{County\_i}} \times 1000 \tag{5}$$

where $NAARD_{State}$ represents the NAARD of each U.S. State in mm, n represents the total number of counties within a U.S. state, RV_{County_i} represents the average annual runoff volume from the i^{th} U.S. county within a U.S. state in m^3, A_{County_i} represents the area of the i^{th} U.S. county in m^2, and multiplying by 1000 represents the unit conversion from meters to millimeters.

2.2.5 Comparison of population change and urban land change

After obtaining NAARD values for U.S. counties and states, population data for 2001, 2006, and 2011 were obtained from U.S. Census Bureau (http://www.census.gov/) (U.S. Census Bureau, 2006, 2011) and considered in the analysis (Note: the population data for year 2001 are included in the U.S. Census Bureau 2006 dataset). Population change was compared with urban land change in order to evaluate whether population change was a consistent indicator for urban development.

2.3 Results and Discussion

2.3.1 Assessment of NAARD by counties in the contiguous U.S.

During the processes of urbanization, many other types of land, such as forest, shrub land, and herbaceous land, are converted to urban land; while some urban land can also be changed to other types of land, those changes are far less than urbanization as evidenced by the expansion in urban areas. Increased urban land over time can significantly increase runoff volume due to increased impervious surfaces. The spatial variability of urban development impacts spatial variation of runoff (Tang et al., 2005). The NAARD was used to assess urbanization impacts on surface runoff, which allows direct comparison of runoff among different regions regardless of area differences. The NAARD calculated for each county in the contiguous U.S. was represented using the standard deviation classification method that shows how spread out the values are and their spatial distribution. As shown in fig. 2.4, green represents low runoff, which means that NAARD values are less than 0.5 standard deviation of the NAARD value in 2001 (0 ~ 17.8 mm); yellow represents medium runoff, which means that NAARD values range between 0.5 to 1.5 standard deviation of

NAARD value in 2001 (17.8 mm ~ 37.4 mm); orange represents high runoff, which means that NAARD values range between 1.5 and 2.5 standard deviation of NAARD value in 2001 (37.4 mm ~ 57.0 mm); red represents very high runoff, which means that NAARD values are greater than 2.5 standard deviation of NAARD value in 2001 (57.0 mm ~ 331 mm).

Fig. 2.4 reveals an uneven spatial pattern of urban growth, which confirms that urbanization occurred non-uniformly across the contiguous U.S. from 2001 to 2011; the majority of counties in the contiguous U.S. belong to the category of low runoff with a NAARD value of less than 17.8 mm. The brighter colors including yellow, orange and red, representing medium, high, and very high runoff, occur mainly in large metropolitan areas, where a high proportion of impervious surfaces occur, such as Miami (FL), Orlando (FL), Tampa (FL), Jacksonville (FL), and Atlanta (GA) (see three insets in fig. 2.4), as well as Chicago (IL), Houston (TX), Dallas (TX), San Antonio (TX), Seattle (WA), New York (NY), Memphis (TN), Charlotte (NC), and Boston (MA). From 2001 to 2011, the color change patterns included: green was turned to yellow, yellow was turned to orange or red, orange was turned to red, and more yellow or orange emerged around the red. Those color change patterns over time indicate the occurrence of urban expansion and intensification.

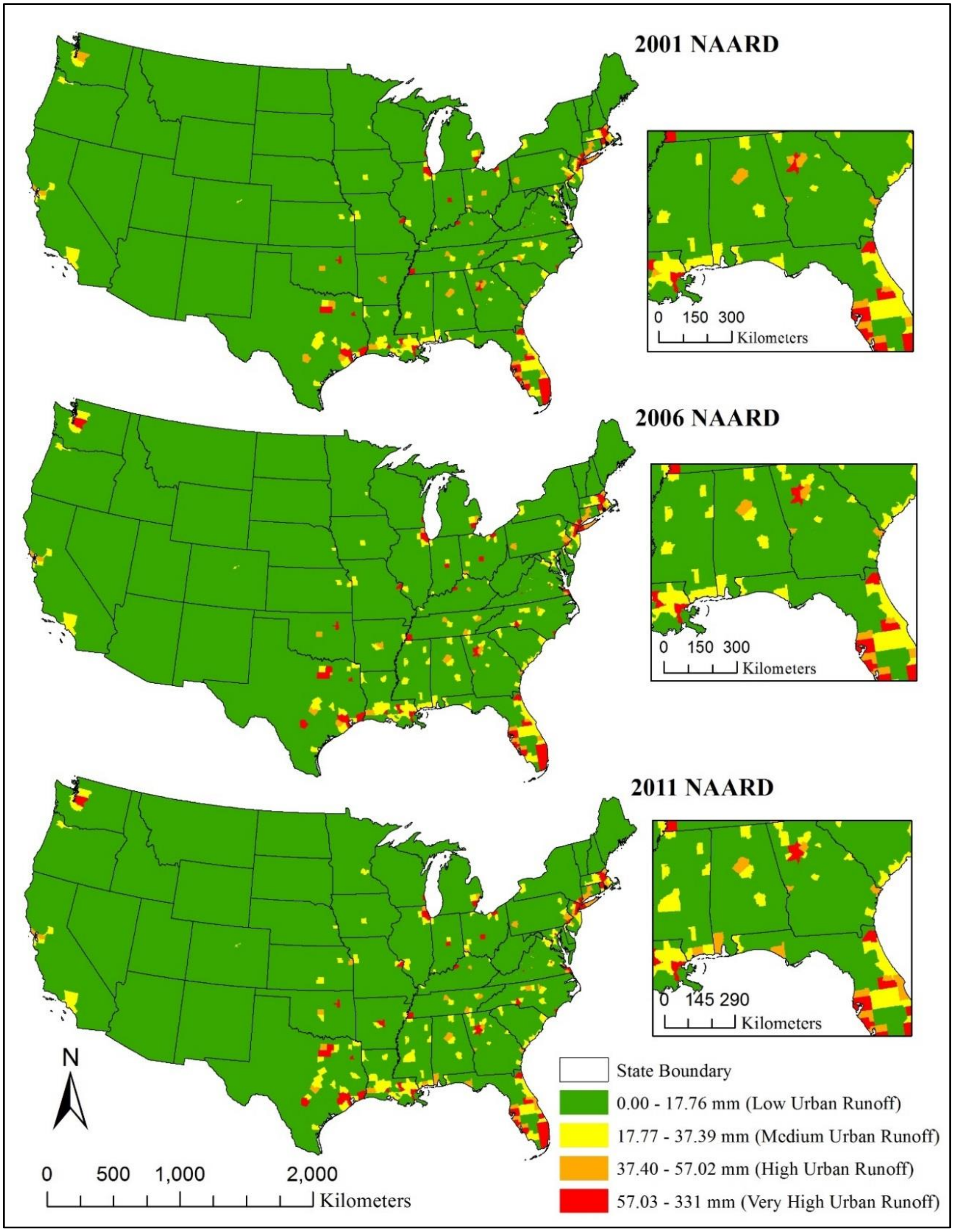


Figure 2.4 County-based long-term (50 years) normalized average annual urban runoff depth maps of contiguous U.S. in 2001, 2006, and 2011 based on L-THIA simulations

Table 2.2 lists the number of counties belonging to four categories of runoff. Low runoff counties are the dominant category among 2001 (91.9 %), 2006 (91.2%), and 2011 (90.5%). Total number of counties with low runoff decreases from 2001 to 2011, total number of counties with high runoff remains stable during the ten-year time period, while total number of counties with medium runoff and very high runoff increases from 2001 to 2011 with an increasing percentage of 21.8% and 23.3%, respectively. The increasing percentage of medium and very high runoff counties from 2001 to 2006 (12.1% and 16.4%, respectively) was greater than that from 2006 to 2011 (8.6% and 5.9%, respectively), which corresponds to the increasing trend in urban land during the two five-year time periods. Urban sprawl and urban intensification resulted in higher NAARD and more medium and very high runoff counties.

Table 2.2 Number of counties in the contiguous U.S. in low (green), medium (yellow), high (orange), and very high (red) urban runoff categories for 2001, 2006, and 2011

Categories	NAARD Range (mm)	2001		2006		2011		Change Percentage (%)		
		Amount	Percent (%)	Amount	Percent (%)	Amount	Percent (%)	01 to 06	06 to 11	01 to 11
Green	0 ~ 17.76	2858	91.93	2835	91.19	2815	90.54	-0.80	-0.71	-1.5
Yellow	17.77 ~ 37.39	124	3.99	139	4.47	151	4.86	12.10	8.63	21.77
Orange	37.40 ~ 57.02	54	1.74	50	1.61	53	1.70	-7.41	6	-1.85
Red	57.03 ~ 331.00	73	2.35	85	2.73	90	2.89	16.44	5.88	23.29

2.3.2 NAARD, urban land, and population of counties with very high runoff

For 90 counties with very high runoff in 2011, analysis of urban land change and population change were conducted. The top ten counties with increased percentage of NAARD from 2001 to 2011 and their associated urban land change as well as population change are shown in fig. 2.5. The percentage increase of NAARD from 2001 to 2011 for the top ten counties with very high runoff ranges from 18.0% to 44.8%, and their high percentage increase of NAARD are associated with increased population (11.3% to 60.9%) as well as a large increase of urban land (10.9% to 34.6%). Typically, large urbanization rates are driven by large population growth rates rather than by economic growth (Buhaug and Urdal, 2013; Cincotta et al., 2003). For the top ten counties with very high runoff, large population increases likely stimulated urban development in order to fulfill people's life needs. Population could be considered a driving force for increasing the extent of developed land. However, as depicted in fig. 2.6, many exceptions existed; some counties with very high runoff experienced an increase in NAARD and developed land from 2001 to 2011, but the population decreased during the same ten-year time period. This result is consistent with UN statistics (UN, 2010; UN-Habitat, 2010), which show that the global urban population increased more than four times during the 20th century. Further, the statistics show that urban growth remained persistent even while overall population growth is slowing. Thus, population growth is not a consistent factor stimulating urban development.

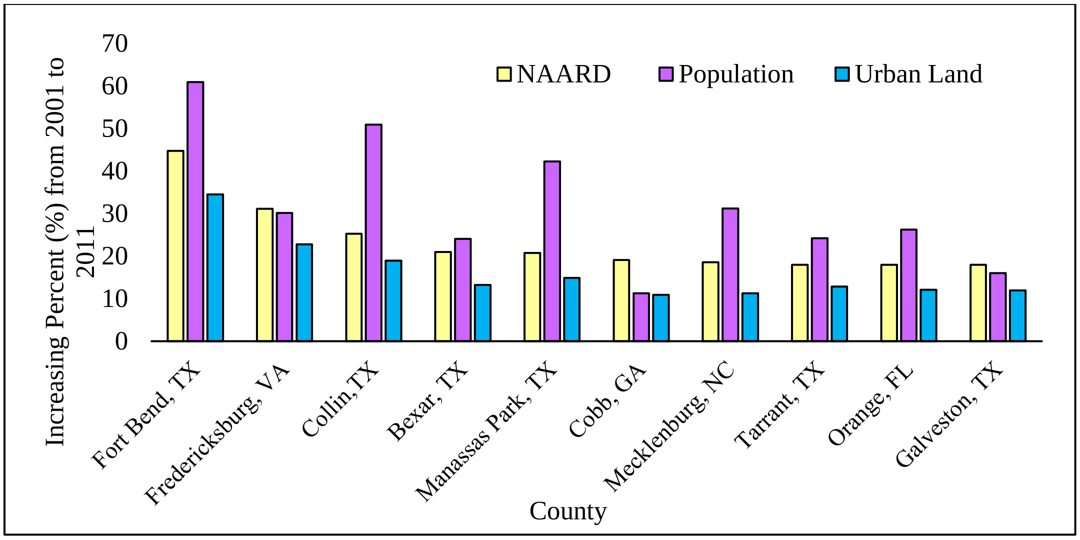


Figure 2.5 Change percentage of NAARD, population, and urban land from 2001 to 2011 of top ten counties with very high runoff based on L-THIA simulations

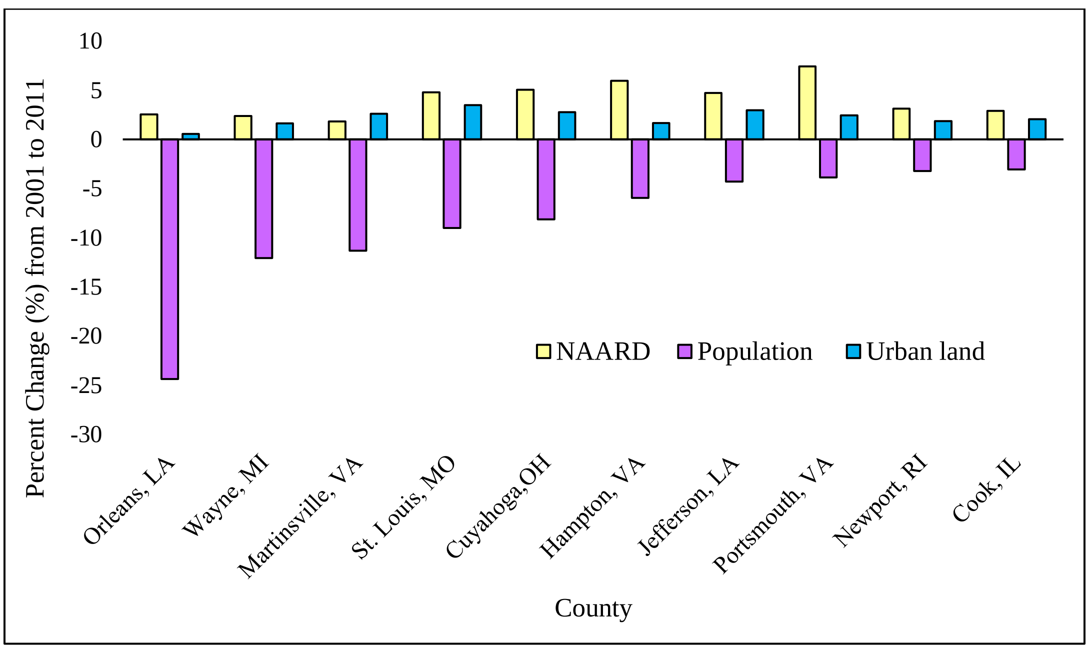


Figure 2.6 Change percentage of NAARD, population, and urban land from 2001 to 2011 of some counties with very high runoff based on L-THIA simulations

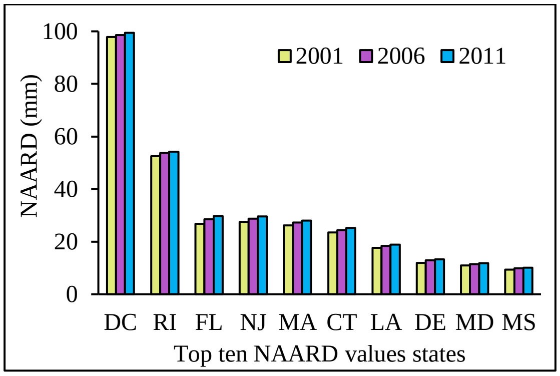
2.3.3 Assessment of NAARD by states in the contiguous U.S.

The NAARD value of each state in the contiguous U.S. was calculated in order to conduct direct comparisons of runoff among different states regardless of area. The top ten NAARD values states in 2011 (group 1) as well as the top ten NAARD change percentage states (group 2) in 2011 were selected for the state level analysis. For each group, the NAARD values, urban land change percentage among 2001, 2006, and 2011, as well as population change among 2001, 2006, and 2011, are depicted in fig. 2.7. Sub-figures a through c represent group 1 and sub-figures d through f represent group 2. There are similarities between the two groups: NAARD values increased from 2001 to 2011; urban land had increasing percentages in both five-year periods, and the percentage increase from 2001 to 2006 was higher than that from 2006 to 2011, except in DC and Wyoming (WY).

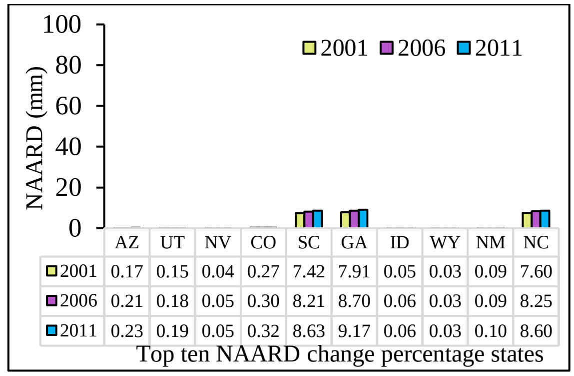
The NAARD values of the top ten NAARD value states, as shown in sub-figure a of fig. 2.7, range from 10.2 mm to 99.4 mm in 2011. Those ten states are mainly located in the northeast, east, southeast, and southern United States. One reason for their high NAARD values is that higher precipitation often occurs in those areas relative to other US locations. A large percentage of urban land also contributes to high NAARD values as depicted in sub-figure b of fig. 2.7. Population change within the decade from 2001 to 2011 varied in different states, as shown in sub-figure c of fig.7. Rhode Island (RI) experienced a population increase from 2001 to 2006, but saw a population decrease from 2006 to 2011 which was of a higher magnitude than the prior increase, resulting in an overall population decrease from 2001 to 2011. Louisiana (LA) experienced a population decrease from 2001 to 2006 attributable to natural disasters such as Hurricane Katrina in 2005 (Burby, 2006; Hartman and Squires, 2006; Kates et al., 2006), while population increased from 2006 to 2011 with higher magnitude, which led to an overall increased population from 2001 to

2011. Seven other states and DC have undergone population increases in both five-year time periods, among them, population percentage increases from 2001 to 2006 in four states including FL, New Jersey (NJ), Delaware (DE), and Maryland (MD) were higher than that from 2006 to 2011; the population increase of Massachusetts (MA), Connecticut (CT), Mississippi (MS) and DC from 2001 to 2006 was lower than that from 2006 to 2011.

For group 2, as shown in sub-figure d of fig. 2.7, seven out of the top ten NAARD change percentage states are distributed in the western U.S. except South Carolina, Georgia, and North Carolina. The NAARD values of those western states are low due to low precipitation in those areas. Their high percentage increases in NAARD values were greatly influenced by large urban land increases within the decade from 2001 to 2011 (ranging from 3.1% to 12.3% in 2011), as shown in sub-figure e of fig. 2.7, which indicated that areas that experienced more urban growth had a larger potential for increased average annual surface runoff. Similar studies also found that rapid urban expansion increased annual runoff, daily peak flow, and flood volume (Barron et al., 2013; Du et al., 2012; Weng, 2001; White and Greer, 2006). Population increased consecutively in the two five-year time periods from 2001 to 2011 with different magnitudes, as depicted in sub-figure f of fig. 2.7. The increasing percentage ranges from 13.6% to 29.9% from 2001 to 2011, which could stimulate urban development. By comparing groups 1 and 2, urban growth had a higher magnitude in areas with larger population increases, while urban growth continues to occur even if population decreases. This indicates that population should only be considered as one of the possible factors stimulating the growth of urban land, and that only considering population growth is not a good way to analyze the increase in urban development, consistent with Hasse's (Hasse and Lathrop, 2003) study on land resource impact indicators of urban sprawl.

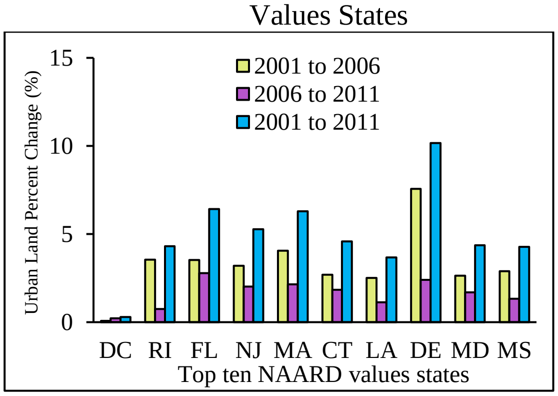


a. NAARD values of Top Ten NAARD Values States

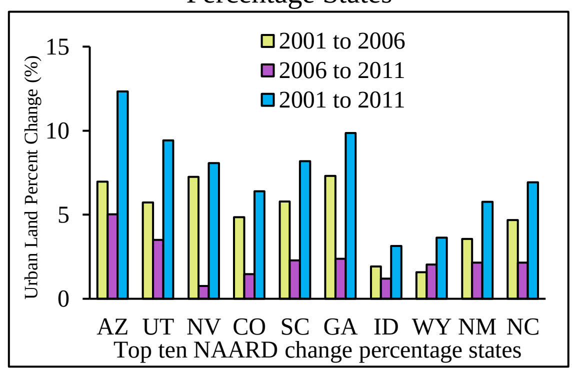


	AZ	UT	NV	CO	SC	GA	ID	WY	NM	NC
2001	0.17	0.15	0.04	0.27	7.42	7.91	0.05	0.03	0.09	7.60
2006	0.21	0.18	0.05	0.30	8.21	8.70	0.06	0.03	0.09	8.25
2011	0.23	0.19	0.05	0.32	8.63	9.17	0.06	0.03	0.10	8.60

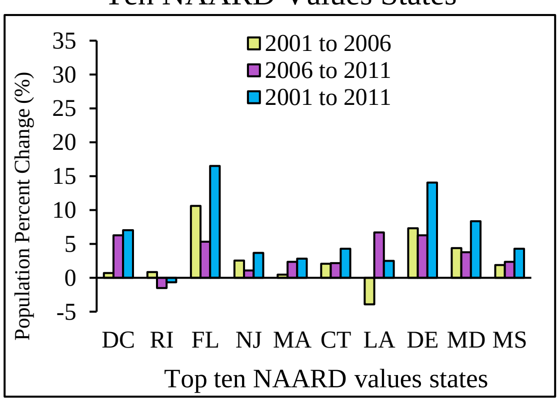
d. NAARD Values of Top Ten NAARD Change Percentage States



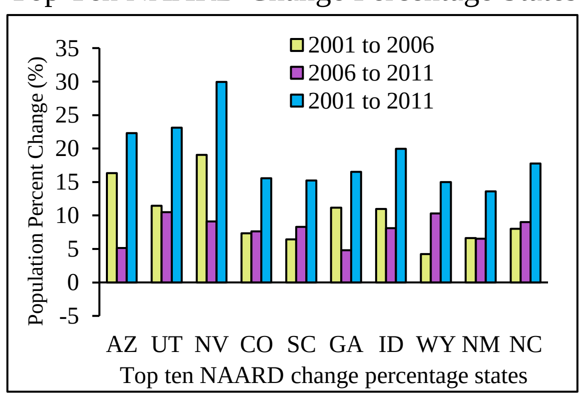
b. Urban Land Change Percentage of Top Ten NAARD Values States



e. Developed Land Change Percentage of Top Ten NAARD Change Percentage States



c. Population Change Percentage of Top Ten NAARD Values States



f. Population Change Percentage of Top Ten NAARD Change Percentage States

Figure 2.7 Comparison of top ten NAARD values states in 2011 (group 1, left side a, b and c) with top ten NAARD change percentage states in 2011 (group 2, right side d, e and f). Note: population data were from U.S. Census Bureau population (U.S. Census Bureau, 2006, 2011)

2.3.4 Assessment of runoff volume increase due to urbanization

The national average annual runoff volume was calculated by summing runoff volume from each county. As depicted in fig. 2.8, for the contiguous U.S., about 1.9 billion cubic meters of average annual runoff were generated due to urbanization from 2001 to 2006 and about 1.4 billion cubic meters of average annual runoff were gained from urbanization from 2006 to 2011, totaling 3.3 billion cubic meters of average annual runoff gained due to urbanization for the decade from 2001 to 2011, which is about 10% of the total amount of urban runoff in 2001. This increased runoff can have substantial impacts on many issues, such as flooding, reduced groundwater recharge, and water quality degradation.

The scale of the national average annual runoff volume increase due to urbanization from this study can be used to strengthen decision makers' awareness of potential long-term impacts of urban expansion and intensification. It also implies that future urban planning or urban re-development should consider mitigation approaches, such as low impact development to reduce the impacts of urbanization.

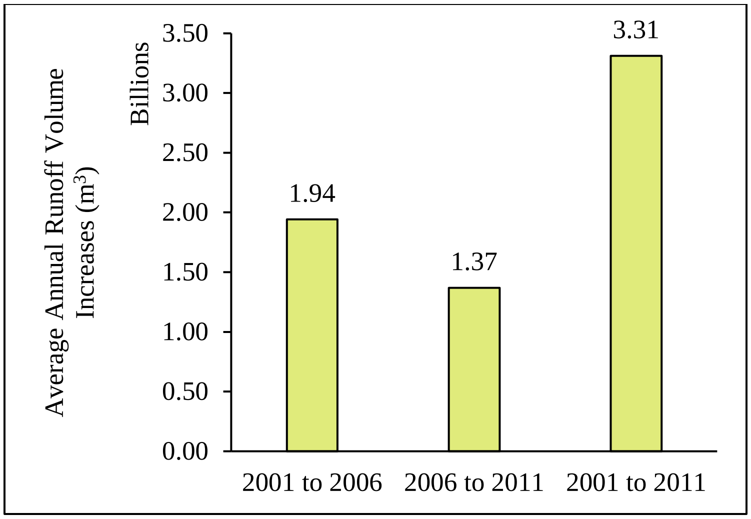


Figure 2.8 National average annual runoff volume increases due to urbanization based on L-THIA simulations

2.4 Conclusions

This study quantified urbanization impacts on surface runoff at a national scale. The contiguous United States underwent urbanization in the decade from 2001 to 2011. Urbanization occurred non-uniformly across the nation, and urban expansion and intensification served as driving forces altering surface runoff. The runoff change analysis revealed that: (i) the majority of counties in the contiguous Unites States were low runoff counties during the period 2001 to 2011 and had predicted long-term normalized average annual runoff depth from urban land less than 17.8 mm. However, spread of urban sprawl to suburban areas around metropolitan as well as newly urbanized areas within metropolitan areas resulted in more medium and very high runoff counties; (ii) For the top ten NAARD states in 2011, NAARD values were jointly influenced by high precipitation and increasing urban land, while the top ten NAARD change percentage states in 2011 were mainly in the western U.S. in areas with low precipitation, and their NAARD values were mainly influenced by high increases in urban land; (iii) Nationally, about 3.3 billion cubic meters of predicted average annual runoff were gained due to urbanization from 2001 to 2011; and (iv) Population increases are a factor in urban development, however population is not a good predictor of urbanization levels because some areas have undergone decreasing population but increasing urban land area. Therefore, population change alone is not a sufficient proxy with which to analyze the increase in urban development.

This study also demonstrated that the L-THIA Tabular Tool is capable of generating useful information about urbanization impacts on surface runoff using the stochastic weather generator, CLIGEN, together with the NLCD datasets and STATSGO soil dataset. Potential future research directions include exploring urbanization impacts on water quality, for instance, computing non-point source pollution estimates using the runoff volume and the CN-land use pairs to choose the appropriate EMC; comparing the results from this study to results obtained by applying nationwide spatial-distributed observed daily precipitation; comparing the results from this study to results by applying Soil Survey (SSURGO) geographic data; or simulating future land use change scenarios and their hydrological and environmental impacts, among others. The results of this study can be used to strengthen a decision maker's awareness of potential long-term impacts of urbanization. The tool can also be used for analyzing trade-offs between the advantages and possible unintended consequences of urbanization in future urban planning or urban re-development.

2.5 Acknowledgements

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CHAPTER 3. SUITABILITY OF CLIGEN PRECIPITATION ESTIMATES BASED ON AN UPDATED DATABASE AND THEIR IMPACTS ON URBAN RUNOFF: A CASE STUDY OF THE GREAT LAKES REGION[2]

Abstract

Weather generators rely on historical meteorological records to simulate time series of synthetic weather inputs, the quality of which has direct influence on model applications. The weather generator CLIGEN's database has recently been updated to comprise consistent historical records from 1974 to 2013 (updated CLIGEN database, UCD) compared to the current database in which records are of different lengths. In this study, CLIGEN's performance in estimating precipitation using UCD and the subsequent impact on urban runoff simulations were evaluated. Generally, UCD-based precipitation could replicate observed daily precipitation up to the 99.5th percentile but maximum precipitation was underestimated. Results from the Long-Term Hydrologic Impact Assessment model using UCD-based precipitation showed about 0.57 billion cubic meters more (14.9%) average annual runoff being simulated compared with simulations based on the current CLIGEN database. Overall, CLIGEN with the updated database was found suitable for providing precipitation estimates and for use with modeling urban runoff or urbanization effects.

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3.1 Introduction

Weather inputs are frequently required for many applications such as to: evaluate the impacts of watershed changes on hydrology, water quality, or erosion; design hydraulic structures; conduct land use planning; and/or assess alternative crop or range management strategies (Hanson et al. 2002, Richardson and Wright 1984). Weather generators are computer models with mathematical algorithms that use historical meteorological records to simulate time series of synthetic weather data that, ideally, have similar statistical properties as observed data for different spatial and temporal resolutions (Chen et al. 2010, Peleg et al., 2017). The past decade has seen a significant increase in interest in weather generators, and they are widely used in hydrological, ecological, and agricultural modeling (Chen et al. 2010, Ivanov et al. 2007, Kou et al. 2007, Liu et al. 2016, Wang et al. 2014, Wang et al. 2017, Wang and Kalin 2017). The quality of synthesized weather inputs has a direct influence on the responses of model applications. Compared to historical weather records that may be too short or may contain considerable amounts of missing data due to equipment servicing or malfunction, the output of weather generators can provide complete meteorological values for any desired period of time that can enhance application of continuous hydrological models; weather generators can also simulate weather data for ungauged areas through spatial interpolation of model parameters from adjacent gauged locations (Baffaut et al. 1996, Semenov and Barrow 1997, Zhang and Garbrecht 2003). Moreover, weather generators can be combined with dynamical models for developing climate change scenarios to assess the impact of future climate change by modifying weather generator parameters (Fatichi et al. 2011, Flanagan et al. 2014, Li et al. 2011, Mullan 2013, Pruski and Nearing 2002, Semenov and Barrow 1997, Wilks 1992, Zhang 2005).

Many weather generators have been developed and documented in the literature over the years to help reduce the time required to prepare meteorological input datasets, such as the Weather Generator (WGEN) (Richardson 1981, Richardson and Wright 1984), CLImate GENerator (CLIGEN) (Nicks et al. 1995), Long Ashton Research Station-Weather Generator (LARS-WG) (Semenov and Barrow 2002), Generation of Weather Elements for Multiple Applications (GEM) (Hanson et al. 2002), a weather generator for hydrological, ecological, and agricultural applications (Ivanov et al. 2007), Matlab-based daily scale weather generator (WeaGETs) (Chen et al. 2010, Chen et al. 2012), Advanced WEather GENerator for a two-dimensional grid (AWE-GEN-2d) (Peleg et al. 2017), and so forth. Among these, CLIGEN is widely used. It was first developed as the Water Erosion Prediction Project (WEPP) weather generator to simulate daily climatic inputs, including precipitation, temperature, dew point, wind, and solar radiation for a single geographic point, using monthly parameters (means, standard deviations, skewness, etc.) derived from the historical measurements (Nicks et al. 1995, Flanagan et al. 2001). It also produces individual storm parameter estimates, including time to peak, peak intensity, and storm duration (Nicks et al. 1995, Flanagan et al. 2001). The approach used in identifying stations for CLIGEN parameterization was to have one parameterized station for each grid of one degree latitude by one degree longitude (Nicks et al. 1995).

The performance and application of CLIGEN have been extensively studied worldwide, with work including: 1) performance evaluation of the generator at different locations (Al-Mukhtar et al. 2014, Chen et al. 2009, Elliot and Arnold 2001, Fan et al. 2013, Kou et al. 2007, Min et al. 2011); 2) simulation of individual rainfall events over long periods to calculate runoff and soil loss (Baffaut et al. 1996, Chen et al. 2017) and estimate rainfall erosivity (Hoomehr et al. 2016, Lobo et al. 2015); 3) integration with national agricultural pesticide risk analysis system (Lim and Engel 2003, Thomas et al. 2007); 4) prediction of seasonal wet and dry spell lengths (Arnold and Elliot 1996); and, 5) disaggregation of daily precipitation estimates into hourly data (Elliot et al. 1992). Studies have also been carried out to investigate the minimum simulation period required to obtain stable output predictions, such as soil loss with WEPP (Baffaut et al. 1996). The default CLIGEN database (Pruski and Nearing 2002) and user defined station parameter files (Hoomehr et al. 2016, Kim et al. 2009, Li et al. 2011, Pyke 2005, Yu 2003, 2005, Zhang and Garbrecht 2003, Zhang 2003, 2005, 2006, 2007, Zhang et al. 2010) have been used to generate precipitation time series as a baseline for climate change studies. Work has also been done to combine CLIGEN with adjusted parameters based on future conditions to generate daily series data for different climate change scenarios (Hoomehr et al. 2016, Li et al. 2011, Pruski and Nearing 2002, Pyke 2005, Yu 2005, Zhang 2005, 2006, 2007, Zhang et al. 2004, Zhang et al. 2010) and to adjust CLIGEN parameter values to reproduce the declining precipitation trend on a regional scale (Vaghefi and Yu 2016).

The currently available CLIGEN (version 5.3) database contains 2,527 weather stations covering the contiguous United States. Available historic measurements for each station varies from 10 to 117 years and includes data through 1992 (current CLIGEN database, CCD). Recently, the CLIGEN database was updated to include 2,702 weather stations across the nation with each station having the same time period of historic measurements from 1974 to 2013 (updated CLIGEN database, UCD). Historical records of 20, 30, 40, 50, and 60 years were statistically compared in order to determine a suitable length of historical measurements that would be used for CLIGEN parameterization. Based on these evaluations, 40 years were considered as a suitable period for UCD allowing it to provide up-to-date and temporally-consistent statistical parameterization for weather stations across the U.S.

The Long-Term Hydrologic Impact Assessment (L-THIA) model is a user-friendly tool to assess surface runoff based on the Curve Number method (Bhaduri et al. 2000, Harbor 1994, Natural Resources Conservation Services (NRCS) 1986). The L-THIA model has been successfully applied in numerous studies to assess land use change impacts on surface hydrology and water quality (Bhaduri et al. 2000, Davis et al. 2010, Gunn et al. 2012, Tang et al. 2005, Wilson and Weng 2010). The combination of CLIGEN and L-THIA provides a way to estimate the runoff from urban land based on daily precipitation estimates for long-term simulation (Chen et al. 2017). Chen et al. (2017) adopted CCD to simulate 50 years of daily precipitation for each of the 2,527 weather stations at a national scale and then used the simulated values in the L-THIA model (L-THIA Tabular Tool version) to evaluate urbanization impacts on urban surface runoff across the contiguous United States. They concluded that annual urban runoff volume increased 10% nationally from the National Land Cover Database (NLCD) 2001 data to NLCD 2011 data due to urbanization (Chen et al. 2017). The performance of CLIGEN utilizing the updated database, with respect to precipitation estimates and how these changes can impact urban surface runoff simulations compared to the Chen et al. (2017) results are of interest.

Therefore, the first objective of this study was to evaluate the performance of CLIGEN using the updated database in generating daily, monthly, and yearly precipitation amounts, and how well these matched the statistical parameters of the historical records. The second objective was to quantify the impacts of CLIGEN applications with the updated database on daily precipitation estimates for urban runoff simulations using the L-THIA Tabular Tool. In this paper, we first present the comparison of observed average annual precipitation between the current and updated CLIGEN databases across the contiguous United States. We then narrow our analysis to the five states of the U.S. Great Lakes Region (Wisconsin (WI), Illinois (IL), Indiana (IN), Michigan (MI), and Ohio (OH)) and present the statistical comparisons between the observed precipitation records and the CLIGEN precipitation estimates at eight weather stations in the region. Finally, we present the differences in L-THIA-predicted normalized average annual runoff depth (NAARD) from urban land based on NLCD 2001, 2006, and 2011 in the U.S. Great Lakes Region by using CLIGEN precipitation estimates created with the current and updated databases.

3.2 Methods and Materials

3.2.1 CLIGEN weather generator

CLIGEN can be used to generate continuous simulations of climate components including daily precipitation occurrence, amount, duration, time to peak, peak storm intensity, daily values of maximum, minimum, and dew point temperatures, solar radiation, and wind speed and direction (Nicks et al. 1995). Only the principles of precipitation simulation are presented here. Other principles of CLIGEN climate output generation can be found in the associated documentation (Flanagan et al. 2001, Nicks et al. 1995).

Parameters for a specific weather station needed to generate daily weather data include monthly mean, standard deviation, and the skewness of daily precipitation on a wet day (directly extracted from the historical daily precipitation records of a station), and monthly probabilities of precipitation occurrences of a wet day following a wet day P(W|W), and a wet day following a dry day P(W|D). The conditional probabilities P(W|W) and P(W|D) are generated by using a first-order two-state Markov chain (Zhang and Garbrecht 2003). A precipitation event is predicted to occur if a random number that is drawn from a uniform distribution for each day is less than the precipitation probability for the previous day (Zhang and Garbrecht 2003). A skewed normal distribution is used to generate daily precipitation amounts (Nicks et al. 1995). For UCD, historical precipitation records for 2,702 weather stations from 1974 to 2013 were used to derive parameter files for each station.

3.2.2 Preliminary comparison of two CLIGEN databases

A comparison of observed average annual precipitation (AAP) was conducted between UCD and CCD at the national scale. The AAP was calculated by adding up observed monthly average precipitation from CLIGEN output for every station of CCD (2527 stations) and UCD (2702 stations).

A Thiessen polygon (Thiessen 1911) map was generated based on CCD weather stations in order to show the AAP comparison results between the two databases. Figure 3.1 (upper map) shows the differences (percentage) in AAP between the current and updated databases with current as the baseline. White polygons represent areas with no data. There were 758 stations among the total of 2527 stations that did not match between the two CLIGEN databases. For those locations, UCD included new stations that had available historical measurements from 1974 to 2013. Other colored polygons (1769 stations) indicate stations that are matched or where the same stations are used in the two CLIGEN databases. Based on UCD, light yellow denotes the 1,236 stations for which AAP increased up to 15%, orange denotes the 35 stations for which AAP increased 15% to 30%, red denotes the 3 stations for which AAP increased 30% to 260%, light green represents the 479 stations for which AAP decreased 0.1% to 15%, and dark green represents the 16 stations for which AAP decreased 16% to 34%. These results indicated that differences existed in observed average annual precipitation. The historical records might not have been complete for each of the weather stations, which might have resulted in inaccuracies in computation. Differences could also have been due to different lengths of historical measurements.

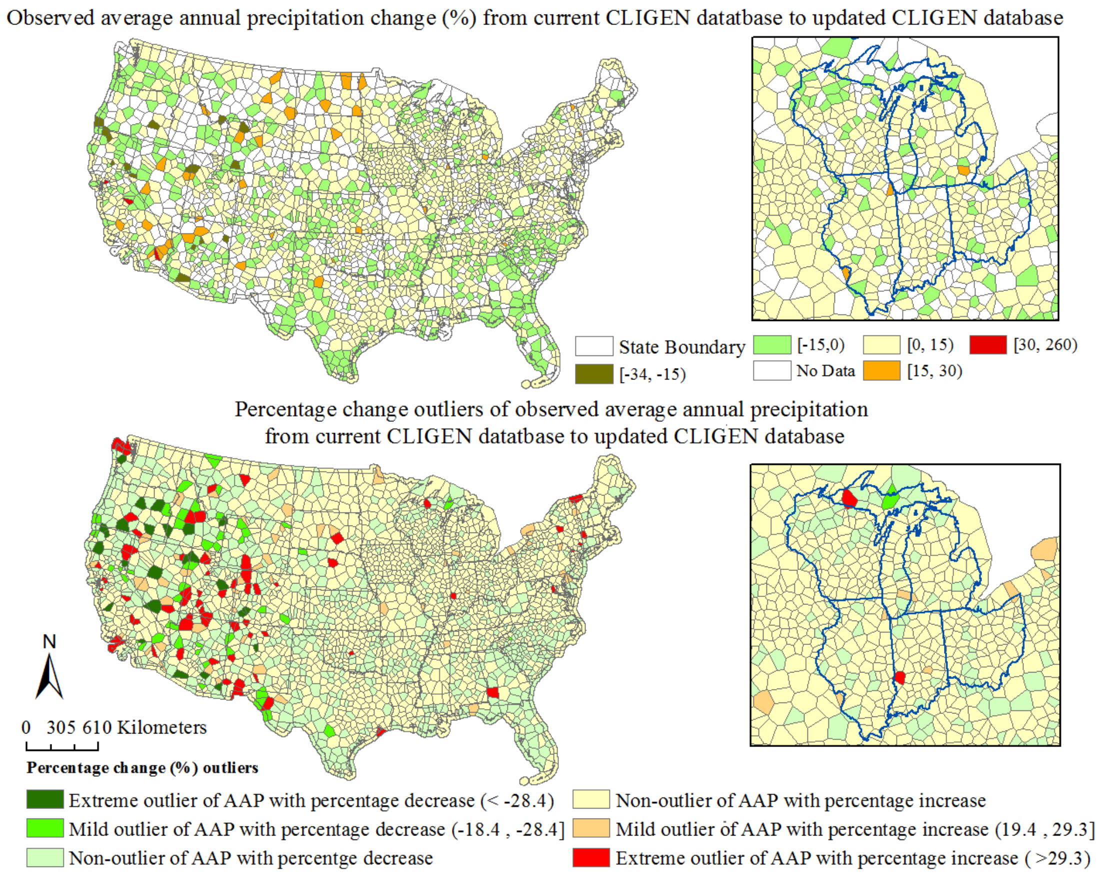


Figure 3.1 A comparison of observed average annual precipitation (AAP) between the current and updated CLIGEN databases

In order to have a complete picture regarding AAP change between the two CLIGEN databases at a national scale, those 758 unmatched stations in CCD were matched using the nearest station in UCD. After the matching process, outliers of the observed AAP percentage changes from CCD to UCD were determined (figure 3.1, lower map; table 3.1). The majority of stations (2,335) belonged to non-outliers based on AAP percentage change which was within 19.4%. Fifty-two stations (2% of the total weather stations) were mild outliers (between 1.5 interquartile range (IQR) and 3.0 IQR) of AAP with increase in percentage from 19.4 % to 29%, and 70 stations (3% of the total weather stations) were extreme outliers (greater than 3.0 IQR) of AAP with increase in percentage from 29.3% to 260%. A total of 44 stations accounting for 2% of the total weather stations were mild outliers of AAP with decrease in percentage from 18.4% to 28%. There were 26 extreme outliers of AAP with decreased percentage (accounting for 1% of total weather stations) from 28.4 % to 67.9%. Overall, 1756 stations had higher average annual precipitation while 771 had lower average annual precipitation than that obtained based on CCD.

Table 3.1 A summary of AAP outlier analysis for the lower map in figure 3.1

	Stations with increases in AAP from current to updated CLIGEN database			Stations with decreases in AAP from current to updated CLIGEN database		
	Non-outlier	Mild outlier	Extreme outlier	Non-outlier	Mild outlier	Extreme outlier
Total number of stations	1634	52	70	701	44	26
Percentage (%)	65	2	3	27	2	1

From the perspective of outlier spatial distribution, the majority of outliers were distributed in the western U.S. where mountainous areas and forests are located and where climate is highly variable, and only a few outliers were distributed in the eastern U.S. Historical data periods were not necessarily longer for outliers in the eastern U.S. than those in the western U.S. They ranged from 15 - 85 years. More work and evaluation would be needed in the future to better understand the differences. Based on the analysis, the Great Lakes Region had more matched weather stations in the two CLIGEN databases and less AAP percentage change outliers (figure 3.1 insets). Therefore, further efforts were focused on the five states in the Great Lakes Region, namely WI, IL, IN, MI, and OH.

3.2.3 Statistical comparison between observed precipitation and precipitation estimates based on UCD

A comparison of statistical characteristics between the historical precipitation time series and UCD-based simulated precipitation time series were conducted to evaluate the accuracy of precipitation simulation based on the updated database. Eight weather stations covering four counties which showed large increases in urban runoff (based on preliminary L-THIA modeling with UCD-based precipitation estimates and NLCD 2011 data) were selected from the study area (table 3.2).

Table 3.2 Weather station information for selected sites

Site location	Station #	Latitude (°N)	Longitude (°W)	Elevation (m)	Precipitation (mm/yr)*	% of missing data
Cook, IL	IL111577	41.78	87.75	185	985	0.5
	IL112736	42.03	88.28	249	915	4.6
Lorain, OH	OH332599	41.38	82.07	222	1007	1.9
	OH336118	41.25	82.62	219	967	0.1
Kent, MI	MI203429	43.18	85.25	256	887	0.2
	MI204078	42.97	85.08	225	835	7.8
Ottawa, MI	MI203858	42.78	86.12	182	962	9.8
	MI205712	43.17	86.23	192	856	0

*Average Annual precipitation was calculated by summing up the observed monthly average precipitation from CLIGEN output.

Historical daily precipitation time series from 01/01/1974 to 12/31/2013 for the eight weather stations were downloaded from the National Oceanic and Atmospheric Administration website (http://www.ncdc.noaa.gov/cdo-web/search?datasetid=GHCND). Six out of the eight weather stations had relatively complete historical precipitation records, as the percentages of missing data were less than 4.7%. Two weather stations had fairly good historical precipitation records with 7.8% and 9.8% of values missing. Among the weather stations that had more than 0.1% missing data, the missing values occurred with different patterns, such as missing one or two values here and there, missing for a whole month, or missing several months in a row (only seen at station IL112736 in 1984).

Fifty years of daily precipitation for the eight weather stations were generated by CLIGEN v5.3 using UCD, with the default random number seed and without monthly interpolation. This was done so as to maintain consistence with previous work (Chen et al., 2017) as subsequent results from L-THIA modeling would be compared with those from the previous work. Historical and CLIGEN-generated precipitation time series, including daily (with and without zero precipitation), maximum daily per year, monthly, and yearly precipitation time series, were compared. Statistics included mean, standard deviation, skew coefficient, kurtosis coefficient, percentiles, and maximum daily precipitation. A preliminary analysis of daily precipitation distributions revealed that non-zero precipitation occurred after the 50th percentile. Therefore, deviations of daily precipitation were presented for the 75th, 90th, 97.5th, and 99.5th percentiles. Relative errors were calculated for the annual precipitation time series at the 10th, 25th, 50th, 75th, 90th, 97.5th, and 99.5th percentiles. A nonparametric test, the Wilcoxon rank sum test at 0.05 significance level, was used to test the null hypotheses that the two groups (observed precipitation and CLIGEN-simulated precipitation) did not express significant differences (Zhang and Garbrecht 2003, Haynes 2013). Besides a test of significance, an effect size that measures the size of differences between two groups, Cohen’s d (Cohen 1988, Kotrlik and Williams

2003, Walker 2007), was also computed. Cohen's d was estimated using the following equations (Cohen 1988),

$$Cohen's\ d = \frac{Difference\ between\ sample\ means}{Pooled\ standard\ deviation\ (SD_{pooled})} \quad (1)$$

$$SD_{pooled} = \sqrt{\frac{(SD_1^2 + SD_2^2)}{2}} \quad (2)$$

where SD_1 is the standard deviation for group 1 (observed precipitation) and SD_2 is the standard deviation for group 2 (UCD-based precipitation estimates). Cohen suggested that the d values of 0.2, 0.5, and 0.8 are considered as small, medium, and large effect sizes, which denote that the two means from observed precipitation and UCD-based precipitation estimates differ by 0.2, 0.5, and 0.8 of a standard deviation respectively (Cohen 1988, Kotrlik and Williams 2003, Walker 2007).

3.2.4 Assessment of UCD-based precipitation estimates impacts on urban runoff

With the recent updates to the CLIGEN database, it is important to know how the new input dataset may affect the CLIGEN precipitation estimates and to what extent any changes in estimated precipitation may affect the urban runoff simulations. By assuming a representative climate generated using long-term statistics for stations in the regions, the L-THIA Tabular Tool (Chen et al. 2017) was configured based on the new precipitation datasets generated from the CLIGEN simulations using an updated database and default random seed with all other being kept the same so as to avoid introducing confounding effects. The tool was used to quantitatively assess UCD-based precipitation estimates impacts on urban runoff simulations in WI, IL, IN, MI, and OH.

Input data for the L-THIA Tabular Tool datasets included the 50-year daily precipitation time series generated based on UCD, the NLCD 2011 edition datasets for 2001, 2006, and 2011 (Fry et al. 2011, Homer et al. 2007, Homer et al. 2015), and the State Soil Geographic hydrologic soil group data (Wolock 1997). The NLCD 2011 edition product is a decade of consistently produced land cover datasets for the U.S. (Homer et al. 2015). Homer et al. (2015) suggested that NLCD 2001 (2011 Edition) and NLCD 2006 (2011 Edition) must be used when any comparison for direct change is conducted. The L-THIA Tabular Tool was set up based on U.S. county boundaries. A total of 437 counties were included in the analysis for the study area. The average annual runoff depth was calculated from pixels belonging to developed land as indicated by categories 21 (developed open space), 22 (developed low intensity), 23 (developed medium intensity), and 24 (developed high intensity) in NLCD and then normalized by county area to obtain NAARD to facilitate comparisons among counties. The final output of NAARD from urban land for each county as obtained using UCD-based precipitation estimates was compared with results from the Chen et al. (2017) study on NAARD from urban land, which used CCD for precipitation estimates.

3.3 Results and Discussion

3.3.1 Statistical comparison of UCD-based precipitation estimates with observed data

As shown in table 3.3, UCD-based estimates were able to adequately replicate the means, standard deviations, and skewness coefficients of daily precipitation (non-zero) compared to historical values across all eight stations with small relative errors ranging from 1% to 4%. Deviations obtained for daily precipitation with zero values did not differ substantially from those obtained when zero values were not included, even though the values (means, standard deviations, and skew coefficients) were different. Maximum daily precipitation was underestimated at six stations except station MI203858 (1% overestimated) and station OH332599 (exactly the same). Kurtosis coefficient, which is the indicator of the peakedness of a distribution (Ruppert 1987), tended to be underestimated by the updated CLIGEN across all scales. This is possibly due to the CLIGEN parameterization, which is based on average monthly values. More details are presented in the general discussion (section 3.4). Maximum monthly precipitation tended to be underestimated across all scales for UCD-based estimates. For the annual maximum daily precipitation time series, the relative errors of the means and standard deviations were larger than those of the daily precipitation time series. However, these relative errors (range 2% to 11% for the means) were small. With the updated database, CLIGEN simulated precipitation had fairly similar skewness when compared with that for the annual maximum daily precipitation from observed data. For monthly precipitation, UCD-based estimates were generally close to observed data with respect to means, standard deviations, and skewness coefficients at seven stations (not at station MI203858). Large fluctuations in historical monthly precipitation at station MI203858 were seen in 2000 to 2003 and in 2011 (Appendix A, figure A2). The CLIGEN parameter files were created based on historical monthly means. The approaches that CLIGEN employs to generate the precipitation are based on monthly average values, so these large fluctuations could be the reason for the larger deviations between simulated and observed values. The historical monthly precipitation time series for the other seven stations (Appendix A, figure A1) were relatively more stable than MI 203858. The means of the annual precipitation were closely simulated compared to the observed values with an average relative error of 3.25% across all eight stations. Based on Fatichi et al. (2016) the internal climate variability (stochasticity) for precipitation is likely to be between 10% - 30% at the annual scale. Thus, results from this study were reasonable. Large relative errors in standard deviations were seen at IL112736 (40%) and MI203858 (49%). Large magnitude fluctuations were observed from the historical annual precipitation time series of these two stations (figure 3.2). Results suggest that large variations in precipitation over time may decrease the reliability of CLIGEN-generated values.

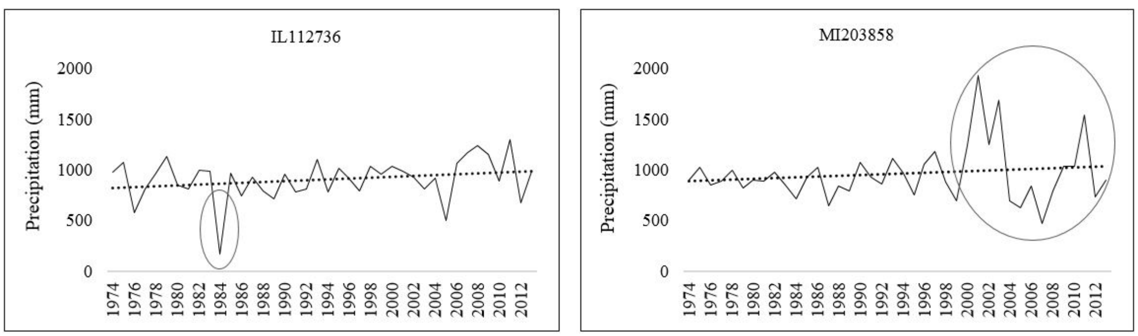
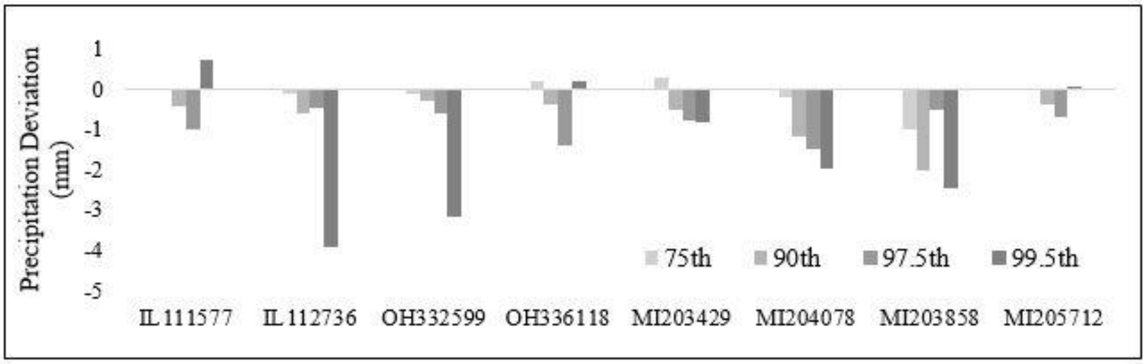


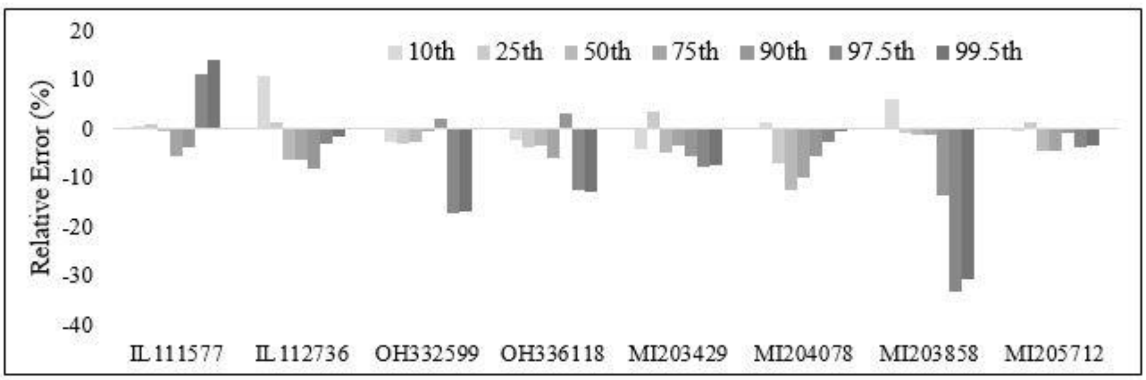
Figure 3.2 Historical annual precipitation depths for IL112736 and MI203858

The distributions are important from a modeling point of view, especially the daily precipitation simulations. The extremes are of greater importance from a runoff point of view. Figure 3.3 shows the deviations and relative errors of precipitation by percentiles, which could reflect the differences in distributions, between observed and CLIGEN-generated values at daily (with zero), annual maximum daily, and annual scales. The positive values indicate CLIGEN overestimated the precipitation and negative values indicate precipitation was underestimated by CLIGEN using the updated database.

Deviations of the daily distributions of observed and simulated precipitation values were small (0 – 4 mm) up to the 99.5th percentile across all stations (figure 3.3a), indicating that CLIGEN v5.3 using the updated database simulated daily precipitation fairly well. Well simulated daily precipitation time series could add confidence when these data are used in hydrological modeling applications. Deviations obtained for data without zero values did not differ substantially from those obtained when zero values were included, even though the values were different (1.6 – 6 mm). Exceptions were seen at station MI203858 with a difference of 21 mm at the 99.5th percentile. Monthly precipitation deviations were small (1 – 26 mm) up to the 97.5th percentile except at station MI203858. A comparison between annual maximum daily precipitation based on simulated values and the corresponding observed values (Figure 3.4) showed that CLIGEN performance based on the updated database is station dependent. UCD-based annual maximum daily precipitation estimates captured observed values very well at station OH332599. However, for other stations, CLIGEN simulations using the updated database tended to underestimate the extremes, especially for storms exceeding the 50-year return period. From a runoff point of view, underestimated maximum precipitation may lead to runoff underestimation. For annual precipitation distributions, relative errors were relatively small (2% - 8%) up to the 90th percentile except at station MI203858 (figure 3.3b).



a. Deviations of daily precipitation (with zero) between historical records and UCD-based estimates



b. Relative errors of annual precipitation between historical records and UCD-based estimates

Figure 3.3 Deviations and relative errors of precipitation between historical records and CLIGEN simulations using updated database by percentiles and time scales. (Note: deviation = UCD-based estimates - Historical records. Positive values indicate that CLIGEN simulations overestimated precipitation and negative values indicate CLIGEN underestimated precipitation.)

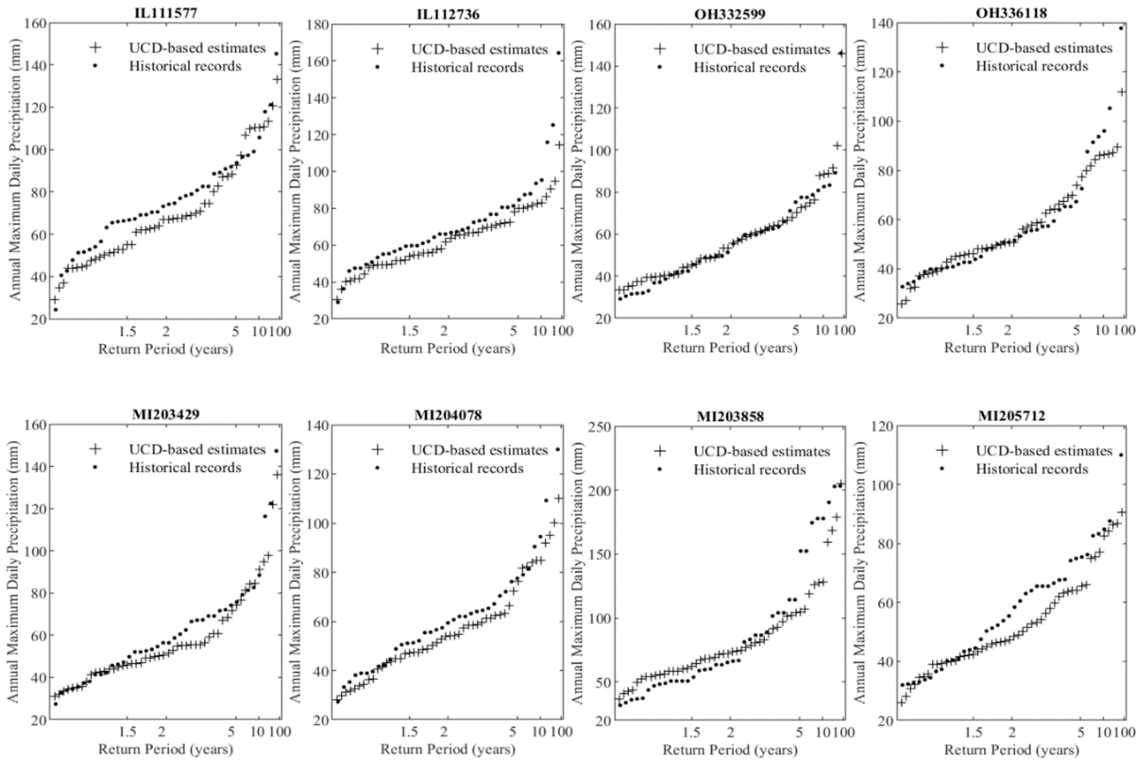


Figure 3.4 Comparison of annual maximum daily precipitation with historical records

The Wilcoxon rank sum test ($\alpha = 0.05$) rejected the null hypothesis that, for any of the stations, pairs of observed and simulated time series (daily–with and without zero- and monthly) were likely to be derived from the same population ($p < 0.0001$ at all stations for daily, and p values were up to 0.0009 for monthly). However, Cohen's d values at these sites were small (0.01 to 0.19), indicating that the difference of means of the observed and simulated data could be considered trivial, even if the differences were found statistically significant (Walker 2007). Wilcoxon p values give the probability of detecting the differences in the data by chance alone, and this probability goes down as the size of the sample goes up (Walker 2007). This test of significance may be biased by large sample size, particularly with respect to daily precipitation time series (Zhang and Garbrecht

2003). For annual precipitation, the Wilcoxon rank sum test failed to reject the null hypothesis at 6 stations except at MI204078 and MI203858. Cohen's d values of these two sites (0.39 and 0.26, respectively) were less than 0.5, suggesting that differences between simulated and observed values were not large. For annual maximum daily precipitation time series, the Wilcoxon rank sum test rejected the null hypothesis for half of the stations; however, the effects between the two samples were still considered small based on Cohen's d values (0.05 - 0.37).

Table 3.3 Statistics for daily, maximum daily per year, monthly, and annually precipitation by sites and sources.

(Note: H = Historical records; C = UCD-based estimates; RE = Relative Error = [(UCD-based estimates- Historical records) / Historical records] *100%)

3a. Statistics of daily precipitation with non-zero values of UCD-based estimates and historical records by eight sites

	Cook, IL						Lorain, OH						Kent, MI						Ottawa, MI					
	IL111577			IL112736			OH332599			OH336118			MI203429			MI204078			MI203858			MI205712		
	H	C	RE (%)	H	C	RE (%)	H	C	RE (%)	H	C	RE (%)	H	C	RE (%)	H	C	RE (%)	H	C	RE (%)	H	C	RE (%)
Mean (mm)	7.9	7.8	-1	8.3	8.2	-1	7.3	7.3	-1	7.3	7.2	-1	6.3	6.1	-3	7.0	6.9	-2	8.9	8.5	-4	6.0	5.9	-2
Standard deviation (mm)	11.7	11.6	-1	11.9	11.1	-6	9.4	9.3	-1	9.7	9.4	-4	9.6	9.1	-5	10.1	9.6	-5	14.9	13.6	-9	8.8	8.4	-5
Skewness coefficient	3.5	3.6		3.4	2.8		3.3	3.5		3.4	3.1		3.9	3.7		3.5	3.4		5.4	4.2		3.7	3.3	
Kurtosis coefficient	18.9	19.7		19.3	10.8		20.4	21.5		21.3	15.3		26.6	23.6		20.6	17.1		46.1	31.3		22.5	16.4	
Maximum daily precipitation (mm)	145	133	-8	164	114	-30	146	146	0	138	112	-19	147	136	-8	130	110	-15	203	205	1	110	91	-18
Wilcoxon P value		<0.0001			<0.0001			<0.0001			<0.0001			<0.0001			<0.0001			<0.0001			<0.0001	
Cohen's d value		0.01			0.01			0.00			0.01			0.02			0.02			0.03			0.01	

Table 3.3. continued

3b. Annual Maximum daily precipitation statistics of UCD-based estimates and historical records by eight sites

	Cook, IL						Lorain, OH						Kent, MI						Ottawa, MI					
	IL111577			IL112736			OH332599			OH336118			MI203429			MI204078			MI203858			MI205712		
	H	C	RE (%)	H	C	RE (%)	H	C	RE (%)	H	C	RE (%)	H	C	RE (%)	H	C	RE (%)	H	C	RE (%)	H	C	RE (%)
Mean (mm)	75.9	70.1	-8	70.6	62.8	-11	56.9	58.0	2	58.9	57.2	-3	61.6	57.6	-6	62.3	56.9	-9	88.0	83.5	-5	58.5	52.6	-10
Standard deviation (mm)	23.4	24.7	6	24.4	16.7	-32	22.4	21.2	-5	24.7	18.9	-23	25.0	22.3	-11	23.1	19.6	-15	51.5	36.5	-29	20.4	16.5	-19
Skewness coefficient	0.6	0.7		1.7	0.6		1.7	1.7		1.6	0.7		1.5	1.6		1.2	0.8		1.1	1.5		0.7	0.7	
Kurtosis coefficient	1.2	-0.2		4.9	0.5		5.1	4.8		2.3	0.1		3.1	2.9		1.5	0.1		-0.1	2.3		0.1	-0.3	
Wilcoxon P value		0.0762			0.0008			0.5840			0.3179			0.0361			0.0302			0.0583			0.0129	
Cohen's d value		0.24			0.37			0.05			0.08			0.17			0.25			0.10			0.31	

Table 3.3. continued

3c. Monthly precipitation statistics of UCD-based estimates and historical records by eight sites

	Cook, IL						Lorain, OH						Kent, MI						Ottawa, MI					
	IL111577			IL112736			OH332599			OH336118			MI203429			MI204078			MI203858			MI205712		
	H	C	RE (%)	H	C	RE (%)	H	C	RE (%)	H	C	RE (%)	H	C	RE (%)	H	C	RE (%)	H	C	RE (%)	H	C	RE (%)
Mean (mm)	82.4	80.7	-2	78.2	74.8	-4	83.9	82.8	-1	80.8	79.0	-2	74.0	71.4	-3	75.0	66.9	-11	81.1	75.4	-7	71.7	69.7	-3
Standard deviation (mm)	50.9	49.2	-3	53.2	45.8	-14	43.0	39.6	-8	43.8	41.9	-4	45.2	41.5	-8	46.4	39.5	-15	74.6	49.7	-33	41.5	37.6	-9
Skewness coefficient	1.1	1.2		1.4	0.8		1.4	1.0		1.2	1.1		1.6	1.4		1.5	1.3		4.2	1.2		1.5	1.4	
Kurtosis coefficient	1.7	1.6		3.5	0.3		3.7	1.4		2.6	1.9		5.1	3.3		5.0	2.1		29.5	2.1		4.7	4.1	
Maximum monthly precipitation (mm)	331	283	-15	398	225	-43	326	257	-21	323	266	-18	340	338	-0.4	374	263	-30	767	291	-62	344	300	-13
Wilcoxon P value		0.0008			0.0002			0.0046			0.0009			<0.0001			<0.0001			<0.0001			<0.0001	
Cohen's d value		0.03			0.07			0.03			0.04			0.06			0.19			0.09			0.05	

Table 3.3. continued

3d. Annual precipitation statistics of UCD-based estimates and historical records by eight sites

	Cook, IL						Lorain, OH						Kent, MI						Ottawa, MI					
	IL111577			IL112736			OH332599			OH336118			MI203429			MI204078			MI203858			MI205712		
	H	C	RE (%)	H	C	RE (%)	H	C	RE (%)	H	C	RE (%)	H	C	RE (%)	H	C	RE (%)	H	C	RE (%)	H	C	RE (%)
Mean (mm)	985	969	-2	911	898	-2	1007	994	-1	969	948	-2	886	856	-3	859	803	-7	961	905	-6	860	836	-3
Standard deviation (mm)	156	153	-2	206	124	-40	152	136	-11	150	136	-9	121	108	-11	153	132	-13	277	142	-49	132	126	-5
Skewness coefficient	0.0	0.5		-1.2	0.8		0.9	0.3		0.1	0.3		0.2	0.0		-0.6	0.7		1.6	0.6		0.4	0.5	
Kurtosis coefficient	-0.6	1.0		3.3	0.9		2.9	-0.8		0.9	-0.7		-0.7	-0.5		1.0	0.1		3.9	0.4		-0.5	0.1	
Maximum annual precipitation (mm)	1261	1437	14	1297	1275	-2	1542	1281	-17	1391	1213	-13	1170	1083	-7	1160	1157	-0.2	1931	1339	-31	1169	1131	-3
Wilcoxon P value		0.3274			0.2369			0.4484			0.3041			0.1023			0.0041			0.0055			0.0885	
Cohen's d value		0.10			0.08			0.09			0.15			0.25			0.39			0.26			0.19	

3.3.2 UCD-based precipitation estimates impacts on L-THIA predicted urban runoff

The NAARD from urban areas based on NLCD 2011 data were assessed by using UCD-based daily precipitation estimates. The NAARD results were classified using the same criteria -standard deviation classification method- as the published results (Chen et al. 2017) (upper left: published results, upper right: simulated results, figure 3.5). Runoff from urban areas in each county of the five states of the U.S. Great Lakes Region (WI, IL, IN, MI, and OH) were classified into four categories, including low (green), medium (yellow), high (orange), and very high (red) urban runoff. By using UCD-based precipitation estimates, more yellow, orange, and red appeared in the NAARD map (upper right, figure 3.5). The total number of counties in these five states belonging to the medium, high, and very high urban runoff classes increased by 31%, 67%, and 14%, respectively.

The NAARD changes by using precipitation estimates generated based on CCD and UCD with NLCD 2011 data are shown in figure 3.5 (lower map). The standard deviation classification method was also adopted. Light blue represents the NAARD from urban areas that decreased when UCD-based precipitation estimates were used. The magnitude and the total amount of counties with decreased NAARD were relatively small. Light green, yellow, orange, and red represents low increases, medium increases, high increases, and very high increases in NAARD from urban areas, respectively. The magnitudes of increases were much higher than those of decreases. A majority of counties (88%) in the study region showed increases in NAARD from urban areas based on UCD-based precipitation estimates and NLCD 2011 data. Overall, about 0.57 billion cubic meters more of average annual runoff from urban areas were simulated by using UCD-based precipitation estimates compared to those using the CCD-based one considering the NLCD 2011 data.

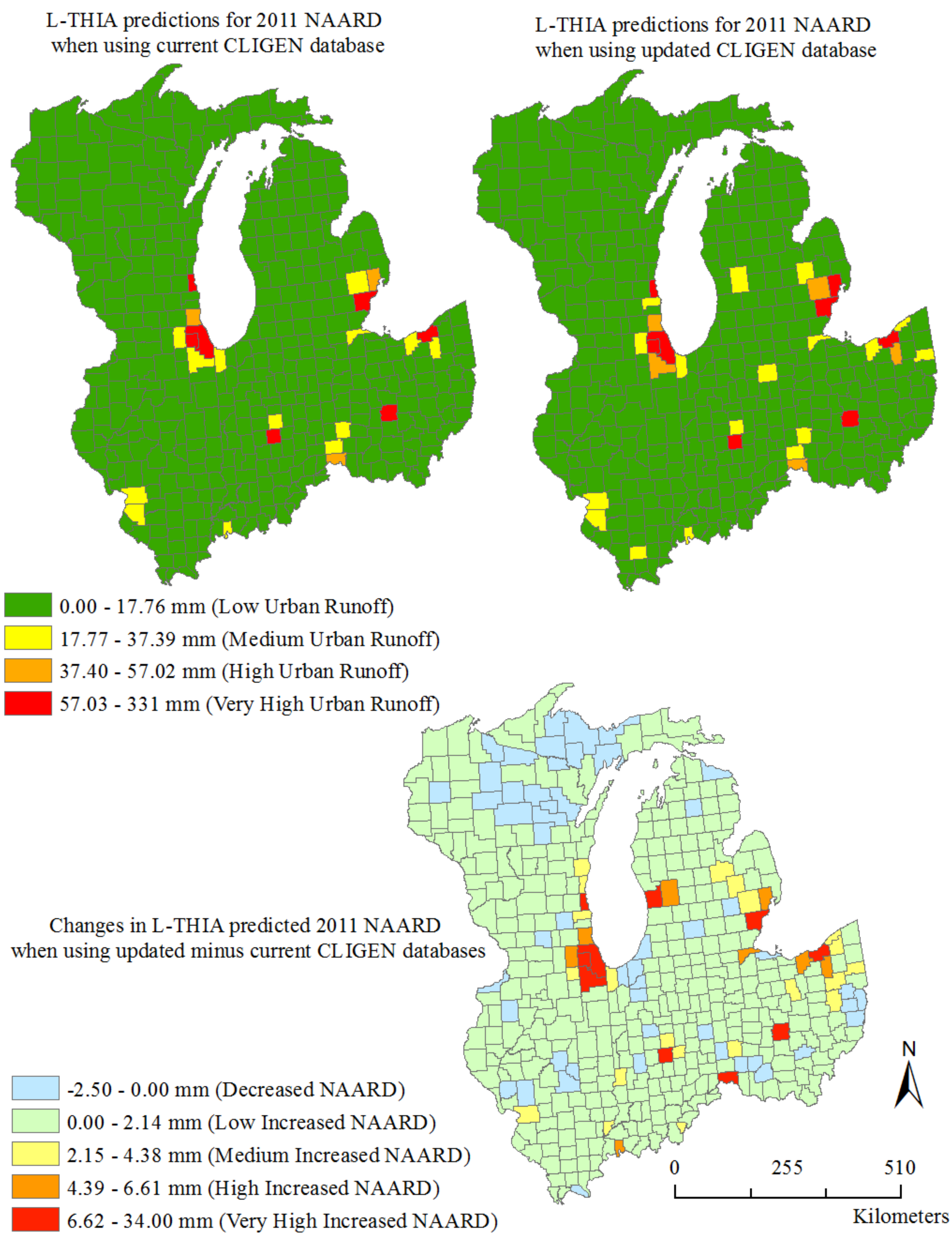


Figure 3.5 L-THIA predicted NAARD results by using the current (Chen et al., 2017) and updated CLIGEN databases based on NLCD 2011 data

Eleven counties that had changed urban runoff categories by using UCD-based precipitation estimates compared with using CCD-based precipitation estimates were selected in order to further explore how the historical precipitation had changed. Thirty-two weather stations covering the eleven counties were selected. Their historical period included in the databases as well as the observed and simulated average annual precipitation were reported (table 3.4). The historical period included in CCD ranged from 41 years to 93 years until 1992. Thirty weather stations out of 32 had increased observed AAP with the differences ranging from 23 mm to 167 mm. Only two stations had decreased AAP; moreover, those two stations covered just a small portion of Oakland County, MI and Mahoning County, OH. Average annual precipitation obtained from CCD-based and UCD-based precipitation values did not differ substantially from the observed AAP, even though the values were different (AAP differences ranged from 12 mm to 166 mm, and these two stations had decreased AAP with different magnitudes). These results indicated that the increases in urban runoff might be caused by increases in the CLIGEN precipitation estimates based on the updated database in comparison to those based on the current database. However, the missing values in the historical precipitation records might also impact AAP calculations from the CLIGEN outputs. Further research could be done to explore how the missing values in historical records impact AAP. The impacts of increases in precipitation estimates on urban surface runoff simulations were particularly prominent in highly urbanized areas (counties in red, figure 3.5, lower map), such as Cook County (IL), DuPage County (IL), Cuyahoga County (OH), Wayne County (MI), Franklin County (OH), Milwaukee County (WI), Marion County (IN), Will County (IL), and Hamilton County (OH). Since CLIGEN utilizing the updated database could simulate daily precipitation fairly well up to the 99.5th percentile but tended to underestimate the extremes, it is possible that actual urban runoff might have been greater.

Table 3.4 Historical period included in the databases and the observed and simulated AAP of stations in the counties that had changed urban runoff categories by using CCD-based and UCD-based precipitation estimates

County	CLIGEN station	Historical period used in CCD (Years)	Observed AAP of CCD station (mm)	Observed AAP of UCD station (mm)	Observed AAP differences from CCD to UCD (mm)	Simulated AAP of CCD station (mm)	Simulated AAP of UCD station (mm)	Simulated AAP differences from CCD to UCD (mm)
Racine, WI	WI475479	1949-1992 (44)	823	874	51	820	854	34
	WI476922	1949-1992 (44)	850	889	39	847	873	26
	WI471205	1952-1992 (41)	811	837	26	800	816	16
Will, IL	IL111577	1928-1992 (65)	818	985	167	803	969	166
	IL116616	1952-1992 (41)	904	985	81	862	952	90
	IL110338	1901-1992 (92)	871	941	70	857	905	48
Williamson, IL	IL111265	1910-1992 (83)	1089	1127	38	1075	1087	12
Allen, IN	OH336465	1936-1992 (57)	862	893	32	849	897	48
	IN123037	1948-1992 (45)	918	956	38	914	949	35
	OH338609	1936-1992 (57)	922	946	23	911	925	14
Kent, MI	MI203429	1948-1992 (45)	828	887	59	811	856	45
	MI203661	1948-1992 (45)	825	940	115	799	933	134
	MI203858	1948-1992 (45)	886	962	76	842	905	63
	MI205712	1948-1992 (45)	819	856	38	791	836	45
	MI203769	1948-1992 (45)	792	831	39	772	814	42
Genesee, MI	MI202846	1948-1992 (45)	774	803	29	748	791	43
	MI206300	1948-1992 (45)	729	769	41	708	752	44
	MI201299	1948-1992 (45)	731	775	44	710	733	23
	MI205452	1948-1992 (45)	781	736	-45	757	723	-34
	MI203477	1950-1992 (43)	823	869	46	807	859	52
	MI202015	1952-1992 (41)	772	838	66	753	834	81
Macomb, MI	MI206680	1948-1992 (45)	793	833	40	795	807	12
Summit, OH	OH331657	1948-1992 (45)	942	1002	60	932	984	52
	OH333780	1900-1992 (93)	991	1094	103	982	1085	103
	OH331541	1936-1992 (57)	944	999	55	951	992	41
	OH330058	1948-1992 (45)	918	1011	93	910	1002	92

Table 3.4 continued

County	CLIGEN station	Historical period used in CCD (Years)	Observed AAP of CCD station (mm)	Observed AAP of UCD station (mm)	Observed AAP differences from CCD to UCD (mm)	Simulated AAP of CCD station (mm)	Simulated AAP of UCD station (mm)	Simulated AAP differences from CCD to UCD (mm)
Lake, OH	OH336389	1950-1992 (43)	923	998	75	912	978	66
	OH331458	1945-1992 (48)	1149	1261	112	1136	1239	103
Mahoning, OH	OH339406	1948-1992 (45)	952	1007	54	937	1001	64
	PA366233	1926-1992 (67)	970	999	29	971	992	21
	OH338769	1936-1992 (57)	921	1009	88	907	1012	105
	OH335315	1936-1992 (57)	958	930	-28	954	907	-47

Note: Historical period used in UCD is 40 years from 1974 to 2013. Observed AAP of CCD and UCD were calculated based on observed monthly average precipitation shown in CLIGEN output. Simulated AAP based on CCD and UCD were calculated based on 50 years of simulated daily precipitation.

3.3.3 Comparison of NAARD change based on NLCD 2001, 2006 and 2011 by using CCD-based and UCD-based precipitation estimates

Different precipitation databases for CLIGEN result in changes in urban runoff simulations. Table 3.5 summarizes the number of counties with decreases or increases in NAARD and the range of NAARD changes based on NLCD 2001, 2006, and 2011. In the study region (MI, IN, IL, WI, and OH), the total number of counties that had increased urban runoff is about 7.5 times the number of counties with decreased urban runoff across the three NLCDs (table 3.5). Based on the NAARD change range, the maximum decreases of urban runoff from NLCD 2001 to NLCD 2006, NLCD 2006 to NLCD 2011, and NLCD 2001 to NLCD 2011 are 0.02mm, 0.01mm, and 0.03mm. However, the maximum increases of urban runoff from NLCD 2001 to NLCD 2006, NLCD 2006 to NLCD 2011, and NLCD 2001 to NLCD 2011 are 0.34mm, 0.28mm, and 0.62mm. These indicated that the increases in simulated urban runoff were higher than the decreases.

Table 3.5 NAARD change results of using CCD-based and UCD-based precipitation estimates

	Number of counties with decreased NAARD	NAARD decrease range (mm)	Number of counties with increased NAARD	NAARD increase range (mm)
NLCD 2011	51	(-2.45, -0.005)	386	(0.0003, 33.80)
NLCD 2006	52	(-2.44, -0.004)	385	(0.0024, 33.52)
NLCD 2001	52	(-2.42, -0.003)	385	(0.0028, 33.18)

Table 3.6 Changes in the number of counties in different NAARD categories by using CCD-based and UCD-based precipitation estimates

NAARD Categories	NLCD 2001	NLCD 2006	NLCD 2011
-2.5 - 0 mm (Decreased NAARD)	52	52	51
0 - 2.14 mm (Low Increased NAARD)	353	351	347
2.15 - 4.38 mm (Medium Increased NAARD)	17	18	21
4.39 - 6.61 mm (High Increased NAARD)	7	8	8
6.62 - 34 mm (Very High Increased NAARD)	8	8	10

Compared to the changes in the number of counties in each NAARD categories based on NLCD 2001, more counties are categorized in medium, high, and very high increased NAARD categories based on NLCD 2006 and NLCD 2011 (table 3.6). This likely indicates that urbanization over time can intensify precipitation impacts on simulated urban runoff. Adams County, IL (purple circle in figure 3.6) changed from a category of decreased NAARD (NLCD 2001 and NLCD 2006) to a category of low increased NAARD (NLCD 2011). The observed AAP of the weather station covering Adams County decreased 15.9 mm from CCD to UCD. However, the increase in urban land was large (4 square kilometers) from NLCD 2001 to NLCD 2011, suggesting that urbanization also can offset decreased precipitation impacts on simulated urban runoff.

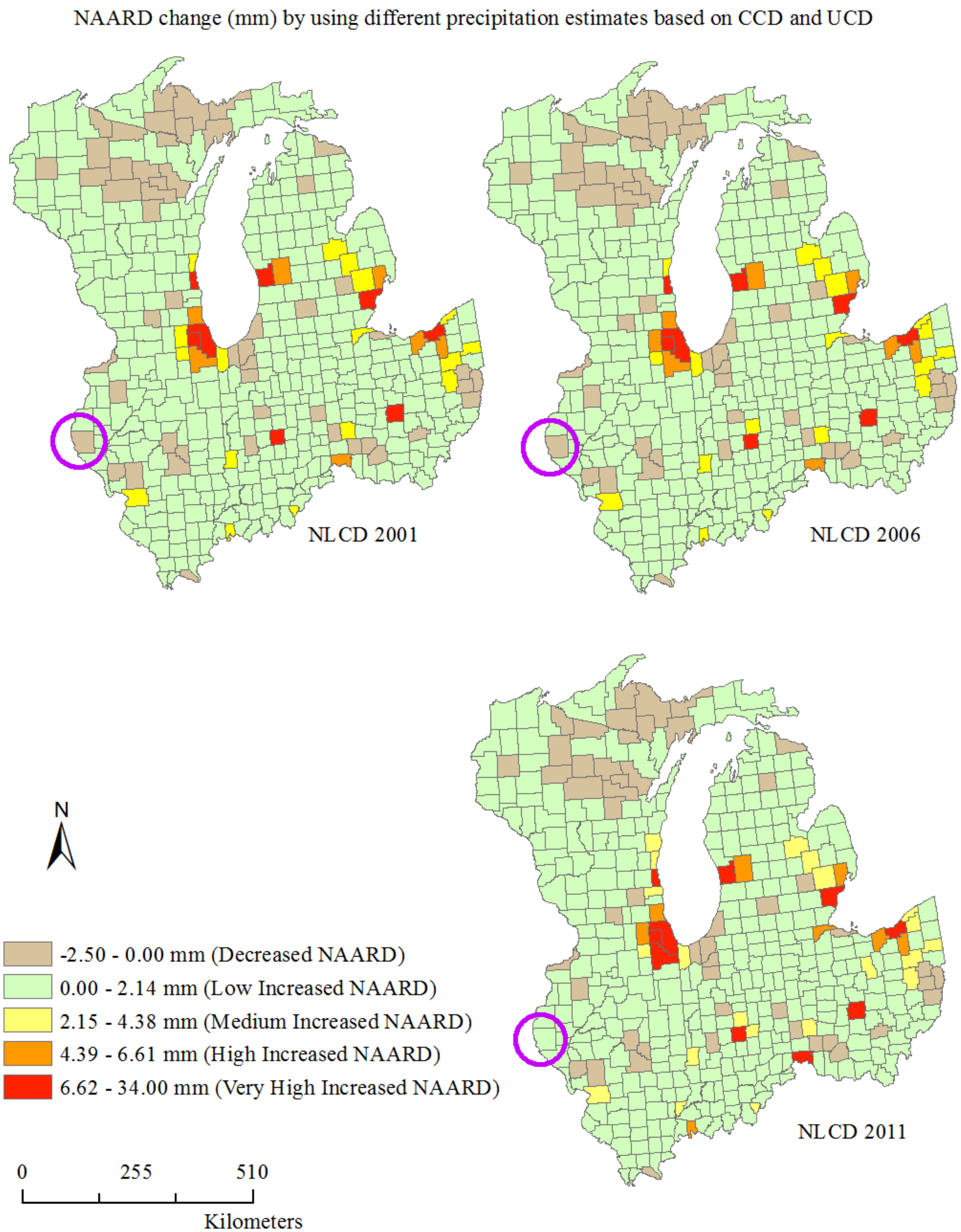


Figure 3.6 NAARD change based on NLCD 2001, 2006 and 2011 by using CLIGEN precipitation estimates generated using the current and updated databases

3.3.4 General discussion

Up-to-date historical meteorological records and their statistical parameterization are of importance for the usefulness of weather generators. The performance of CLIGEN using the updated database on precipitation estimates was evaluated in this study. Results indicated that the UCD-based precipitation estimates could adequately replicate observed daily precipitation up to the 99.5th percentile in relation to observed data in five states (WI, IL, IN, MI, and OH) of the U.S. Great Lakes Region. Other researchers who worked with CLIGEN have also reached similar conclusions. For example, Mehan et al. (2017) did not find appreciable differences in precipitation values until about the 99.5th percentile based on analysis at three stations in the Western Lake Erie Basin; Chen and Brissette (2014) concluded that CLIGEN had good performance for simulating daily precipitation amounts; and Zhang and Garbrecht (2003) evaluated CLIGEN (v5.107) on four Oklahoma sites and found that the model simulated daily and monthly precipitation and frequencies of wet and dry spells reasonably well for use in WEPP runoff and soil erosion impact studies.

However, discrepancies existed with respect to simulation of extreme precipitation events by CLIGEN. For instance, results of this study showed that CLIGEN tended to underestimate the maximum precipitation at daily, monthly, and annual scales based on 50 years of generated daily precipitation time series compared to observed data. Johnson et al. (1996) concluded that CLIGEN could adequately replicate annual and monthly extremes, while daily extreme precipitation values in any given year were not entirely satisfactorily generated, based on 300 years of simulated precipitation at six widely dispersed locations across the U.S. Chen and Brissette (2014) concluded that CLIGEN was capable of simulating extreme precipitation events in the Loess Plateau of China based on work at 54 stations and simulated precipitation for periods ranging from 320 to 520 years.

Mehan et al. (2017) found that CLIGEN tended to overestimate the maximum precipitation at three stations in the Western Lake Erie Basin based on 50 realizations of 50-year daily precipitation time series simulated by CLIGEN. These discrepancies in extreme precipitation simulations might be attributable to different weather station locations, different lengths of historical measurements in the database, and different numbers of realizations. Other factors might include different lengths of simulated data in relation to length of input time series and selection of base time period (Gitau et al. 2017). Large variations in precipitation over time may decrease the reliability of CLIGEN-generated values since CLIGEN parameterization is based on average monthly values. The more fluctuations or unstable trends there are in the historical precipitation, the greater the possibility that CLIGEN will overestimate or underestimate the generated values. However, CLIGEN parameterization could be based on any desired historical period to improve precipitation estimates in relation to project objectives. Parameter values also need to be examined when precipitation climatology has significantly changed (Yu 2005).

The CLIGEN database has been recently updated to include 2,702 weather stations across the nation with each station having the same time period of historic measurements from 1974 to 2013. To determine a suitable length of historical measurements that would be used for CLIGEN parameterization, historical records of 20, 30, 40, 50, and 60 years were statistically compared, and 40 years were considered as a suitable time period for the updated CLIGEN database. Given its nationwide scope and the amount of effort required to update the database, a consistent time period across all stations was adopted to save time, money, and energy. There is, however, no restriction on the length of base input time series for CLIGEN parameterization, although users would need to generate parameter files manually. Different lengths of base input time series have been adopted based on historical records availability (Mehan et al. 2017, Hoomehr et al. 2016, Kim et al. 2009, Li et al. 2011, Pyke 2005, Yu 2003, 2005, Zhang and Garbrecht 2003, Zhang 2003, 2005, 2006, 2007, Zhang et al. 2010). Providing users with the flexibility to select/change the base period for input data with automated parameter file generation when using CLIGEN would help facilitate its use and increase the reliability of generated values.

Few studies have evaluated the impacts of missing values in the historical records on CLIGEN performance. The program for CLIGEN parametrization excludes a station if there are 6 or more of the same month missing within 40 years. For example, if a station only contains 33 June records out of 40, it is not included in the station list. For partial months where some of the days have missing data, only the days in the month that have valid data are used in calculating the .par file results. The program might be improved to identify those months that could be completely excluded if a certain number of days are missing.

In this study, one realization of 50 years of daily precipitation was generated using CLIGEN with the updated database and based on the default random seed, consistent with prior work done using the current database. Results showed that the maximum precipitation was underestimated. To better capture variability within the generated values by CLIGEN, more realizations could be considered. Based on recent research (Guo et al., 2017; Gitau et al., 2018), 25 realizations would be sufficient to capture the essential characteristics of observed climate including precipitation extremes. Since results from L-THIA modelling are considered at an average annual level, data from one of the stations in the study region were examined to determine how average annual precipitation based on different realizations would fare in comparison to observed data. Figure 7 shows a comparison between the average annual precipitation based on historical records and that from 50 realizations of fifty years daily precipitation from CLIGEN. Based on the analysis, the average annual precipitation from UCD-based estimates with the default random seed (highlighted with red circle in figure 3.7) was relatively close to the observed value. This value could have been overestimated or underestimated, depending on the random seed, thus the strength of multiple realizations.

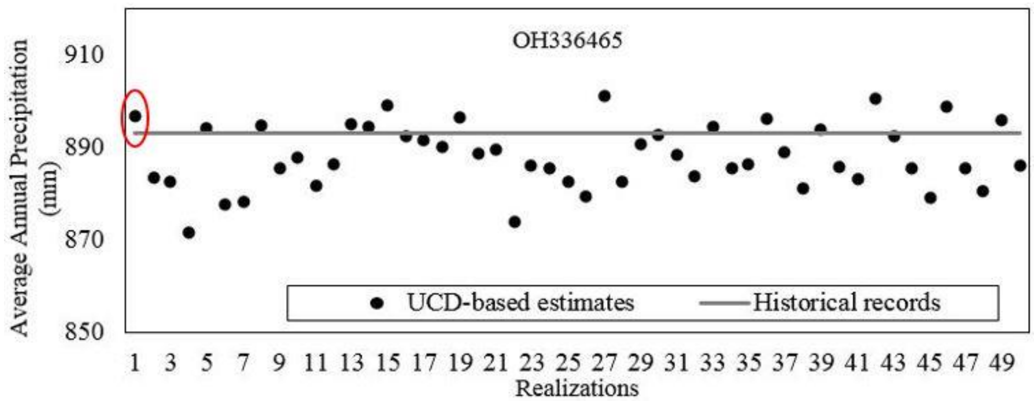


Figure 3.7 Average annual precipitation between historical records and predicted with 50 realizations using CLIGEN with the updated database

The current CLIGEN model and database have been used in numerous studies, and these studies have achieved valuable results in different applications (Baffaut et al. 1996, Chen et al. 2017, Pruski and Nearing 2002, Zhang et al. 2010). However, use of the updated CLIGEN database would provide more recent and temporally-consistent climatic information. It is recommended to adopt use of the updated CLIGEN database after its performance has been more completely evaluated. In this study, urban runoff simulations based on the current and updated CLIGEN databases showed similar trends for increases in L-THIA simulated urban runoff, but with different magnitudes. Although urban runoff measurements with which to validate the simulated values are scarce, the results of hydrologic model simulation could provide information regarding the magnitude of changes or impacts to stakeholders. This study was focused on the Great Lakes region and results might not necessarily translate to other areas. Methodologies and approaches would, however, be widely applicable.

3.4 Conclusions

In this paper, the observed average annual precipitation from the current and the updated CLIGEN databases was compared nationally in the contiguous U.S. The performance of the CLIGEN precipitation estimates when using the updated database were evaluated further in five states (WI, IL, IN, MI, and OH) of the U.S. Great Lakes Region. Finally, the impacts of CLIGEN precipitation estimates utilizing the updated database on L-THIA urban runoff simulations were assessed and also compared with published results based on CLIGEN precipitation estimates generated with the current database.

The national comparison of observed average annual precipitation between weather stations in the current and updated CLIGEN databases indicated that differences existed in average annual precipitation, possibly due to different observation period lengths and consistency of historical records. Statistical analysis revealed that CLIGEN precipitation estimates with the updated database could adequately replicate daily precipitation up to the 99.5th percentile compared to observed data. However, maximum precipitation at the daily, monthly, and annual scale tended to be underestimated. Large precipitation variations over time were found to decrease the reliability of CLIGEN-generated values. CLIGEN parameterization could be based on any desired historical period to improve precipitation estimates in relation to project objectives. However, exclusion of recent data in CLIGEN parameterization while using generated values to represent present or future climate conditions could be problematic. This study demonstrated that CLIGEN based on the updated database is suitable for simulating precipitation in five states (WI, IL, IN, MI, and OH) of the U.S. Great Lakes Region. CLIGEN with the updated database was also found preferable with respect to modeling urban runoff or urbanization effects since it includes up-to-date historical and temporally-consistent measurements. The assessment of urban runoff based on simulated precipitation from CLIGEN with the updated database can provide insights for public and planning agencies to assist sustainable development. When the only change in input of the L-THIA model is climate parameters used by CLIGEN, variations in simulation results can reflect the impacts of precipitation characteristics on runoff from urban areas.

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CHAPTER 4. EVALUATION OF THE EFFECTIVENESS OF GREEN INFRASTRUCTURE ON HYDROLOGY AND WATER QUALITY IN A COMBINED SEWER OVERFLOW COMMUNITY

Abstract

The adverse impacts of urbanization and climate can converge in synergistic ways generating more urban runoff. Green infrastructure (GI) practices are on-site management approaches aimed at reducing urban runoff and controlling the movement of pollutants. Evaluation of the effectiveness of GI practices on improving hydrology and water quality and their associated cost could provide valuable information for decision makers when making development/re-development strategies. In this study, a watershed scale rainfall-runoff model (the Long Term Hydrologic Impact Analysis - Low Impact Development model, the L-THIA-LID 2.1 model) was enhanced to improve its simulation of urban management practices. The enhanced model (L-THIA-LID 2.2) was capable of simulating more detailed impervious surfaces including sidewalks, roads, driveways, and parking lots, to conduct cost calculations for converting these impervious surfaces to porous pavements, and to consider the actual suitable area for bioretention in the study area. The effectiveness of GI practices on improving hydrology and water quality in a combined sewer overflow urban watershed - the Darst Sewershed in the City of Peoria, IL - was examined in 10 simulation scenarios using 8 practices including rain barrels/cisterns, green roofs, green roofs plus rain barrels/cisterns, bioretention, porous roads, porous parking lots, porous sidewalks, and porous driveways. The total cost and the cost effectiveness for each scenario considering a 20-year GI practice lifetime were examined as well. Main findings included: (1) combined implementation of GI practices performed better than applying individual practices alone; (2) the various adoption levels and combinations of GI practices could potentially reduce runoff volume by 0.2-23.5%, TSS

by 0.18-30.8%, TN by 0.2-27.9%, and TP by 0.20 to 28.1%; (3) based on site characteristics, adding more GI practices did not necessarily mean that substantial runoff and pollutant reduction would be achieved; (4) the most cost-effective scenario (S8, which is adoption of 50% rain barrels/cisterns, 100% bioretention, 100% porous pavements on eligible parking lanes of the roads, 50% porous parking lots, 50% porous sidewalks, and 50% porous driveways) has an associated cost of $9.21 to achieve 1 m^3 runoff reduction per year and $119 to achieve 1 kg TSS reduction per year. This, however, assumes cooperation from the general public in implementing GI practices on their properties; and (5) adoption of GI practices on all possible areas could potentially achieve the greatest runoff and pollutant load reduction, but would not be the most cost-effective option. The enhanced model from this research can be applied to various locations to support assessing the beneficial uses of GI practices. This model and its user's manual are available upon request.

4.1 Introduction

Combined sewer systems are conduits designed to gather and transfer sanitary wastewater (domestic sewage from residential, industrial, and commercial wastewater, plus stormwater) through a single pipe to a wastewater treatment facility (USEPA, 2004). These are typically owned by states or municipalities (USEPA, 2004). Combined sewer systems were the earliest sewer systems built in the U.S. and were first introduced in 1855 (Tibbetts, 2005; USEPA, 2004). During dry weather, a combined sewer system carries all sewage to the wastewater treatment plant. However, during rain or snowmelt periods, the waste water treatment facility may not be able to handle the high flow rate, and, in this case, the combined sewer is designed to overflow directly into surface waters such as a river or lake. If the combined sewer system did not release the excess sewage through the combined sewer overflow (CSO), raw sewage would back up into basements and streets.

CSO and SSOs as threats to human health and the environment, based on U. S. EPA's Report to Congress on Impacts and Control of CSO and Sanitary Sewer Overflows (SSOs) (USEPA, 2004). According to this report, there are 746 communities with combined sewer systems with 9,348 asscociated CSO outfalls; EPA estimates that about 3.2 billion cubic meters (850 billion gallons) of untreated wastewater and stormwater are released by CSO into surface waters each year in the U.S.; combined sewer systems are found in 32 states and primarily in the Northeast and Great Lakes regions (Figure 4.1). Among those 746 CSO communities, more than half of them are found in Illinois, Indiana, Ohio, Wisconsin, and Michigan (USEPA, 2008).

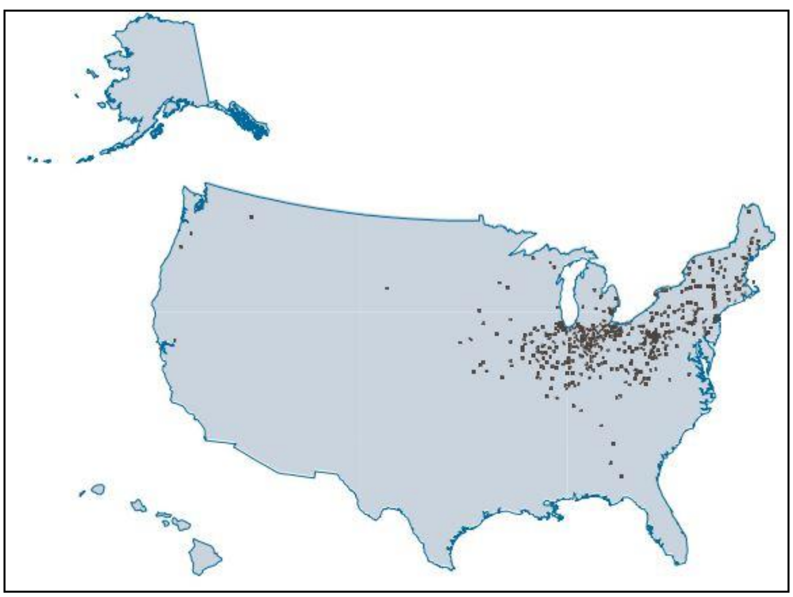

Figure 4.1 Distribution of CSO outfalls in the U.S. (brown dots) Source: USEPA, 2004

Historical weather records showed that precipitation increased from 1900 to 1994 within the contiguous U.S. and that local increases of nearly 20% were common, especially in the Great Lakes Region (Karl et al., 1996). Increased precipitation typically results in more surface runoff. A U.S. EPA screening assessment of the potential impact of climate change on CSO in the Great Lakes suggested that projected long-term (2060-2099) changes in precipitation would increase the frequency and volume of overflow discharged to rivers or lakes if CSO mitigation efforts are designed based on historical precipitation data (USEPA, 2008).

Urbanization increases the frequency and volume of urban runoff (Chen et al., 2017; Guan et al., 2016). The urban land area in the United States has expanded substantially in recent decades (USEPA, 2013). Urban expansion and intensification serve as driving forces altering surface runoff, resulting in more counties with medium and very high runoff in the contiguous U.S. (Chen et al., 2017). An estimated additional 3.3 billion cubic meters of average annual runoff were generated in the contiguous U.S. due to urbanization from 2001 to 2011 (Chen et al., 2017). Increased urbanization can amplify the impacts of increased precipitation as well as offset the impacts of decreased precipitation. Dewalle et al. (2000) estimated the potential effect of climate change and urbanization on mean annual streamflow for 39 urbanizing and 21 nearby rural basins in four regions in the U.S. These authors found that urbanization could significantly offset flow decreases or augment flow increases resulting from climate change.

Strategies that can be used to reduce the negative impacts that urbanization can have on hydrology and water quality typically involve reducing stormwater runoff rates and volumes. Sustainable urban development or re-development solutions include green infrastructure (GI) practices as on-site stormwater management approaches that increase infiltration and storage, delay runoff peaks, reduce runoff rates and volumes, and control the movement of pollutants (Benedict & McMahon, 2012; Gill et al., 2007; Tiwary & Kumar, 2014; Tzoulas et al., 2007). Large-scale GI, also known as best management practices (BMPs), include retention ponds and detention basins that are space-intensive and usually collect and treat runoff at drainage outlets (USEPA, 1999a; USEPA, 2009). Small-scale GI practices, also termed low impact development (LID) practices, include porous pavements, green roofs, and bioretention systems, seek to control the timing and volume of stormwater from impervious surfaces (Di Vittorio & Ahiablame, 2015; PGCo, 1999; USEPA, 2004).

Communities with CSO are usually located in densely developed areas, where large-scale GI practices are not always suitable. Small scale GI practices or LIDs have shown promise in these areas, increasing the infiltration and storage of stormwater and contributing to inflow control to the sewer systems (USEPA, 2004). Porous pavements, for example, are special types of pavement forming an infiltration system that allows runoff to pass through and filter some pollutants from a site and surrounding areas (USEPA, 1999b). Typically, areas most suitable for porous pavements have high soil permeability and low traffic volume; some common suitable implementation places include street parking lanes, driveways, parking lots, and sidewalks (USEPA, 2004). By using plants and underlying soil to intercept stormwater, green roofs can postpone runoff peaks, and decrease runoff rates and volume (USEPA, 2004). Green roofs can be designed for many areas, including single/multi-family homes, industrial facilities, commercial buildings, and garages; however, factors, such as roof deck load bearing capacity, the roof membrane resistance for the moisture and root penetration, wind shear, roof shape and slope, and hydraulics, must be considered before implementation (USEPA, 2004). Bioretention areas can collect, filter and infiltrate runoff from impervious areas (streets, parking lots, rooftops, etc) and thus reduce runoff rates and volumes (USEPA, 1999c). They are constructed to imitate natural vegetated areas and can be implemented in new development or be retrofitted into developed areas. Heavily urbanized areas, including street median strips, parking lots, traffic islands, sidewalks, and other impervious areas, are suitable for bioretention (USEPA, 2004).

Computer-based hydrological models can perform temporal and spatial simulations of the effects of hydrologic processes and management activities on hydrology and water quality (Moriasi et al., 2007). The Long-Term Hydrologic Impact Assessment (L-THIA) model is a user-oriented tool that requires only data on hydrologic soil groups (HSG), land use, and long-term precipitation (typically 30 years or more) to estimate surface runoff changes (Bhaduri et al., 2001; Harbor, 1994; Grove et al., 2001; Tang et al., 2005). The L-THIA-Low Impact Development (L-THIA-LID) model integrates LID design practices into a L-THIA model and has been successfully used to assess the impacts of BMPs/LID practices on surface hydrology and water quality (Ahiablame et al., 2012, 2013; Engel and Ahiablame, 2011; Hunter et al., 2010; Liu et al., 2015a, 2015b, 2016a, 2016b; Martin et al., 2015; Wright et al., 2016). For example, Ahiablame et al. (2013) simulated six scenarios of porous pavements and rain barrels/cisterns in two highly urbanized watersheds to find their effectiveness in reducing runoff and pollutant loads using the L-THIA-LID model. These authors found reductions ranging from 2% to 12% associated with the implementation of different LID scenarios. Liu et al. (2015b) evaluated 12 BMPs/LID practices for their impacts on water quantity and water quality in the Crooked Creek watershed using 16 scenarios and found that the various implementation levels and combinations of BMPs/LID practices reduced runoff volume by 0 to 26.5%, total suspended solids (TSS) by 0.3 to 53.6%, total nitrogen (TN) by 0.3 to 34.2%, and total phosphorus (TP) by 0.3 to 47.4%. The adoption of grass strips in the 25% of the watershed where this practice could be applied was found to be the most cost-efficient scenario (Liu et al., 2015b). Wright et al. (2016) examined the impacts of LID practices on surface runoff using the L-THIA-LID model in four neighborhoods in Lafayette, Indiana and found that 10%-70% runoff volume reductions could be achieved depending on the LID practices and adoption level.

Other existing models with the ability to simulate the impacts of GI practices, such as the Storm Water Management Model (SWMM) (Rossman, 2015; Rossman & Huber, 2016) and the System for Urban Stormwater Treatment and Analysis Integration (SUSTAIN) (Lai et al., 2007; Shoemaker, 2009; Shoemaker et al., 2012), have complex structures and require substantial amounts of input data that limit model uses in some large-scale watersheds. Compared to these complex models, the L-THIA-LID model only requires readily available data, is easy to use, and can be applied to a wide range of neighborhoods and watersheds.

Although numerous modeling studies have been conducted to evaluate the effectiveness of GI practices on water quantity and quality, few studies to date have focused on evaluating the possible impacts of GI practices at watershed scales. This includes simulating different implementation levels and combinations of GI practices in series and implementing porous pavements on different impervious surfaces. In addition, research that includes estimating the cost of implementing GI practices and determining their cost-effectiveness - based on different combinations and levels of implementation - at the watershed scales are scarce, despite the fact that cost-benefit analyses are critical for decision making.

In this study we provide an example of a model simulation study that was conducted for the City of Peoria in Illinois and that includes evaluations of the costs and impacts of GI practices on water quantity and quality at a watershed scale. We first enhanced the L-THIA-LID 2.1 model to simulate more detailed impervious surfaces including sidewalks, roads, driveways, and parking lots and to conduct cost calculations for these more detailed impervious surfaces considering the actual suitable area for bioretention in the model. Secondly, the long-term effectiveness of GI practices including rain barrels/cisterns, green roofs, bioretention, porous roads, porous parking lots, porous sidewalks, and porous driveways on hydrology and water quality at the Darst Sewershed was evaluated with the L-THIA-LID 2.2 model (enhanced version) for 10 scenarios with different implementation levels and combinations of practices. Finally, we analyzed the total cost, cost effectiveness, and performance of each scenario and identified the most cost-effective scenario.

4.2 City of Peoria: Description

The City of Peoria, located at 40°43′15″N and 89°36′34″W in Illinois, is the largest city on the Illinois River (population of 115,007) and was the seventh most populated city in Illinois as of the 2011 census (U.S. Census Bureau 2011). Most of the combined sewers in the City of Peoria were originally constructed between 1880 and 1930 and initially discharged directly into the Illinois River without treatment (Peoria CSO Long-Term Control Plan update, PCLTCP, 2010). After the Greater Peoria Sanitary District was formed and the interceptor sewer and treatment plant were constructed in 1931, the combined sewers only discharged directly to the river during large rainfall events (PCLTCP, 2010). The Illinois River provides the water supply for industry, recreational opportunities for boaters and fishermen, and it is also an important habitat for wildlife such as fish and waterfowl. Water quality is of great importance since the Illinois River has rich economic, social and cultural connections with the City of Peoria. Unfortunately, the City has maintained a system of combined sewers and overflows for more than 100 years, and with the developing economy, increasing urbanization, and increasing population, the CSOs have become an increasingly important issue in terms of its impact on hydrology, water quality, wildlife, and human lives (PCLTCP, 2010). In 1973, following the adoption of the federal Clean Water Act (http://www.epa.gov/watertrain/cwa/), the City of Peoria initiated a series of steps designed to control and minimize the CSO, and in 1976 a City of Peoria Facilities Plan was developed for the combined sewer system (PCLTCP, 2010). The City implemented approximately $8 million in sewer system improvements after completing several planning studies in the late 1970s and early 1980s (PCLTCP, 2010). Those improvements contributed to dealing with CSO; for example, the improvements that were completed in 1994 reduced wet weather flow to the combined sewer system, eliminated several overflow events, provided additional control to maximize in-system storage, provided swirl concentrators for the capture of floatables in the downtown area, prevented inflow from the Illinois River into the Riverfront Interceptor up to the 25-year flood level, and installed gates to keep the Riverfront Interceptor from surcharging (PCLTCP, 2010). Most of the improvements focused on outfalls discharging to Peoria Lake (downtown and northeast of downtown). There were no improvements, however, to reduce overflows to the Illinois River downstream of Peoria Lake (southwest of downtown, such as Darst Sewershed, as shown in figure 4.2). A sewershed is an analogy to the concept of a natural watershed that refers to an area draining to a single point in a stream network; sewersheds are controlled by curbs, storm drains, settling basins, pipes, and outfalls to streams. When the sewers were constructed, engineers usually took advantage of natural grades as much as possible, placing sewer pipes in streams and slopes. Darst

Sewershed, which is located in the southwest of downtown of the City of Peoria, is a typical CSO producing sewershed that needs more attention since its outfalls discharge to the Illinois River during every 10-year storm (PDGIP, 2015), and continued urbanization in this sewershed is producing more surface runoff and increased burdens on the sewer system.

4.3 Background and Enhancements of L-THIA-LID 2.1 Model

4.3.1 Model background

The L-THIA-LID 2.1 model, developed by Liu et al. (2015b), is designed to evaluate BMP and LID perfomance at the watershed scales (Liu et al., 2015b; Liu et al., 2016a; Liu et al., 2016b). The input data needed include long term daily precipitation (typically 30 years or more), HSG, and land use types, consistent with other versions of the L-THIA model (Harbor, 1994; Engel et al., 2003; Ahiablame et al., 2012; Liu et al., 2015a),. A total of 12 BMPs or LIDs are represented in this model, including rain barrels/cisterns, bioretention, green roofs, porous pavements, open wooded spaces, permeable patios, retention ponds, detention basins, wetland basins, wetland channels, biofilter-grass strips, and biofilter-grass swales (Liu et al., 2015a). The approaches for runoff volume estimation are the Curve Number method (inlcuding rain barrels/cisterns, bioretention, green roofs, porous pavements, permeable patios, and open wooded spaces) (Sample et al., 2001; Ahiablame et al., 2012) and the Percent Runoff Reduction Method (including retention ponds, detention basins, wetland basins, wetland channels, biofilter-grass strips, and biofilter-grass swales) (Liu et al., 2015a). GI practice effectiveness values are based on changes in runoff volume, the irreducible concentration method, and reductions in pollutant concentrations (Liu et al., 2015a; 2016a).

The implementation of GI practices starts at the level of hydrologic response unit (HRU), and GI practices can be simulated both individually and in series (Liu et al., 2015b). Suitable locations for GI practices need to be selected based on site characteristics. The criteria for selecting suitable locations as well as logistic concerns for rain barrels/cisterns, green roofs, biorentention, and porous pavements are shown in Table 4.1. More informatoion related to the framework for GI practices simulations at watershed scales are detailed in Liu et al. (2015b).

Table 4.1 Criteria for suitable locations for different GI practices (Shoemaker et al., 2009; Clar et al., 2004; Liu et al., 2015b)

GI practices	Drainage slope (%)	Drainage area (ha)	Imperviousness (%)	HSG	Buffer of roads (m)	Buffer of streams (m)	Buffer of buildings (m)	Logistic concens
Biorentention	<5	<0.81	>0	A-D	<30.48	>30.48	/	/
Porous pavements (roads, sidewalks, parking lots, and driveways)	<1	<1.21	>0	A-D	/	/	/	/
Rain barrels/cisterns	/	/	/	A-D	/	/	Close to buidling	/
Green roofs	/	/	/	A-D	/	/	On building	Only applied to commercial and industrial area

The total cost of implementing GI practices and the cost-effectiveness are represented in the L-THIA-LID 2.1 model (Liu et al., 2015). Total cost is estimated using equation (1) (Arabi et al., 2006):

$$Tc = Cc \times (1+s)^{dl} + Cc \times Rmc \times \left[\sum_{i=1}^{dl} (1+s)^{(i-1)}\right] \tag{1}$$

where Tc represents total cost, Cc represents construction cost, s represents interest rate, Rmc represents ratio of annual maintenance cost to construction cost, and dl represents design life of GI practice. The values of Cc and Rmc can be found in Liu et al. (2015b).

Cost per unit component reduction per year (cost-effectiveness) is calculated using equation (2) (Liu et al., 2015b):

$$C_{ur,y} = \frac{T_c}{nR} \tag{2}$$

where $C_{ur,y}$ represents cost per unit reduction per year, Tc represents total cost of implementing GI practices, n represents practice design life, and R represents runoff volume reduction (m^3) or pollutant loads (kg) on a yearly basis. Larger $C_{ur,y}$ values ($\$/m^3$/yr or \$/kg/yr) represent less cost-effective GI practices.

4.3.2 Model enhancements

Since the model will be applied to a highly urbanized downtown sewershed, more details such as different portions of impervious pavements are needed to be represented in the model. Four enhancements of the L-THIA-LID 2.1 model were added in this study, including (1) separate representation of sidewalks, roads, driveways, and parking lots to allow more detailed representation of impervious surfaces; (2) separate cost calculation of applying GI practices on sidewalks, roads, driveways, and parking lots; (3) representation of actual suitable areas for applying bioretention in each HRU (the current model used the assumption that bioretention treats 15% of the remaining runoff after being treated by green roofs, rain barrels/cisterns, porous pavements, and permeable patios, Liu (2015)); and, (4) improvement of model efficiency by modifying the Python code to reduce redundancy. The enhanced version was named the L-THIA-LID 2.2 model, which includes more GI details for simulations at watershed scales than the previous version.

4.4 Methods and Materials

4.4.1 Pilot test of the L-THIA-LID 2.2 model

A pilot test was conducted for porous sidewalks and porous parking lots in a single HRU in a densely developed sewershed (Fulton sewershed) in the City of Peoria. The purpose of this pilot study was to test the L-THIA-LID 2.2 model before applying it to much larger areas. Land use in this sewershed is solely commercial and the land cover consists of building roofs (50%), roads (15%), sidewalks (7%), parking lots (19%), and vegetation (9%). The HSG for the sewershed is "D" (Soil Survey Geographic (SSURGO) database). Fifty years of observed daily rainfall (1966 to 2015) at the Peoria International Airport was downloaded from the National Oceanic and Atmospheric Administration National Climatic Data Center (https://www.ncdc.noaa.gov/cdo-web/datasets). The L-THIA-LID 2.2 model was set up based on the National Land Cover Database (NLCD) 2011 and four scenarios were developed based on applying porous pavements to sidewalks, parking lots, and streets, including the baseline scenario (current condition, no GI), scenario 1 (porous pavements implemented in all sidewalks areas, 7% of the total area), scenario 2 (porous pavements implemented in all parking lots, 19% of the total area), and scenario 3 (porous pavements implemented on all sidewalks and 2m width in parking lanes on both sides of street, 10% of the total area).

Based on these simulations, incorporating porous pavements could achieve average annual runoff reduction by 7%-21% and TSS reduction by 9%-24% in the Fulton Sewershed. Porous pavement implementation on all current parking lot spaces in the Fulton street sewershed would result in the highest reduction in runoff volume and TSS. These results indicate that GI implementation in urban communities can provide sustainable benefits. This test shows that the L-THIA-LID 2.2

model is able to simulate GI practices on sidewalks, roads, driveways, and parking lots, and the methodologies developed were appropriate and could be applied to the larger area.

4.4.2 Study area: Darst Sewershed

The City of Peoria has made a series of improvements to deal with CSO over time, but so far there have been no improvements to reduce overflows to the Illinois River downstream of Peoria Lake (southwest of downtown) (PCLTCP, 2010). From 2001 to 2011, urbanization contributed 826 hectares of additional developed land in the City of Peoria, among which 160 hectares of developed land have been converted to higher intensity uses (Appendix D). The L-THIA simulated average annual runoff volume increased about 437,850 m^3 and the simulated total suspended solids (TSS) load increased about 27.2 tons from 2001 to 2011 due to urbanization (Appendix D).

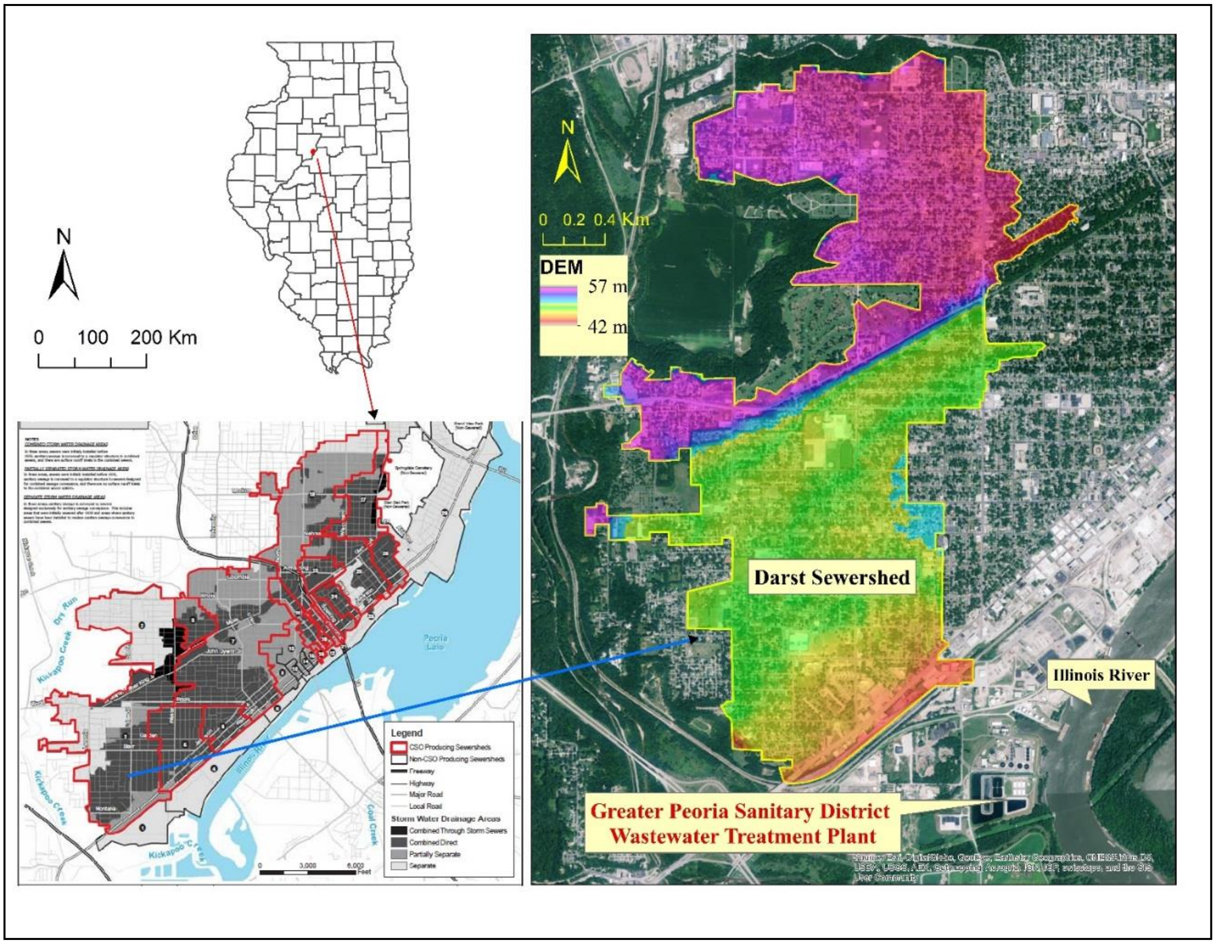

Figure 4.2 CSO locations (marked in red in lower left subfigure) in the City of Peoria, IL and Darst Sewershed location and Digital Elevation Model (DEM). There is a 15 m decrease in elevation in the direction of the Illinois River at the ridge along Martin Luther King Jr. Drive (blue line shown in right sub figure in figure 4.2)

Darst Sewershed is a highly urbanized community, and the predominant landuse types are residential land, commercial land, and some industrial land in the south. It has a total of 638 ha (6.4 km^2) in areas with 41.5% impervious surfaces (Table 4.2). Buildings are the largest source of impervious surfaces (14.4% of the sewershed) followed by roads (13.3%). Parking lots (6.0%) and driveways (5.5%) contribute similar portions to impervious surfaces. Sidewalks account for 2.3% of the total area. GI infiltration and storage practices are primary potential methods to mitigate the

CSO impacts on hydrology and water quality. Based on impervious surfaces statistics (Table 4.2), Darst Sewershed has extensive areas available for GI applications. Those impervious surfaces could potentially provide opportunities for GI practices implementation.

Table 4.2 Impervious surfaces statistics for the Darst Sewershed

	Total Area of Darst Sewershed	Sidewalks	Driveways	Buildings	Roads	Parking lots	Total Impervious
Area (ha)	638	15	35	92	85	38	265
Percent (%)	100	2.3	5.5	14.4	13.3	6.0	41.5

4.4.3 Data requirements and pre-processing

This study used land use data based on NLCD 2011, HSG data extracted from the SSURGO database, and 50 years of observed daily precipitation from 1966 to 2015 at the Peoria International Airport that was downloaded from the National Oceanic and Atmospheric Administration National Climatic Data Center (https://www.ncdc.noaa.gov/cdo-web/datasets). The NLCD 2011 land use data for the Darst Sewershed were pre-processed using ArcGIS and reclassification rules in Liu et al. (2015b) as follows: (1) developed open space and developed low intensity areas to low density residential (LDR); (2) medium and high intensity developed lands including high density residential (HDR), industrial and commercial areas to high intensity; (3) industrial and commercial areas were digitized from aerial images and separated from high intensity; and, (4) evergreen forest, deciduous forest, mixed forest, and shrub/scrub to forest/woods. Other reclassification rules related to other land use types in NLCD are provided in Liu et al. (2015b).

GIS data provided by the Peoria County GIS Data (Peoria County IT Services-GIS Division) included boundaries (CSO Sewershed outlines, city boundary), building footprints, roads, driveways, sidewalks, parking lots, streams, and lakes. The building footprints layer defined the roof top areas. NLCD 2011 Percent Developed Imperviousness was downloaded from the Multi-Resolution Land Characteristics Consortium (https://www.mrlc.gov/nlcd11_data.php) and used to provide information the distribution of impervious areas in the study area. The digital elevation model (DEM) data (3 meter) were downloaded from The National Map (http://nationalmap.gov/) forextracting slope and drainage areas. Bioretention in the Darst Sewershed was assumed to be possible in a 2.13-meter buffer at the outside of each road (both sides), but excluding overlap with sidewalks, driveways, parking lots, and buildings.

Eight GI practices that are less space intensive were identified as suitable in the Darst Sewershed, including rain barrels/cisterns, green roofs, rain barrels/cisterns plus green roofs, bioretention, porous roads, porous parking lots, porous sidewalks, and porous driveways. After data pre-processing, criteria for suitable locations for different GI practices, as shown in Table 4.1, were used to create unique categories that represented combinations of land use, hydrological soil group, and areas suitable for specific GI practices (HRU plus GI practice suitable locations).

4.4.4 Evaluation of GI practices performance on hydrology and water quality based on the current condition

The long-term effects of GI practices in Darst Sewershed "what if" scenarios were evaluated without calibration of the model due to limited surface runoff observational data. Since model inputs were based on a detailed inventory of soils and land use that represented the actual watershed conditions very closely, detailed model calibration is not essential (Chaubey et al., 2010). The SCS CN method (NRCS, 1986) is a well-recognized and extensively applied runoff estimation approach and its usefulness and credibility have been demonstrated widely (e.g.

Ahiablame et al., 2013; Bhaduri et al., 2001; Grove et al., 2001; Kim et al., 2002; Wright et al., 2016; You et al., 2012). In addition, the focus of this impact study is on differences resulting from implementation of GI practices (rather than absolute values), and thus any error in the base model is likely to also occur in the model with GI and thus may largely cancel out when differences are computed. In previous work, calibration produced relatively small changes in parameters. For example, Liu et al. (2015b) calibrated the L-THIA-LID 2.1 model using streamflow data from 1993 to 2001 in a highly urbanized watershed near Indianapolis and found that decreasing the curve number by 1% achieved good model performance when validated using streamflow data from 2002 to 2010. Therefore, we assert that the uncalibrated L-THIA-LID 2.2 model is sufficient to provide results that can guide decision makers in understanding the impacts associated with implementing GI practices over the long-term.

Previous work has shown that combined implementation of BMPs/LID practices perform better than applying each alone (Ahiablame et al., 2012; Damodaram et al., 2010). Ten scenarios that implement different hypothetical levels and combinations of 8 GI practices were simulated (Table 4.3) compared to current conditions (baseline scenario, S0) in which no GI practices have been implemented. The first scenario represented the City proposed plan 1 (PDGIP, 2015), which involves applying up to 50% of GI practices on both eligible parking lanes of the roads. In the L-THIA-LID 2.2 modeling, S1 means that 50% of the eligible parking lanes (located in the suitable locations) of the roads were converted to porous roads (porous pavement). S1_p represented the City proposed plan 2, which added and converted 17% of eligible parking lots to porous parking lots based on S1. The proportion (17%) represents the extent of parking lots controlled by the City. Scenarios 2 through 10 placed different percentages and combinations of GI practices within the study area as follows: 1) for the L-THIA-LID 2.2 modeling, rain barrels/cisterns were assumed to be applied to all roof tops of residential, industrial, and commercial areas; 2) green roofs were assumed to only be applicable in industrial and commercial areas. Green roofs and rain barrels/cisterns could be combined in series and applied in industrial and commercial areas. For scenarios 8 and 9, 50% of rain barrels/cisterns implementation represented that 50% of the eligible roof tops in the residential areas were connected to rain barrels/cisterns. Scenario 10 was the maximum implementation of GI practices in the suitable locations.

Four major environmental concerns were simulated, including average annual runoff volume, average annual TSS, average annual TN, and average annual TP. For the baseline scenario (before implementing GI practices), the runoff volume and pollutant loads were estimated based on the Curve Number method and Event Mean Concentration (EMC) method. The Curve Number values of GI practices were based on Ahiablame et al. (2012), and the EMC values used in the L-THIA-LID 2.2 model for different land use were taken from Liu et al. (2015a).

Table 4.3 Scenarios developed for different GI practices

Scenario	Rain barrels/ cisterns (%)	Green roofs (%)	Green roofs +Rain barrels/cisterns (%)	Bioretention (%)	Porous roads (%)	Porous parking lots (%)	Porous sidewalks (%)	Porous driveways (%)	Details
S1	0	0	0	0	50	0	0	0	City proposed plan 1 (applying 7-feet/2.13m wide street GI on both parking lanes of the roads)
S1_p	0	0	0	0	50	17	0	0	City proposed plan 2 (City owned up to 17% of parking lots)
S2	0	0	0	0	50	50	50	0	Add 50% more porous pavements on parking lots and sidewalks compared to S1_p
S3	0	0	0	0	100	50	50	0	Add 50% more porous pavements on eligible parking lanes of the roads compared to S2
S4	0	0	0	50	0	0	0	0	Only bioretention
S5	0	0	0	50	50	0	0	0	Add 50% porous roads compared to S4
S6	0	0	0	100	100	50	50	0	Add 100% bioretention compared to S3
S7	0	0	0	100	100	50	50	50	Add 50% porous driveways compared to S6
S8	50	0	0	100	100	50	50	50	Add 50% rain barrels/cisterns to suitable area compared to S7
S9	50	50	0	100	100	50	50	50	Add 50% green roofs to suitable area compared to S8
S10	100	0	100	100	100	100	100	100	Add GI practices to all suitable areas

4.4.5 Calculation of total cost and cost effectiveness of implementing GI practices

The total cost and cost effectiveness of implementing GI practices were estimated for each scenario using equations presented in section 4.3.1. Table 4.4 shows the construction costs and annual maintenance costs of GI practices. A design life of 20 years was used in the calculations consistent with Strassler et al. (1999), King and Hagan (2011), and Liu et al. (2015b), and the interest rate value used was 4.5% (Liu et al., 2015b). Cost effectiveness was defined as the amount (cost) it would take to achieve a unit reduction of a given component in a year. The baseline/current status is assumed to be no GI practices and that all costs pertain to GI only (not any other practices), and thus the baseline/current cost is assumed to be zero relative to GI implementation.

Table 4.4 Cost of GI practices: Construction and annual maintenance costs based on Liu et al. (2015b)

GI practices	Construction cost ($/m^2 drainage area)	Annual maintenance cost (% of construction cost)	Sources
Rain barrels/cisterns	6.71	1	Adopted from Liu et al. (2015b)
Green roofs	168.34	6	
Bioretention	15.12	6	
Porous roads/ parking lots/ sidewalks/ driveways	59.20	1	

4.5 Results and Discussion

4.5.1 Suitable locations for implementing GI practices in the Darst Sewershed

Based on the site suitability criteria (Table 4.1), suitable locations for GI practices in the Darst Sewershed are as shown in Figure 4.3. Areas that were not suitable for GI practices covered 437 ha (light yellow in Figure 4.3), which accounted for 69% of the total study area. The rain barrels/cisterns could be applied to eligible roof tops in the residential areas covering a total area of 75 ha, as shown in dark green (Figure 4.3). These accounted for 12% of the total area in the Darst Sewershed. Green roofs plus rain barrels/cisterns could be applied to 17 ha of eligible roof tops in the industrial and commercial areas (light green in Figure 4.3). The suitable area to apply bioretention totaled 23 ha (3.6% of the total area). The suitable areas to apply porous pavements to eligible driveways, roads, parking lots, and sidewalks were 26 ha (4%), 27 ha (4.3%), 24 ha (3.8%), and 9 ha (1.4%), respectively (Table 4.5), which indicated that the City proposed plans 1 (50% of the porous roads applying on the eligible parking lanes) and 2 (adding 17% of eligible parking lots based on City proposed plan 1) did not have extensive areas for GI implementations.

Table 4.5 Summary of suitable areas and percentage of the total study areas of different GI practices

	Suitable area (ha)	Percentage to total study area (%)
Rain barrels/cisterns (residential)	75	11.8
Green roofs +rain barrels/cisterns (industrial and commercial)	17	2.6
Bioretention	23	3.6
Porous driveways	26	4.0
Porous roads	27	4.3
Porous parking lots	24	3.8
Porous sidewalks	9	1.4

The total imperviousness in the Darst Sewershed is 41.5% (Table 4.2). Based on work by Hamdi et al. (2011) focused on surface runoff responses to historical urbanization (1960 -1999) in the Brussels Capital Region, changes became detectable in high flows, the frequency of flood events, and the annual series of cumulative surface runoff when impervious areas exceeded 35%. Tang et al. (2005) and Ahiablame et al. (2012) also observed that increases in runoff were due to increases in impervious surfaces. Those results implied the need of GI practices to mitigate potential increases in runoff due to urbanization. In the Darst Sewershed, by applying different levels and combinations of GI practices, the impervious surface area in the watershed could potentially be decreased to 27.9%.

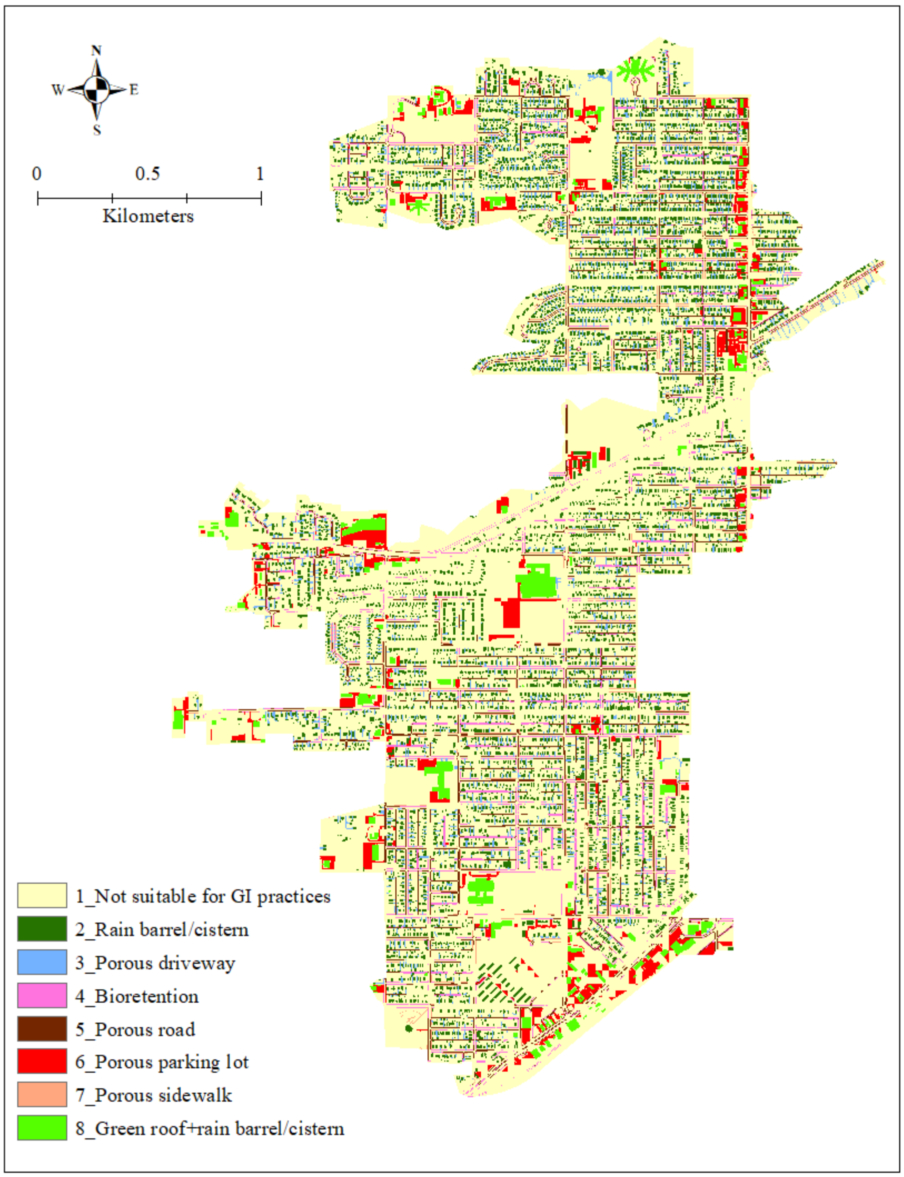

Figure 4.3 Spatial distribution of the suitable locations of GI practices in the Darst Sewershed

4.5.2 Evaluation of the effectiveness of GI practices

The effectiveness of GI practices was evaluated using ten scenarios. The simulated annual runoff and pollutant loads for current condition (baseline scenario, S0) in the Darst Sewershed are detailed in Table C1. The simulated annual runoff ranged from 1079 to 4380 m^3/ha/yr for the study area with an average value of 2427 m^3/ha (about 1.6 million cubic meters in total in the Darst Sewershed, Table 4.6). The simulated annual runoff was consistent with the values reported by Ahiablame et al. (2013) (1303 - 3896 m^3/ha/yr) for two urbanized watersheds near Indianapolis, with similar climatic conditions to those of our study area. Annual TSS loads fluctuated from 56.7 to 229.8 kg/ha (Table C1) with an average value of 127.5 kg/ha (81.3 tons in total, Table 4.8). Annual nutrient loads varied between 2.0 and 8.2 kg/ha for TN, and between 0.8 and 3.3 kg/ha for TP (Table C1). The average annual TN and TP loads were 4.6 kg/ha (2.9 tons in total, Table 4.8) and 1.8 kg/ha (1.2 ton in total, Table 4.8), respectively.

Table 4.6 Average annual runoff and pollutant loadings of the current (baseline) conditions.

	Average Annual Runoff (m^3)	Average Annual Total Suspended Solids (tons)	Average Annual Total Nitrogen (tons)	Average Annual Total Phosphorus (tons)
S0	1,548,992	81.3	2.9	1.2

Figure 4.4 shows the GI/GI combination performance at different implementation levels, with performance being represented based on runoff volume and pollutant load reductions (%) associated with the different implementation scenarios relative to the baseline scenario (S0) of the Darst Sewershed.

The adoption of 50% porous roads applied to eligible parking lanes (S1, Table 4.3) could achieve average annual reductions of runoff volume by 0.5%, TSS by 2.6%, TN by 2.2%, and TP by 2.5%.

Only 2.2% of the total area was covered by porous roads (Table 4.5), hence the small reductions in runoff volume and pollutants associated with S1. The adoption of 50% porous roads and 17% of porous parking lots (S1_p, Table 4.3) resulted in average annual reductions of 1.3%, 3.6%, 3.9%, and 3.2% in runoff volume, TSS, TN, and TP, respectively. These reductions are slightly higher than those in S1. However, only 2.8% of the total area in the Darst Sewershed was covered by porous roads and porous parking lots in this scenario.

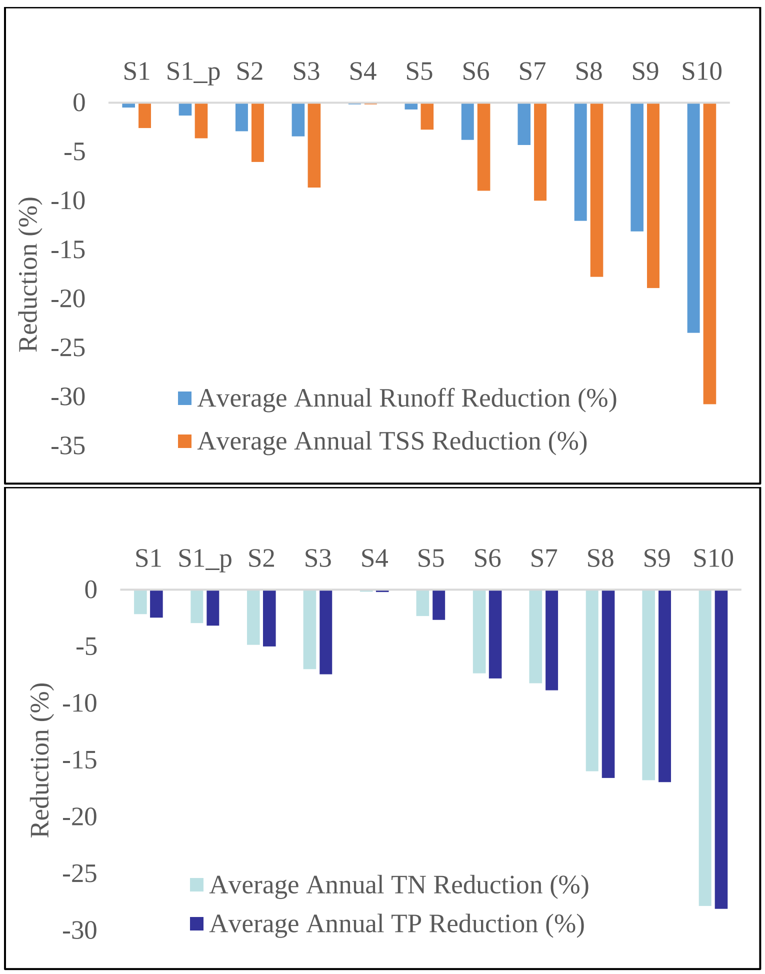

Figure 4.4 Runoff and pollutant loads percent reduction after scenario simulations of different implementation levels and combinations of GI practices compared to baseline scenario

Scenario 2 included more porous pavement on parking lots (50%) and sidewalks (50%) compared to S1_p. This resulted in average runoff volume and pollutant load reductions ranging from 2.9% to 6.1%. Scenario 4 only implemented 50% bioretention to the eligible areas; runoff volume and pollutant reductions for this scenario ranged from 0.18% to 0.2%. Reduction values for this scenario were low, possibly attributable to: 1) suitable locations for bioretention were occupied mainly by grass and the differences of CN numbers used in the L-THIA-LID 2.2 model for grass and bioretention were relatively small; and, 2) no routing method is incorporated in the L-THIA-LID 2.2 model, thus, evaluation of bioretention was confined by limited suitable areas. Compared to S1, scenario 5 added 50% porous roads. However, runoff and pollutant load reductions in S5 differed only slightly from those of S1, ranging from 0.7% to 2.8%. Each percent reduction in average annual runoff volume and pollutant loads in S5 was equivalent to the total of the corresponding reductions in S1 and S4, attributable to the assumption of combined implementation of bioretention and porous pavements. Rain barrels/cisterns could be connected to all roof tops in the commercial, industrial, and residential areas in the L-THIA-LID 2.2 model. Scenario 8 had reductions of 12.1% to 17.8% for average annual runoff volume and pollutant loads, respectively. This scenario added 50% rain barrels/cisterns applied on the residential areas to S7. Scenario 10 added all GI practices applied to all suitable areas in the Darst Sewershed. The higher reductions resulting from this scenario were not surprising given that combinations of all possible GI practices would result in more runoff storage, additional infiltration and evaporation opportunities, and/or more treatment, which could lead to greater runoff and pollutant loads reduction.

Overall, depending on the combinations of GI practices as well as different adoption levels, 0.2% -23.5% of average annual reductions in runoff volume, 0.2% -30.8% of average annual reductions in TSS, 0.2% -27.9% of average annual reductions in TN, and 0.2% -28.1% of average annual reductions in TP could be achieved. Based on the site characteristics, adding more GI practices did not necessarily mean achieving greater runoff and pollutant load reductions, such as S1 vs. S5. Combined implementation of GI practices was found more effective on reducing runoff and pollutant loads than applying individual practices alone.

4.5.3 Cost analysis of implementing GI practices

Tables 4.7 and 4.8 show the annual costs GI implementation and the associated cost effectiveness (cost per unit component reduction) considering a design life of 20 years. The annual cost of S10 was the most among all scenarios because S10 was the maximum implementation of GI practices in the Darst Sewershed. Although rain barrels/cisterns adopted for roof tops in scenario 8 treated a much larger area than did green roofs in the 9th scenario, S9 had a higher annual cost than S8 as green roofs were costlier to implement. This indicated that more investments did not necessarily mean significant additional reductions. The annual cost of S5 was equal to the sum of the annual cost of S1 and S4. Cost effectiveness values were compared to evaluate the cost implications of the different scenarios. The values of cost effectiveness for runoff were consistent with values reported by Liu et al. (2015b) ranging from \$4/m^3/yr to \$26/m^3/yr.

Table 4.7 Annual cost, reductions in runoff and sediment, and cost effectiveness for each scenario after implementing GI practices for different levels and combinations

Scenario	Annual Cost ($)	Average Annual Runoff Reduction (%)	Cost Effectiveness for Average Annual Runoff Reduction ($/m^3/yr)	Average Annual TSS Reduction (%)	Cost Effectiveness for Average Annual TSS Reduction ($/kg/yr)
s1	229,994	0.5	29.84	2.6	109.45
s1_p	427,192	1.3	21.02	3.6	145.25
s2	838,649	2.9	18.50	6.1	170.23
s3	1,068,643	3.4	20.15	8.6	152.06
s4	35,339	0.2	12.66	0.2	243.03
s5	265,333	0.7	25.27	2.8	118.09
s6	1,139,320	3.8	19.44	9.0	155.67
s7	1,292,094	4.3	19.39	10.0	158.71
s8	1,721,095	12.1	9.21	17.8	119.19
s9	3,973,158	13.1	19.53	18.9	258.26
s10	7,415,566	23.5	20.38	30.8	296.30

Table 4.8 Annual cost, reductions in TN and TP, and cost effectiveness for each scenario after implementing GI practices at different levels and combinations

Scenario	Annual Cost ($)	Average Annual TN Reduction (%)	Cost Effectiveness for Average Annual TN Reduction ($/m^3/yr)	Average Annual TP Reduction (%)	Cost Effectiveness for Average Annual TP Reduction ($/kg/yr)
s1	229,994	2.2	3,683	2.5	8,039
s1_p	427,192	3.0	4,994	3.2	11,543
s2	838,649	4.9	5,949	5.0	14,408
s3	1,068,643	7.0	5,253	7.4	12,309
s4	35,339	0.2	6,487	0.2	15,422
s5	265,333	2.3	3,908	2.7	8,587
s6	1,139,320	7.4	5,316	7.8	12,465
s7	1,292,094	8.3	5,386	8.9	12,496
s8	1,721,095	16.0	3,701	16.6	8,901
s9	3,973,158	16.8	8,139	17.0	20,079
s10	7,415,566	27.9	9,155	28.1	22,609

Scenario 8, which implemented 50% rain barrels/cisterns, 100% bioretention and porous roads, 50% porous parking lots, 50% porous sidewalks, and 50% driveways, was most cost-effective in runoff reduction ($9/m^3/yr, Table 4.7), which indicated that it would take $9 to achieve 1 m^3 reduction of runoff in a year. S1 (50% porous roads) had smaller cost-effective values in TSS reductions ($109/kg/yr, Table 4.7), TN reductions ($3683/kg/yr, Table 4.8), and TP reductions ($8039/kg/yr, Table 4.8). However, S8 had very similar values for cost effectiveness with S1 but S8 was much more cost-efficient in runoff reduction. Therefore, S8 was the most cost-effective scenario. S10 achieved the greatest reductions in runoff volume and pollutant loads (Tables 4.7 and 4.8), however, it was not cost-effective. Scenarios that had similar cost-effectiveness values did not necessarily achieve similar reductions. For example, S6, S7, and S9 had similar cost effectiveness values for runoff reduction, but S9 could achieve more runoff reduction (13%) than that of S6 (3.8%) and S7 (4.3%) (Table 4.7); S1, S5, and S8 had similar cost effectiveness values for TN and TP reductions but S8 could achieve more TN and TP reductions (16% for TN and 16.6% for TP) than that of S1 (2.2% for TN and 2.5% for TP) and S5 (2.3% for TN and 2.7% for TP) (Table 4.8).

4.5.4 Discussion

USEPA and the City of Peoria have agreed that overflows will be considered eliminated in the Darst Sewershed, if, once the improvements are constructed, there is no discharge through the combined sewer outfalls for the 10-year storm (the 10-year storm shall mean the modeled rainfall event recorded at the Peoria airport from 01:00 on July 17, 1966 to 16:00 on July 28. 1966, 12 days) ((PDGIP, 2015). The 10-year storm overflow volumes recorded in the PDGIP (2015) for the Darst Sewershed were 25,362 m^3. According to the L-THIA-LID 2.2 model simulations, the runoff could be reduced by 59% compared to the targeted overflow reduction volumes if the most cost-effective scenario, scenario 8, (adoption of 50% rain barrels/ cisterns, 100% bioretention, 100% porous pavements on eligible parking lanes of the roads, 50% porous parking lots, 50% porous sidewalks, and 50% porous driveways, Table 4.3) was adopted. Model results showed that it would cost about $34.4 million for applying scenario 8 in 20 years (Table 4.7). For this to work in practice, the City of Peoria needs to persuade residents and related stakeholders to consent with GI practice adoption. If the maximum application of GI practices (S10, Table 4.3) was adopted, model simulation showed that the Darst Sewershed could achieve 100% percent of the targeted overflow reduction volumes during the proposed 10-year storm. However, scenario 10 implementation would cost about 148.3 million dollars for a 20-year duration, and it is not cost effective (Table 4.7). This scenario needs to use every suitable space in the Darst Sewershed for GI implementation, which is not practical and would be extremely hard for the City of Peoria to implement. The two scenarios proposed by the City of Peoria (S1 and S1_p, Table 4.3) could only reduce the targeted overflow volumes by 1.3% and 2%, respectively, based on model simulations, which is due to limited space (4% of total study area) for porous pavement implementation in the Darst Sewershed.

The bioretention system simulated in this study only achieved 0.2% average annual runoff reduction. The reasons could be attributed to limited spaces in the Darst Sewershed for bioretention system implementation and no routing process being represented in the model. Pervious pavements were found to be desirable to imitate the pre-development circumstances or retrofit existing residential dwellings and were able to reduce various pollutants, such as TSS, metals, and nutrients (Brattebo and Booth, 2003; Collins et al., 2008; Collins et al., 2009; Dietz, 2007, Fassman and Blackbourn, 2010; Pagotto et al., 2000; Zachary-Bean et al., 2007). However, there are also drawbacks with porous pavers, such as subsidence and pore blockage (Haselbach et al., 2006). Grass and weeds could grow within the pavement, and if the homeowner chooses to spray, it would potentially add to pollutant loads. Mosquito control issues come along with rain barrels implementation in some areas, such as Florida. Although GI practices have many advantages, other factors need to be considered when developing implementation plans by balancing pros and cons for specific sites.

Successful implementation of GI practices needs cooperation from the residents when considering private properties as potential locations for GI implementation. The City of Peoria might consider incentives such as reducing residential stormwater fees where owners agree to implement GI practices. Other changes in infrastructure in the City of Peoria could also be helpful in mitigating CSO problems. For example, building large underground storage tank systems to control the timing and volume of stormwater discharge, such as in New York City (Kurtz et al., 2000; Protopapas, 1999), in Germany (Brombach, 2002), and in England (Yuan et al., 1999). Combined implementation of GI practices and construction of additional stormwater infrastructure could achieve runoff and pollutants reductions by increasing infiltration and providing storage for urban runoff. Other benefits include improving water quality of streams (potentially reducing health risks for residents), improving groundwater recharge (potentially slowing or reversing land subsidence), increasing property values by enhancing neighborhood aesthetics, mitigating the urban heat island effect by minimizing impervious surfaces and infiltrating water running off hot pavements and shading, and providing ecosystem services benefits such as restoring aquatic habitat.

Runoff and pollutant loads from each HRU were added up to obtain the values at the outlet of the watershed in the L-THIA-LID 2.2 model. The model does not simulate routing processes in detail. Future study could be focused on representing routing in the model. When running the model, GI practices were assumed to be fully functional in each daily simulation. Full public cooperation was also assumed when calculating the total cost and cost effectiveness. The benefits of implementing GI practices in this study only considered reductions in runoff volume, TSS, TN, and TP. Other benefits could also be quantified in future studies, for instance, the benefits of rain barrels/cisterns in collecting water for irrigation; the benefits of GI practices in increasing infiltration and reducing peak runoff and other pollutant loads; and, the benefits of GI practices, such as bioretention, in enhancing site aesthetics and providing wildlife habitat.

The scenarios developed in this study reflect the effectiveness of GI practices fairly well. Combining the L-THIA-LID 2.2 model with an optimization algorithm could better answer the following questions: (1) within a given budget, how much more runoff and pollutant reductions are obtainable? (2) to achieve certain reduction goals, how much less would green infrastructure cost? and, (3) where are the best locations to place given GI practices so as to achieve the highest pollutant reductions at the least cost? Such assessments are, however, beyond the scope of this current study.

4.6 Conclusions

This work enhanced the L-THIA-LID 2.1 model to: simulate impervious surfaces including sidewalks, roads, driveways, and parking lots in greater detail; conduct cost calculations for these more detailed impervious surfaces; and, consider the actual suitable areas for bioretention in the model. The effectiveness of eight GI practices - including rain barrels/cisterns, bioretention,green roofs, green roofs plus rain barrels/cisterns, porous roads, porous parking lots, porous sidewalks, and porous driveways - on hydrology and water quality at a sewershed level was evaluated using the enhanced (L-THIA-LID 2.2) model (enhanced version) based on 10 scenarios representing different levels and combinations of GI practice implementation. The total cost and the cost effectiveness for each scenario considering a 20-year lifetime for GI practices were also examined.

The study leads to the following conclusions: 1) simulation results showed that combined implementation of GI practices performed better than applying individual practices alone. The various adoption levels and combinations of GI practices could potentially reduce runoff volume by 0.2-23.5%, TSS by 0.2-30.8%, TN by 0.2-27.9%, and TP by 0.2 to 28.1%; 2) adding more GI practices did not necessarily mean that substantial runoff and pollutant reduction would be achieved based on site characteristics; 3) the most cost-effective scenario (S8, adoption of 50% rain barrels/cisterns, 100% bioretention, 100% porous pavements on eligible parking lanes of the roads, 50% porous parking lots, 50% porous sidewalks, and 50% porous driveways) has an associated cost of $9.21 to achieve 1 m^3 runoff reduction per year and $119 to achieve 1 kg TSS reduction per year. This, however, assumes cooperation from the general public in implementing GI practices on their properties; 4) adoption of GI practices on all possible areas could potentially achieve the greatest runoff and pollutant loads reductions but would not be the most cost-effective option.

Results from this study provide potential impacts of GI practices implementation on hydrology and water quality in the Darst Sewershed. The study serves as a preliminary analysis at the planning stage. The enhanced model from this study, L-THIA-LID 2.2, showed the ability to simulate the effectiveness of GI practices as well as calculate the total cost and cost effectiveness of implementing GI practices. The methodologies developed were appropriate and could be applied to other locations to support assessing the beneficial uses of GI practices. This model and its user's manual are available upon request.

4.7 Acknowledgements

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CHAPTER 5. SUMMARY, CONCLUSIONS, AND RECOMMENDATIONS

5.1 Research Overview

This study was conducted to assess urbanization impacts on surface runoff from national and regional scopes and then to evaluate the effects of green infrastructure on hydrology and water quality as mitigation approaches at local scales. The specific objectives of this study were to:

- ❖ Assess urbanization impacts on surface runoff of the contiguous United States using the L-THIA Tabular Tool based on the 2001, 2006, and 2011 National Land Cover Databases.
- ❖ Assess simulated precipitation from updated CLIGEN database impacts on the surface runoff using the L-THIA Tabular Tool in the five states of the Great Lakes Region (WI, IL, IN, MI, and OH).
- ❖ Evaluate the effectiveness of green infrastructure practices as sustainable development or re-development tools on hydrology and water quality in a CSO community.

In the first objective, a newly developed version of the Long-Term Hydrologic Impact Assessment (L-THIA) model, the L-THIA Tabular Tool, was used to assess urbanization impacts on average annual runoff of the contiguous U.S. based on the National Land Cover Databases for 2001, 2006 and 2011. The normalized average annual runoff depths (NAARD) for 2001, 2006, and 2011 by contiguous U.S. counties and states were assessed and whether population change is a consistent indicator for urban development was evaluated. This study also assessed the national average annual runoff volume increase due to urbanization.

In the second objective, the performance of CLIGEN using the updated 2015 database in generating daily, monthly, and yearly precipitation amounts, and how well these matched the statistical parameters of the historical records were evaluated. The impacts of CLIGEN applications with the updated database on daily precipitation estimates for urban runoff simulations using the L-THIA Tabular Tool were quantified. In this study, the comparison of observed average annual precipitation between the 1995 and updated 2015 CLIGEN databases across the contiguous United States was first conducted. The analysis was then narrowed to the five states of the U.S. Great Lakes Region: Wisconsin (WI), Illinois (IL), Indiana (IN), Michigan (MI), and Ohio (OH). Statistical comparisons between the observed precipitation records and the CLIGEN precipitation estimates at eight weather stations in the region were presented. Finally, the differences in L-THIA-predicted normalized average annual runoff depths (NAARD) from urban land based on NLCD 2001, 2006, and 2011 in the U.S. Great Lakes Region by using CLIGEN precipitation estimates created with the 1995 and updated 2015 databases were compared.

In the third objective, a watershed scale rainfall-runoff model with management practices simulation (the Long Term Hydrologic Impact Analysis - Low Impact Development model, the L-THIA-LID 2.1 model) was enhanced. The enhanced L-THIA-LID 2.2 model was able to simulate more detailed impervious surfaces including sidewalks, roads, driveways, and parking lots, to conduct cost calculations for converting these impervious surfaces to porous pavements, and to consider the actual suitable locations for bioretention in the study area. The effectiveness of GI practices on improving hydrology and water quality for a combined sewer overflow urban watershed - the Darst Sewershed of the City of Peoria, IL - was examined in 10 simulation scenarios using 8 practices including rain barrels/cisterns, green roofs, green roofs plus rain barrels/cisterns, bioretention, porous roads, porous parking lots, porous sidewalks, and porous driveways. The total cost and the cost effectiveness for each scenario considering a 20-year GI practices lifetime were examined as well.

5.2 Major Research Findings

Major findings of this research are:

- The majority of counties in the contiguous Unites States were low runoff counties during the period 2001 to 2011 and had long-term normalized average annual runoff depth from urban land less than 17.8 mm. However, the spread of urban sprawl to suburban areas around metropolitan areas as well as newly urbanized areas within metropolitan areas resulted in more medium and very high runoff counties. For the top ten NAARD states in 2011, NAARD values were jointly influenced by high precipitation and increasing urban land, while the top ten NAARD change percentage states in 2011 were mainly in the western U.S. in areas with low precipitation, and their NAARD values were mainly influenced by high increases in urban land. Nationally, about 3.3 billion cubic meters of average annual runoff were gained due to urbanization from 2001 to 2011 based on the model simulation results. Population increases are a factor in urban development, however population is not a good predictor of urbanization levels because some areas have undergone decreasing population but increasing urban land area. Therefore, population change alone is not a sufficient proxy with which to analyze the increase in urban development. The L-THIA Tabular Tool was demonstrated to be capable of generating useful information about urbanization impacts on surface runoff using the stochastic weather generator, CLIGEN, together with the NLCD datasets and STATSGO soil dataset.

- The national comparison of observed average annual precipitation between weather stations in the 1995 and updated 2015 CLIGEN databases indicated that differences existed in average annual precipitation, possibly due to different observation period lengths and consistency of historical records. Statistical analysis showed that CLIGEN precipitation estimates with the updated database could adequately replicate daily precipitation up to the 99.5th percentile compared to observed data in the five states (WI, IL, IN, MI, and OH) of the U.S. Great Lakes Region. However, maximum precipitation at the daily, monthly, and annual scale tended to be largely underestimated. Large precipitation variations over time were found to decrease reliability of CLIGEN-generated values. CLIGEN parameterization could be based on any desired historical period to improve precipitation estimates in relation to project objectives. However, exclusion of recent data in CLIGEN parameterization while using generated values to represent present or future climate conditions could be problematic. CLIGEN based on the updated database was demonstrated to be suitable for simulating precipitation in the five states examined (WI, IL, IN, MI, and OH) of the U.S. Great Lakes Region. CLIGEN with the updated database was also found preferable with respect to modeling urban runoff or urbanization effects since it includes up-to-date historical and temporally-consistent measurements. The assessment of urban runoff based on simulated precipitation from CLIGEN with the updated database can provide insights for public and planning agencies to assist sustainable development. When the only change in input of the L-THIA model is climate parameters used by CLIGEN, variations in simulation results reflect the impacts of precipitation characteristics on runoff from urban areas.

- Combined implementation of GI practices performed better than applying individual practices. The various adoption levels and combinations of GI practices could potentially reduce runoff volume by 0.2-23.5%, TSS by 0.18-30.8%, TN by 0.2-27.9%, and TP by 0.20 to 28.1%. Adding more GI practices did not necessarily mean that substantial runoff and pollutant reduction would be achieved based on site characteristics. The most cost-effective scenario (S8, adoption of 50% rain barrels/cisterns, 100% bioretention, 100% porous pavements on eligible parking lanes of the roads, 50% porous parking lots, 50% porous sidewalks, and 50% porous driveways) had an associated cost of $9.21 to achieve 1 m^3 runoff reduction per year and $119 to achieve 1 kg TSS reduction per year. This, however, assumes cooperation from the general public in implementing GI practices on their properties. Adoption of GI practices on all possible areas could potentially achieve the greatest runoff and pollutant load reductions but would not be the most cost-effective option. Results from this study provide theoretical impacts of GI practices implementation on hydrology and water quality in the Darst Sewershed. The study serves as a preliminary analysis at the planning stage evaluating effectiveness of GI practices. The methodology and the enhanced model from this research can be applied to other locations to support assessing the beneficial uses of GI practices. The work on evaluation of the effectiveness of GI practices was conducted in collaboration with the City of Peoria Public Works Department and the Peoria County IT Services-GIS Division. Preliminary results were presented to community stakeholders (Peoria One Water Committee working group of 15 - 20 stakeholders) through a Peoria Tipping Points meeting for education purpose. Community stakeholders include the Chamber of Commerce, environmental advocacy groups, park district, health department, school districts, government agencies, the business community, and citizens.

An assessment of urbanization impacts on surface runoff at the national scale provides an opportunity to assemble useful scientific datasets (soil, land use, and climate) and transfer these to quantitative information that can be easily conveyed to the general public. For example, model simulation results showed that: urbanization contributed an additional 3.3 billion cubic meters (10%) of average annual runoff nationally from 2001 to 2011; urban expansion and intensification serve as drivers altering surface runoff; urbanization leads to high runoff percent increases in the areas where the precipitation is low. The results can be used to strengthen the public's awareness of negative impacts of urbanization. Model inputs have direct influence on model simulation results, interpretations, and applications. Along with the development and improvements of scientific datasets, up-to-date datasets should always be considered, evaluated, and included in the model simulations to provide more accurate information. Based on the experiences of using simulated precipitation from the current and the updated CLIGEN database as the only change in input of the L-THIA model, variations in simulation results reflect the impacts of precipitation characteristics on runoff from urban areas. In some areas experiencing large increases in precipitation, the impacts of precipitation on surface runoff overweigh those from urbanization. In some areas experiencing decreasing precipitation, high increases in urbanization can lead to increases in surface runoff, which offset the effects from decreases in precipitation. Some areas experiencing both large increases in precipitation and urbanization, have shown greater increases in surface runoff than areas that only have either increasing precipitation or increasing urbanization. The outcomes of this research provide evidence to urban planners and the general public that GI practices could effectively mitigate the impacts of urbanization and climate at local scales. The methodologies regarding evaluating the effectiveness of GI practices can be extended to other areas and/or nationwide. Selecting suitable GI practices is one of the most important factors before GI implementation. A practice that works well in one location could potentially bring about problems in other areas. For example, rain barrels work well in places such as Indiana and Illinois, however, these could bring mosquito control issues in humid subtropical areas, such as in Florida. Some practices could also raise additional pollution concerns: for example, grass or weeds may grow in interspaces in porous pavements—depending on how they are designed—which might prompt the use of pesticides as a means of control thereby raising concerns related to pesticide pollution. Practices needed in areas with water scarcity issues are different from places where precipitation is ample. For instance, practices needed in California might have less capacity design than that of in Florida. GI practices are usually effective when dealing with high frequency, low intensity precipitation events. However, they are not perfect/one-for-all options especially when it comes to extreme precipitation events in which case everything could fail. The costs of different GI practices implementation vary largely, and the affordability of GI practices adoption needs to be evaluated based on the City's economic status. GI practices also have social impacts, for example, bioretention could enhance site aesthetics and provide wildlife habitat, which could potentially promote physical activity and the health of urban residents. Researchers need to collect more information regarding social, health, economic, and climatic conditions when selecting GI practices to find suitable GI practices for specific locations. Communication with the public is also important. From our modeling experiences, implementing GI practices only on city-owned properties could not achieve targeted overflow volume reduction. With collaborations with the public, more spaces and more GI practices could be considered to achieve greater environmental benefits.

5.3 Recommendations for Future Research

Beyond the national scale analysis, potential future research directions include: extending analysis to incorporate NLCD 2016; exploring urbanization impacts on streamflow; exploring urbanization impacts on water quality, for instance, computing non-point source pollution estimates using the runoff volume and the CN-land use pairs to choose the appropriate event mean concentration; comparing the results from this study to results obtained by applying nationwide spatially-distributed observed daily precipitation; comparing the results from this study to results by applying Soil Survey (SSURGO) geographic data; or simulating future land use change scenarios and their hydrological and environmental impacts, among others.

In the regional analysis, only one realization of 50 years of daily precipitation was generated using CLIGEN with the updated database and based on the default random seed, in order to keep consistent with prior work done using the current database. To better capture variability within the generated values by CLIGEN, more realizations should be considered. The comparison of the CLIGEN databases could be conducted in other regions beyond the Great Lakes Region.

The L-THIA-LID 2.2 model, does not simulate a routing processes in detail. Future research could be conducted in adding a routing method in the model. When running the model, GI practices were assumed fully functional in each daily simulation. Full public cooperation was assumed when calculating the total cost and cost effectiveness. Cost could incorporate more benefit options, for instance, the benefits of rain barrels/cisterns in collecting water for irrigation; the benefits of GI practices in increasing infiltration and reducing peak runoff and other pollutant loads; the benefits of GI practices, such as bioretention, in enhancing site aesthetics and providing wildlife habitat. Further, model results can be compared with other more sophisticated models, such as SWMM and SUSTAIN. In addition, combining the L-THIA-LID 2.2 model with an optimization algorithm could better answer the following questions: (1) within a given budget, how much more runoff and pollutant reductions are obtainable? (2) to achieve certain reduction goals, how much less would green infrastructure cost? and, (3) where are the best locations to place given GI practices so as to achieve the highest pollutant reductions at the least cost?

APPENDIX A

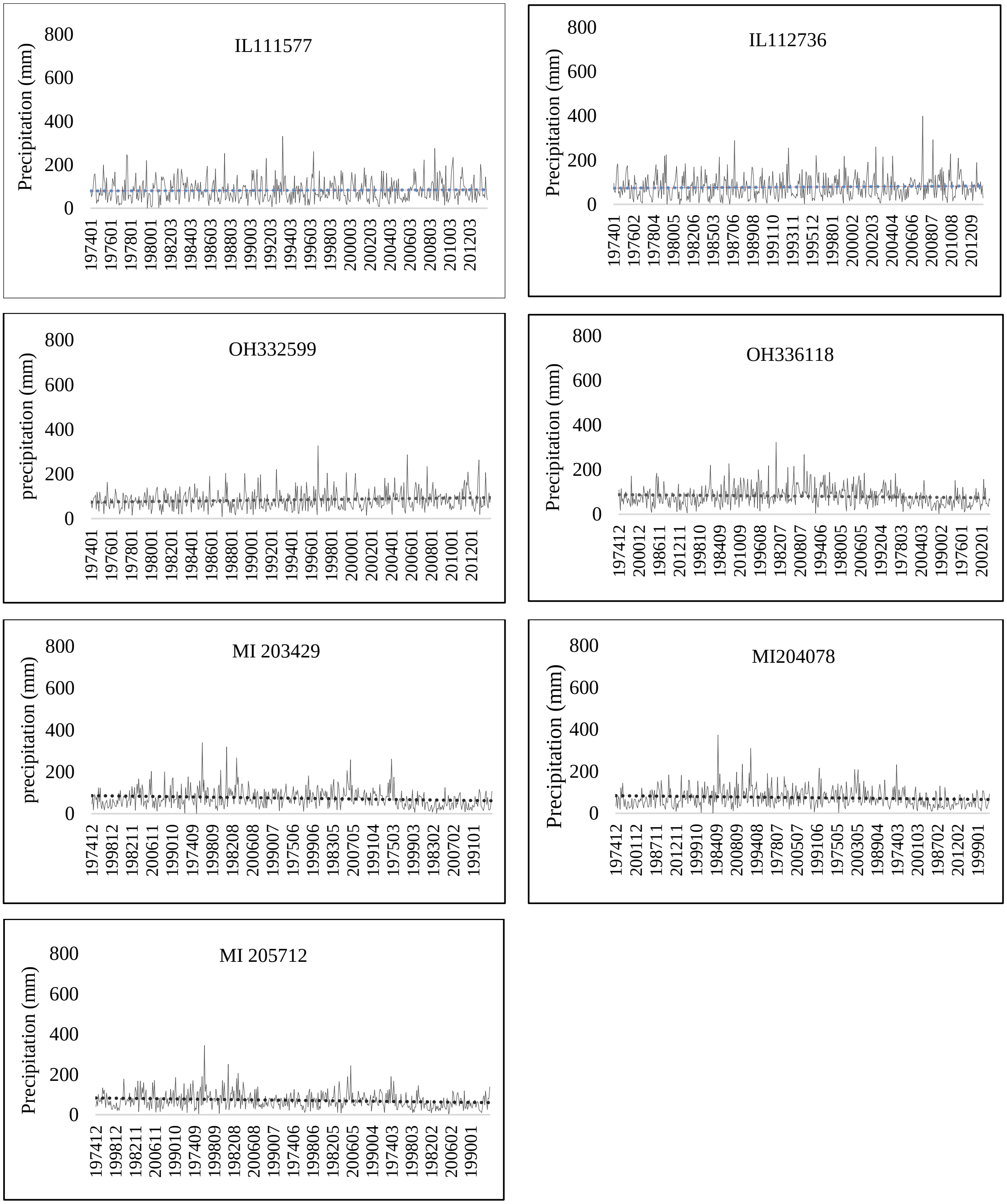

Figure A1 Historical monthly precipitation from 1974 to 2013 of seven weather stations.

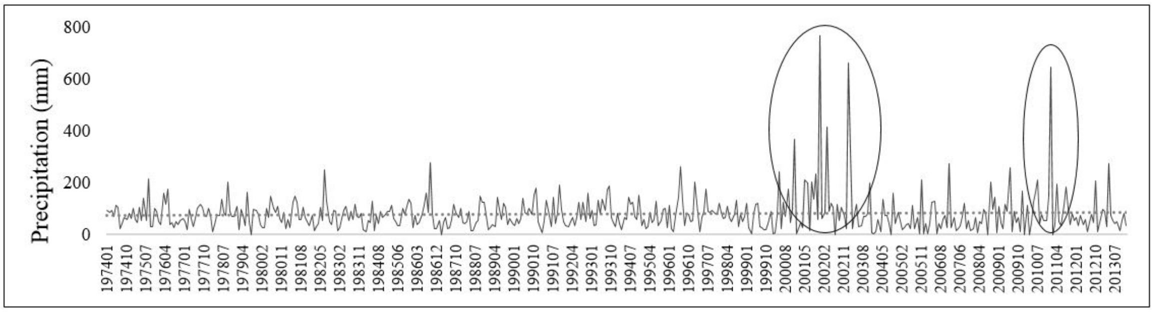

Figure A2 Historical monthly precipitation depths for station MI203858.

APPENDIX B

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INTRODUCTION

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APPENDIX C

Table C1 Simulated values of annual runoff volume and pollutant loadings associated with the baseline scenario (S0) in the Darst Sewershed

Year	Precipitation (mm)	Simulated annual values			
		Runoff (m^3/ha)	Total Suspended Solids (kg/ha)	Total Nitrogen (kg/ha)	Total Phosphorus (kg/ha)
1966	842.2	2345	123.2	4.4	1.8
1967	913.7	2220	116.6	4.2	1.7
1968	861.7	2450	128.6	4.6	1.9
1969	856.5	2220	116.5	4.2	1.7
1970	1137.1	3399	178.5	6.4	2.6
1971	670.9	1631	85.7	3.1	1.2
1972	921	2251	118.2	4.2	1.7
1973	1276.6	3363	176.7	6.3	2.5
1974	1080.3	2955	155.1	5.6	2.2
1975	1047.9	2859	150.1	5.4	2.2
1976	793.6	2232	117.2	4.2	1.7
1977	976.5	2447	128.5	4.6	1.8
1978	816.3	1873	98.4	3.5	1.4
1979	736.9	1705	89.6	3.2	1.3
1980	896.3	2564	134.6	4.8	1.9
1981	1010.5	2718	142.7	5.1	2.1
1982	1146.6	3439	180.5	6.5	2.6
1983	1067.1	2867	150.6	5.4	2.2
1984	1035.6	2444	128.4	4.6	1.8
1985	1113.6	2697	141.7	5.1	2.0
1986	951.5	2671	140.3	5.0	2.0
1987	697.3	1594	83.7	3.0	1.2
1988	563.8	1224	64.3	2.3	0.9
1989	573.3	1079	56.7	2.0	0.8
1990	1406.6	4380	229.8	8.2	3.3
1991	918.1	2184	114.7	4.1	1.6
1992	828.2	1939	101.9	3.6	1.5
1993	1373.5	3859	202.6	7.3	2.9

1994	640.7	1410	74.1	2.6	1.1
1995	858	2192	115.1	4.1	1.7
1996	750.3	1771	93.0	3.3	1.3
1997	809.3	1928	101.3	3.6	1.5
1998	1075.6	2721	142.9	5.1	2.6
1999	769.1	1897	99.7	3.6	1.4
2000	666.5	1280	67.3	2.4	1.0
2001	958.8	2364	124.2	4.4	1.8
2002	846.7	2383	125.1	4.5	1.8
2003	788	1939	101.9	3.6	1.5
2004	875.5	2149	112.9	4.0	1.6
2005	645.9	1499	78.7	2.8	1.1
2006	811.4	1824	95.8	3.4	1.4
2007	940.3	2303	121.0	4.3	1.7
2008	1183	3504	183.9	6.6	2.7
2009	1385	3856	202.5	7.2	2.
2010	1121.5	2759	144.9	5.2	2.1
2011	1013.4	2834	148.8	5.3	2.1
2012	688.3	1611	84.6	3.0	1.2
2013	1106.9	3070	161.2	5.8	2.3
2014	1011.4	2756	144.7	5.2	2.1
2015	1269.7	3714	195.0	7.0	2.8

APPENDIX D

Assessment of land use / land cover change impacts on surface runoff and nonpoint source pollution in the City of Peoria

An analysis of how land use / land cover change has impacted surface runoff and nonpoint source pollution in the City of Peoria was conducted using the L-THIA Tabular Tool (Chen et al., 2017) based on National Land Cover Datasets (NLCD 2001, 2006, and 2011), Soil Survey Geographic (SSURGO) Database, and 50-year of daily precipitation from CLIGEN (Climate Generator).

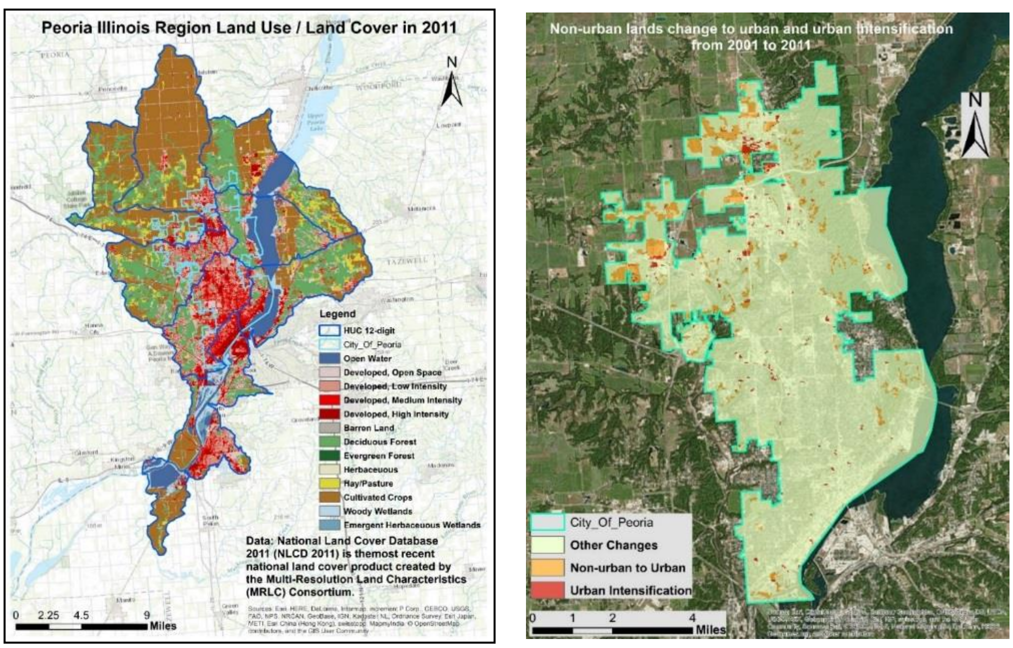

Figure D1 Land use / land cover of the City of Peoria and 12-digit hydrologic unit code (HUC 12) watersheds covering the City in 2011 (left) and urbanization from 2001 to 2011 in the City of Peoria, including non-urban change to urban and urban intensification

The City of Peoria is experiencing significant urbanization. From 2001 to 2011, approximately 826 hectares of land changed from non-urban to urban, and about 160 hectares of urban land were converted to higher intensity uses (Figure D1, and Table D1); average annual runoff volume increased about 437,850 m^3; Total Suspended Solids (TSS) increased about 27.2 tons from 2001 to 2011. Total Phosphorus (TP) loads changed within ± 0.6% with an approximate amount of 4.4 tons. Total nitrogen (TN) loads varied within ± 0.7% with an approximate amount of 16.1 tons (Table D2). These results provide the background of urbanization as well as pollutant load information, which could be informative for decision makers in the City of Peoria.

Table D1 Land use / land cover change in the City of Peoria among 2001, 2006, and 2011, red denotes urban land and urbanization increased from 2001 to 2011, and other colors denote decreased or unchanged land types.

	Area (km^2)				Change Percentage		
Land Use / Land Cover	2001	2006	2011	Sparkline	01 to 06 (%)	06 to 11 (%)	01 to 11 (%)
Open Water	7.07	7.05	7.04		-0.31	-0.18	-0.48
Developed, Open Space	13.20	15.51	16.24		17.48	4.72	23.03
Developed, Low Intensity	34.18	34.87	35.56		2.04	1.97	4.05
Developed, Medium Intensity	29.97	31.71	32.77		5.81	3.33	9.34
Developed, High Intensity	9.40	10.15	10.44		7.96	2.86	11.06
Barren Land	0.00	0.00	0.00		0.00	0.00	0.00
Deciduous Forest	24.20	21.41	20.50		-11.52	-4.27	-15.29
Evergreen Forest	0.01	0.01	0.01		0.00	0.00	0.00
Herbaceuous	0.28	0.09	0.06		-66.45	-37.86	-79.15
Hay/Pasture	1.38	0.99	0.73		-28.27	-26.79	-47.49
Cultivated Crops	9.21	7.28	5.76		-20.94	-20.81	-37.39
Woody Wetlands	1.39	1.21	1.18		-13.28	-2.09	-15.09
Emergent Herbaceuous Wetlands	0.22	0.22	0.22		0.00	0.00	0.00

Table D2 Environmental concerns simulated by the L-THIA Tabular tool for 2001, 2006, and 2011

	Value			Percentage Change			Trend
	2001	2006	2011	01 to 06 (%)	06 to 11 (%)	01 to 11 (%)	
Runoff Volume (million m3)	10.12	10.43	10.56	3.02	1.27	4.33	
Runoff Depth (mm)	78	80	81	3.02	1.27	4.33	
TSS (Ton)	536	557	564	3.75	1.26	5.06	
TP(Ton)	4.39	4.41	4.38	0.26	-0.61	-0.35	
TN(Ton)	15.98	16.09	16.05	0.71	-0.26	0.45	

VITA

Jingqiu Chen was born in Yunnan Province, China. She received her Bachelor of Science degree in Agricultural Resources and Environment from Nanjing Agricultural University in June 2010. She earned a Master of Science degree majoring in Soil Science from Nanjing Agricultural University in June 2013. Her master's work focused on the decision tree method for land use information extraction based on remote sensing images. She joined the Department of Agricultural and Biological Engineering at Purdue University in August 2013 to pursue a Ph.D. degree, majoring in Agricultural Engineering. Her dissertation research has primarily focused on assessment of urbanization impacts on surface runoff and effects of green infrastructure on hydrology and water quality.

PUBLICATIONS

Publications during Ph.D. study:

Chen, J., Gitau, M. W., Engel, B. A., & Flanagan, D. C. (2017). Suitability of CLIGEN precipitation estimates based on an updated database and their impacts on urban runoff: a case study of the Great Lakes Region. (Accepted for publication, Hydrological Sciences Journal, IF: 2.222)

Chen, J., Theller, L., Gitau, M. W., Engel, B. A., & Harbor, J. M. (2017). Urbanization impacts on surface runoff of the contiguous United States. Journal of Environmental Management, 187, 470-481. http://dx.doi.org/10.1016/j.jenvman.2016.11.017 (IF: 4.010)

Gitau, M.W., **Chen, J.**, & Ma, Z. (2016). Water quality indices as tools for decision making and management. Water Resources Management. 30 (8), 2591-2610. DOI: 10.1007/s11269-016-1311-0. (Invited Review Paper, IF: 2.848)