POTENTIAL OF STORMWATER GREEN INFRASTRUCTURE TO RECREATE THE NATURAL WATER BUDGET OF A SEMIARID URBAN CATCHMENT

by

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ABSTRACT

As one of the most important earth systems, the water cycle is significantly disrupted by changes to land cover and water management accompanying urbanization. Recently, researchers have developed a concept of near-natural hydrology to guide ecological engineering of urban systems to mitigate the impacts of development on the water cycle. Stormwater green infrastructure (GI) is one of the practices that has been used to restore the urban hydrology. The goal of this research is to answer the overarching question: Can GI implemented in a semiarid watershed restore the water budget to its predevelopment condition? Field experiments and hydrologic modeling were conducted in a semiarid city, Salt Lake City, Utah, U.S to answer this question.

This work created, for the first time, an ET observation dataset for the semiarid intermountain west of the U.S. Based on the new dataset, empirical parameters for Penman-Monteith ET methods, including crop coefficients and surface resistances for green roofs, were identified and calibrated for this region, also for the first time. Their values can be directly used for ET modeling of green roofs in similar climates.

An urban stormwater model, EPA SWMM, was modified to be able to represent spatially heterogeneous ET rates in one catchment for up to six types of land covers, including GI (bioretention, green roof), landscapes (turf, deciduous trees, coniferous trees), and water surface. This creates an improved platform to study the hydrologic response of

urban watersheds by addressing the limitation of hydrologic models, not including GI and stormwater models with poor representation of ET. Also, the EPA SWMM was modified to be able to operate using subdaily ET time series input for the first time.

With the updated model, the final part of this work studies the potential of restoring the predevelopment urban water budget by adopting GI strategies in a semiarid watershed. Based on the proposed water budget restoration coefficient, the water budgets have been restored due to GI applications 94%, 94%, and 82% of the predevelopment state in the dry, average, and wet years, respectively.

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CHAPTER 1

INTRODUCTION

Following economic opportunity and other pursuits, humans have increasingly concentrated in urban areas in the past century. Recently, the world's population surpassed 50% urban, with this number expected to increase to 67% by 2050 (Heilig 2012). The global trend of urbanization significantly alters the surface energy budget, hydrological cycle, and biogeochemical cycles at different scales and as a function of climate, ecosystem, and terrain characteristics (Foley et al. 2005; Grimm et al. 2008; Pauleit et al. 2005; Scalenghe and Marsan 2009; Seto and Shepherd 2009). This causes an array of environmental problems, like water resources degradation and scarcity (Bao et al. 2012), urban heat island (UHI) (Bornstein 1968; Brazel et al. 2000; Coutts et al. 2007; Gartland 2010; Godowitch et al. 1985; Kerr 2013; Morris and Simmonds 2000; Steinecke 1999; Todhunter 1996; Vez et al. 2000; Yamashita 1996), energy and resources consumption (Foley et al. 2005; Grimmond 2007; Medina Jr 2010), increased aerosols emissions (Seto and Shepherd 2009), increased greenhouse gas emissions (Grimmond 2007; Liu et al. 2013; Seto and Shepherd 2009), biodiversity loss (Grimm et al. 2008; MacIvor and Ksiazek 2015; McKinney 2006), and net primary productivity (NPP) loss (Seto and Shepherd 2009). Although the precise situation may vary between urban locations, one consistency is almost always urbanization leads to a modification of the water cycle.

1.1 Urbanization and Urban Hydrology

Urbanization effects on the water cycle are caused by land surface modification (Guan et al. 2016; Jacobson 2011; Pauleit et al. 2005; Powell et al. 2008; Scalenghe and Marsan 2009; Shuster et al. 2005; Whitford et al. 2001; Yao et al. 2016) and changes to water management. Human modifications of the water cycle remain an area of great uncertainty, with a limited ability to project implications on underlying mechanisms and feedbacks to associated interconnected systems. The majority of research agrees that urbanization can significantly modify the precipitation variability and intensity in urban areas (Seto and Shepherd 2009; Shem and Shepherd 2009; Trusilova et al. 2008), but it is still not clear enough to conclude whether it increases (Hand and Shepherd 2009; Shepherd 2006) or decreases (Kaufmann et al. 2007; Rosenfeld 2000) urban storm occurrences and characteristics. The introduction of impervious areas and wide-spread stormwater drainage networks tends to lower groundwater recharge (Jeppesen et al. 2011; Rose and Peters 2001). However, for some cases there may be a substantial increase of indirect recharge from storm drainage systems, irrigation return flows, and leaks from pipe networks (Barron et al. 2013; Foster et al. 1994; He et al. 2009; Hibbs and Sharp 2012; Lerner 1990; Lerner 2002; Zhang and Kennedy 2006). Similarly, baseflow varies with alterations related to urbanization (Fletcher et al. 2013; Hamel et al. 2013), which may decrease due to reduced infiltration (Brun and Band 2000; Nie et al. 2011), or increase due to external water inputs or stormwater management practices (Jacobson 2011; White and Greer 2006).

Uncertainties also exist for the variations of urban evapotranspiration (ET). Some studies report impervious surfaces can generate a significant magnitude of ET (Ramier et al. 2011). Irrigation on urban surfaces and heat advection at microscale and mesoscale may partially compensate for reduced ET from a decreased amount of green spaces (Dimoudi and Nikolopoulou 2003; Gober et al. 2009; Grimmond and Oke 1986; Oke 1979; Shields and Tague 2012). However, other studies noted increased surface sealing and underdrains in shallow unconfined aquifers often results in a significant decline of the overall urban ET, accompanied with a significant increase in sensible heating of the atmosphere (Barron et al. 2013; Dow and DeWalle 2000; Haase 2009; Jeppesen et al. 2011; Rose and Peters 2001; Wijesekara et al. 2012).

Due to the reduction of ET, storage, and direct infiltration, increases of stormwater runoff volumes and peak discharge magnitudes have been widely reported (Boggs and Sun 2011; Lee and Heaney 2003; Rose and Peters 2001; Weng 2001; Wu 2015; Zhang et al. 2013). Increased runoff is directly connected to a wide array of environmental stressors (Hasse and Lathrop 2003), such as flood risk (Du et al. 2012; Haase 2009; Liu et al. 2006; Rutland and Dukes 2012; Wijesekara et al. 2012), sediment erosion and transport (Nie et al. 2011), stream quality degradation (Astaraie-Imani et al. 2012; Foley et al. 2005; Interlandi and Crockett 2003; Zgheib et al. 2012), aquifer pollution (Chisala and Lerner 2008; Hibbs and Sharp 2012; Lerner and Barrett 1996), waterborne diseases (Narain 2012; Vörösmarty et al. 2000), acidification of water bodies (Kelly et al. 2011; Xiao et al. 2012), and aquatic species loss (Gillies et al. 2003). Stormwater runoff issues have been targeted by environmental regulation (e.g., Clean Water Act, National Pollutant Discharge Elimination System (NPDES) regulations (Lehner 2001; USEPA 2000) and design advances.

1.2 Urban Water Cycle Restoration

A critical direct impact of urbanization on ecosystems is caused by alterations to the hydrologic cycle, which controls ecosystem energy and matter fluxes (Sun and Lockaby 2012). Historically, and with renewed interest recently, attention has focused on restoring or rehabilitating urban hydrology to the predevelopment or near-natural state, with the goal to improve ecological functioning and system linkage to the urban environment (Findlay and Taylor 2006; Vollmer 2009). Green Infrastructure (GI) is a relatively new set of practices promoted by the United States Environmental Protection Agency (EPA) and others to manage stormwater in a sustainable way (USEPA 2000). GI as an approach provides an alternative to traditional grey infrastructure for stormwater runoff control. The overarching goal of GI is to restore the natural hydrologic cycle, when implemented as part of Low Impact Development (LID) strategies (USEPA 2000), which are similar in concept to Water Sensitive Urban Design (WSUD) (Lloyd et al. 2002; Wong 2006), Sustainable Urban Drainage Systems (SUDS) (Fletcher et al. 2008), and Stormwater Control Measures (SCMs) (Wadzuk et al. 2013). Notably, as EPA's definition (http://water.epa.gov/infrastructure/greeninfrastructure/gi_what.cfm), the concept of GI could also be expanded to include "green" landscapes in cities, such as residential lawns, parks, golf courses, and more, which occupy large portions (40-70%) of surface areas in European and North American cities (Oke 1982). Contrary to the traditional centralized stormwater conveyance system, which may significantly increase the peak discharge and the flashiness of storm runoff (Miller et al. 2014), GI adopts the philosophy of distributed networks to process stormwater and pollutants at their sources. Via various forms, GI can apply surface storage to reduce the amount of (and delay) urban stormwater runoff, supplement water supply through rainwater harvesting (RWH) (Hoff et al. 2010; Wisser et al. 2010), and introduce low-water-use landscapes to reduce landscape irrigation, often the highest demand in a city (Kenny et al. 2009; Reclamation 2012). Other derived ecosystem benefits include baseflow restoration (Endreny and Collins 2009; Hamel et al. 2013), stream erosion prevention (Tillinghast et al. 2011), surface cooling and UHI relief (Coutts et al. 2013; Nakayama and Hashimoto 2011; Scherba et al. 2011; USEPA 2003; Wong et al. 2003), building energy demand reduction (Kumar and Kaushik 2005), water quality improvement (Dietz and Clausen 2008; Kim et al. 2012), air pollutant collection (Currie and Bass 2008; Rowe 2011; Yang et al. 2008), acid rain mitigation (Berndtsson 2010; Davis et al. 2009), greenhouse gas absorption (Gill et al. 2007), habitat protection (Madre et al. 2015; Páll-Gergely et al. 2015), aesthetic landscaping (Kambites and Owen 2006; Sandström 2002), and noise blocking (Kambites and Owen 2006; Sandström 2002). Due to stormwater management and other environmental benefits, GI is a desirable climate-adaptive measure.

Restoring the urban water budget is an important means to mitigate the impacts of urbanization on the environment, economy, and inhabitants of cities. A critical, yet often overlooked, component of the water budget to restore is ET, as noted in the following:

(1) Restoring predevelopment stormwater runoff volumes and peakflow rates is supported by restoring ET capacity. Reducing stormwater runoff decreases flooding and pollutant loading (Boggs and Sun 2011; Sun and Lockaby 2012), and related economic loss and social disturbances.

(2) Restoring ET capacity can contribute to UHI mitigation (Alexandri and Jones 2008; Krayenhoff and Voogt 2010; Sailor 1995; USEPA 2008) and, in turn, reduced energy (cost)

for cooling during the summer (Gartland 2010; Levallius 2005; Saadatian et al. 2013; Takebayashi and Moriyama 2007). Objects that help to restore ET capacity in cities also help to reduce heating costs during the winter (Castleton et al. 2010; Gartland 2010; Levallius 2005), and the related energy consumptions and power bills (Alexandri and Jones 2008; Barrio 1998; Fioretti et al. 2010; Getter and Rowe 2006; Kumar and Kaushik 2005; Lazzarin et al. 2005; Levallius 2005; Mitchell et al. 2008; Ouldboukhitine et al. 2011; Saadatian et al. 2013; Takebayashi and Moriyama 2007; USEPA 2008).

(3) ET restoration by green roofs generates cool air, which may give rise to strengthened street canyon flow and improve air quality near roads (Baik et al. 2012).

(4) Restoring ET is facilitated by introducing green spaces in cities, which create space for plants and strengthen carbon sinks, especially in arid regions (Sun et al. 2011).

(5) Additional benefits of ET restoring green spaces in cities are improved biodiversity associated with fauna and flora biodiversity protection (Currie 1991).

Most urban stormwater management goals have focused on restoring runoff regimes (Ambrose and Winfrey 2015; Booth et al. 2004; Ellis and Viavattene 2014; Guan et al. 2015a; 2015b; Jarden et al. 2015; Loperfido et al. 2014; Petrucci et al. 2013; Simpson 2007; Wella-Hewage et al. 2016) and groundwater recharge amounts (Kidmose et al. 2015; Moglia et al. 2010; Shuster et al. 2007). Although maintaining ET seems appropriate and is recognized as an important component of the water budget and overall watershed health (Boggs and Sun 2011), critical studies to improve understanding of approaches are still needed (Walsh et al. 2015).

1.3 Urban Water Budget Modeling

As a simplification of complex processes, modeling is often used as a planning tool to simulate urban hydrology and predict the restoration of the urban water budget. Traditional hydrologic models often do not integrate infrastructures of stormwater, water supply, and wastewater (Cleugh et al. 2005; Dupont et al. 2006; Grimmond et al. 1986; Järvi et al. 2011; Lemonsu et al. 2007; Mitchell et al. 2001), while conceptual models are often unable to accurately represent hydrologic processes like routing and water quality (Fletcher et al. 2013; Mitchell et al. 2008; Zoppou 2001). Hydrologic models in general lack capacity to simulate GI units and evaluate their impacts on restoring the urban water budget.

Stormwater models generally do have the capacity to model GI. But most of those models currently are based on the goal of designing stormwater runoff volume reduction and flood control measures (Cuo et al. 2008; Hamel and Fletcher 2014; James and Dymond 2012; Jia et al. 2012; Jia et al. 2002; LeFevre et al. 2010; Lucas 2010; Nanía et al. 2014; Schmitt et al. 2004; Shuster and Rhea 2013; Xiao et al. 2007; Young et al. 2011). Among stormwater models with GI modules, the U.S. Environmental Protection Agency's Storm Water Management Model (SWMM) provides the highest level of fidelity and proven accuracy (Elliott and Trowsdale 2007; Jayasooriya and Ng 2014; Lee et al. 2010).

Stormwater runoff models generally oversimplify hydrologic processes like ET because of the uncertainties associated with modeling ET in urban areas (Berthier et al. 2006) and the historical lack of concern for ET for drainage system design and flood control. The full water budget is also not well represented for the same reasons. For example, irrigation is often excluded, which is an especially important water budget component in arid and semiarid cities. Consequently, the accuracy of stormwater runoff spatial and temporal predictions is compromised. These issues and others noted above highlight the current lack of an integrated model capable of simulating GI hydrology and the urban water budget at the same time (Patrick et al. 2004), which prevents the evaluation of the impacts of GI on restoring the urban water budget under different scenarios and climate conditions (Andrés-Doménech et al. 2012; Fletcher et al. 2013; Forsee and Ahmad 2011; Karamouz et al. 2011; Karamouz and Nazif 2013; Kerr 2013), especially in the potentially warmer and drier southwest U.S. (Acharya et al. 2012; Division et al. 2011; Eum et al. 2010; Gober et al. 2011; Huntington 2006; Interior and Reclamation 2011; Jenerette and Larsen 2006; Kerr 2012; 2013; Mallya et al. 2013; Oki and Kanae 2006; Overpeck 2013; Seneviratne 2012; Sheffield et al. 2012; Tavakoli and De Smedt 2011; Wild et al. 2008).

1.4 Semiarid Urban Ecosystem

Due to population growth and migration, semiarid regions in the U.S. are being developed to accommodate new residents, and some of those developing areas are among the fastest growing in the country (Houdeshel et al. 2012). Compared to humid regions, semiarid regions are generally more sensitive to urbanization in terms of both hydrologic modifications and water resource sustainability (He and Hogue 2011). This leads to many uncertainties in water budgets, because ET, runoff, groundwater recharge, and leakage are poorly constrained (Pataki et al. 2011). Therefore, improving understanding of the relation of land use to hydrologic functions is especially important in water-stressed regions (Johnson et al. 2001).

Most of the existing studies about semiarid and arid regions focus on the human side of the ecosystem. Typically, hydrologic responses related to human impacts are primarily studied as changes to runoff (Ambrose and Winfrey 2015; Hale et al. 2014; Hogue 2009) and recharge (Carlson et al. 2011; Gallo et al. 2013; Perrin et al. 2012). However, environmental aspects of the ecosystem are often neglected, especially when conflicts between human demands and environmental demands emerge. The dominating attitude to consume and modify the ecosystem according to development prevails (Wackernagel and Rees 1998). One solution approach is to conduct land use change in a sustainable manner. This is manifested in many forms, including converting natural landscapes into homogenous drought-tolerant xeriscape or nonvegetated zeroscape during urbanization, which has the potential to reduce water demand and associated environmental impacts. More broadly, there is potential to replace the loss of plant and insect biodiversity and increase the connectivity of fragmented green space (MacIvor and Ksiazek 2015), and moisture feedback to the atmosphere in ecologically sensitive regions.

ET has often been neglected in the urban water budget studies because of lack of interest for engineering design and the perceived lack of importance compared to runoff and recharge in semiarid urban areas. On the contrary, ET is important both spatially and temporally, and yet a highly uncertain component of urban water budgets in semiarid environments (Augustus 2008; Litvak et al. 2014; Shields et al. 2008). There has been and remains a need to continue to improve the understanding and ability to represent ET in water budget and stormwater management studies in semiarid urban environments.

The relative importance of GI compared to other urban landscapes in terms of restoring urban ET and urban hydrology is unknown. Due to the lack of relevant studies, further studies of the effects and efficiency of GI in restoring urban hydrology, especially ET, are needed for semiarid climates to help guide urban water management in the face of uncertainty. Some uncertainties have been identified related to GI in semiarid and arid areas like the intermountain west U.S., where the basis for urban growth and economic development is the ability to manage water with variable resource needs and infrequent but intensive precipitation events (Gober et al. 2011). On one hand, as GI may bring more vegetation in desert cities (Seto and Shepherd 2009), which could also help to cool the hot cities (Avissar 1996; Bonan 2000; Bowler et al. 2010; Buyantuyev and Wu 2010; Emmanuel et al. 2007; Hart and Sailor 2009; Huang et al. 1987; Loughner et al. 2012; Middel et al. 2012; Quattrochi and Ridd 1998; Sailor 1998), more imported water may be needed to keep the plants alive during establishment and during dry periods between rain events (Grimmond and Oke 1999; McCarthy and Pataki 2010; Milesi et al. 2005; Pataki et al. 2011; Williams et al. 2010). On the other hand, irrigation amounts and patterns need to be wisely designed so that irrigation will not affect GI stormwater management performance for the next storm event (Voyde et al. 2010). Also, having urban green space, including GI and traditional landscapes, in semiarid and arid climates may raise the cities' water demand. However, irrigation can also increase green space and GI's cooling effects on urban microclimate (Grossman-Clarke et al. 2010; House-Peters and Chang 2011). Thus, an ability to conduct a comprehensive analysis of water budget responses to GI and traditional landscape applications is needed in semiarid cities, where the decisions eventually are based on the tradeoffs between investments of resources like water and multiple benefits to society, economy, and environment (Bonan 2000; Gober et al. 2009; Gober et al. 2012; Mitchell et al. 2008; Pataki et al. 2011; Shashua-Bar et al. 2009; Williams et al. 2010).

1.5 Framework

The motivation of this work is therefore to explore the potential of near-natural stormwater management in a semiarid climate. The final research question to be answered is, Can GI implemented in a semiarid urban watershed restore the water budget to its predevelopment condition? However, a lack of urban water budget modeling tools capable of simulating ET in a physically realistic way creates a barrier, as ET might be a more significant component of the water budget in the semiarid climate than other wetter climates where ET models are often oversimplified. A research question that must be answered before the final research question then is, Which methods are appropriate to model GI ET and can be incorporated into an urban watershed model? However, the lack of a realistic ET model for GI is often due to a lack of existing datasets generated from high-quality field experiments, which are costly and difficulty to conduct. While hydrologic performance of stormwater bioretention units in semiarid climates has been studied (Heiberger 2013; Houdeshel 2013; Orr 2013; Steffen 2012), there are few studies of green roofs in this climate. So the first step of this research is to establish a field experiment to answer the research question, What are the rate and seasonal pattern of ET of GI, and in particular a green roof, in a semiarid climate?

While three research questions are identified and linked in a top-to-bottom logic line, they need to be answered in a bottom-to-top order (Fig. 1). Research activities related to research question 1 will provide data to develop the ET model to be incorporated into the updated water budget model to answer research question 2. The research activities related to research question 2 will provide the necessary modeling platform to test the hypothesis to answer research question 3.

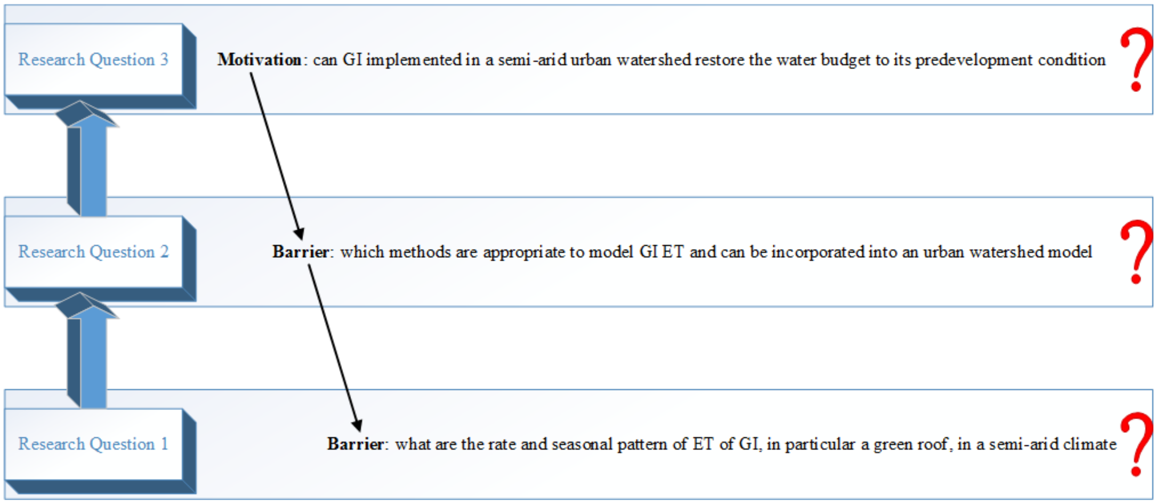

Fig. 1. Driving questions of the dissertation.

The research activities to answer research question 1 are summarized in Chapter 2. This question was approached by performing in 2014 a series of field experiments using weighing lysimeters, which have been used to study green roofs in wet climates (DiGiovanni et al. 2013; Feller 2011; Wadzuk et al. 2013). The key finding from this study is that the Penman-Monteith-related methods can provide a good estimate of potential ET for green roofs in the studied semiarid climate, and the methods using the annually averaged parameters can achieve similar accuracy as using the monthly parameters.

The research activities to answer research question 2 are summarized in Chapter 3. This question is designed to address the gap in model capacity by introducing and testing an enhanced ET modeling component in the EPA SWMM. Two common types of GI, including bioretention and green roofs, are studied by building on field experiments in this project and from others and incorporating them into the model development process. Compared to the original version of EPA SWMM, the updated SWMM shows more realistic ET patterns in temporal and spatial resolutions.

The research activities to answer research question 3 are summarized in Chapter 4. The hypothesis of this research is that GI can effectively restore the urban water budget to the predevelopment condition in the studied semiarid climate. The updated SWMM is used as the main platform to simulate the urban water budget. The estimated water budgets in different scenarios of development conditions exposed to different annual precipitation amounts (for a given year) are compared. This study demonstrates that the water budget is more influenced by the impacts of development in the studied semiarid climate than other wetter climates, but GI can also more effectively restore the water budget in this climate. A proposed metric can be useful to evaluate GI plans for the goal of achieving near-natural hydrologic response of a watershed for other locations.

CHAPTER 2

GREEN ROOF EVAPOTRANSPIRATION OBSERVATION AND ESTIMATION IN A SEMIARID CLIMATE

2.1 Introduction

A green roof is a common type of green infrastructure (GI) used in cities, and has been specifically applied to stormwater management since the 1960s (Levallius 2005). One of its most significant stormwater management benefits is that 100% of its footprint provides stormwater management services in the form of storage and filtration, in addition to other purposes (Fletcher et al. 2013). Hence, it can be an effective option for introducing vegetation into older city centers (USEPA 2008) or where surface areas are limited and expensive (Fletcher et al. 2013). In addition, a green roof is regarded as one means to cool an urban environment and mitigate the urban heat island effect (Barrio 1998; Kosareo and Ries 2007; Ouldboukhitine et al. 2011; Saadatian et al. 2013; Scherba et al. 2011; Wong et al. 2003; Wong et al. 2003). Compared to other types of cool roof alternatives and stormwater management practices, green roofs provide additional environmental benefits for atmospheric pollutant and carbon dioxide absorption, habitat and biodiversity protection, noise reduction, landscape aesthetics, and roof membrane protection (Castleton et al. 2010; Getter and Rowe 2006; Levallius 2005; Smith and Roebber 2011; USEPA 2008). Further, green roofs also have a lower life cycle cost (Berndtsson 2010; Spatari et al. 2011; USEPA 2008; Wong et al. 2003) in spite of typically higher initial cost than regular roofs (Carter and Keeler 2008; Levallius 2005; Saadatian et al. 2013; USEPA 2008; Wong et al. 2003).

Previous studies on the hydrologic performance of green roofs have largely focused on stormwater runoff controls. Most studies conclude that green roofs extend runoff duration (Getter and Rowe 2006; VanWoert et al. 2005a) and reduce volume (Berghage et al. 2009; DiGiovanni et al. 2010; Fassman-Beck et al. 2013; Jarrett et al. 2007; Levallius 2005; VanWoert et al. 2005a). However, other components of the water budget of green roofs also need attention to fully understand the mechanism for reducing stormwater runoff.

A recent study has proven that accurately estimating potential evapotranspiration (ET) rates is essential in representing the retention capacity regeneration and stormwater modeling (Krebs et al. 2016). A green roof, different from other types of GI, relies primarily on ET as the mechanism to reduce the amount of retained water between storm events (Voyde et al. 2010). In addition to its insulation (Barrio 1998; Castleton et al. 2010; Ouldboukhitine et al. 2011), reflection (Tsang and Jim 2011), and shading (USEPA 2008) properties, a green roof can also act as an evaporative device which uses ET as a way to cool roof surfaces in warm seasons (DiGiovanni et al. 2010; Feller 2011; Jarrett et al. 2007; Klein and Coffman 2015; Levallius 2005; VanWoert et al. 2005a; Voyde et al. 2010). The ability to model ET processes of green roofs is critical to quantifying the benefits needed for planning and designing urban areas to meet multiple sustainability objectives.

To date, most studies of green roof ET have been conducted in wetter climates (Berretta et al. 2014; DiGiovanni et al. 2013; Schneider et al. 2011; Wadzuk et al. 2013), leaving ET processes of green roofs less understood in dry regions like the western U.S. Further, since transpiration may contribute to less than 50% of the ET from green roofs (Voyde et al. 2010), evaporation from the soil covered by plants plays an important role with green roofs. "Soil roofs" have also been found to have similar and even higher evaporation rates compared to the ET rates of green roofs (Takebayashi and Moriyama 2007), especially in low and high initial soil moisture conditions (Berretta et al. 2014). But these theories have not been tested in semiarid climates.

Relative to research in humid environments, very few research studies have been conducted to quantify green roof hydrologic performance in semiarid or arid areas, where higher temperatures and vapor pressure deficits boost evapotranspiration (ET) rates and affect the amount of rainwater absorbed and runoff reduced. In addition, the Intermountain West of the U.S. generally has less intense and lower amounts of precipitation concentrated during winters with long dry periods during summers. This precipitation pattern and seasonal temperatures affect the ET rates and hydrologic performance of green roofs for stormwater management.

Irrigation is often needed for GI, such as green roofs in semiarid regions, to sustain vegetation during dry periods (Pataki et al. 2011; Williams et al. 2010). Applying irrigation water to city landscapes has been shown to enhance urban ET (Dimoudi and Nikolopoulou 2003; Gober et al. 2009; Grimmond and Oke 1986; House-Peters and Chang 2011; Oke 1979), especially in hot and dry regions where high atmospheric demands and intensified heat advection prevail. The water budget implications of implementing green roofs in the Intermountain West is not well understood.

Collectively, these issues highlight a general lack of understanding of the hydrologic performance, especially ET mechanisms, of green roofs in water-limited climates

(Oberndorfer et al. 2007). Common studies use simplified versions of the potential ET (PET) equation developed from agricultural reference crop without the adjustment of plant-specific parameters (DiGiovanni et al. 2013; Wadzuk et al. 2013), which is recognized as a need for refinement to better reflect the processes occurring within green roof configurations (Krebs et al. 2016; Stovin et al. 2013). A lack of high-quality experiments for the development, validation, and calibration of green roof ET models (Voyde et al. 2010) currently constrains the understanding and modeling of green roof ET processes and their environmental benefits in arid and semiarid environments (Berndtsson 2010).

The study presented here seeks to address these gaps. A series of weighing lysimeter-based experiments were conducted in the semiarid metropolitan area of Salt Lake City, Utah. Four replicates were constructed to directly observe ET and other components of the water budget. Three P-M equation methods were tested and compared with ET observations. The research results were expected to contribute to the knowledge base to improve modeling of green roof ET and more generally the hydrologic performance of green roofs.

2.2 Study Site

This study was conducted in Salt Lake City, Utah, U.S., which is at an average elevation of 1,320 m. According to the Köppen climate classification (Peel et al. 2007), Salt Lake City has a semiarid continental climate (Bailey 1979; Bair 1992; Eubank and Brough 1979; Quattrochi and Ridd 1998; Ramamurthy and Pardyjak 2011; Russell and Cohn 2012). From 1980 to 2010, the average annual precipitation was 409 mm and the average annual air temperature was 11.5˚C (NOAA 2016). The city experiences cold, snowy winters, hot, dry summers, and a relatively wet transition period (Russell and Cohn 2012). The lake-effect from the Great Salt Lake is a large contributor to the snowfall in the winter (Alcott and Steenburgh 2013). The primary source of summer precipitation is monsoon moisture moving in from the Gulf of California. In the winter, the temperature frequently stays below freezing, and evening fog and daytime haze are common (Russell and Cohn 2012).

The main study site was located at the intensive green roof on top of the third floor outdoor mezzanine of the Marriott Library (Library) at the University of Utah campus (40.7623°N, 111.8468°W), which has a 632 m^2 area (305 mm to 457 mm deep soil). The total plant coverage on the roof is approximately 51%, which mainly consists of herbaceous plants (74%) and shrubs (19%). Blue Grama Grass, *Bouteloua gracilis* is the dominant species in the garden, which covers 37% of vegetated area; while sedums, *Sedum spurium 'Red Carpet'* and *Sedum kamtschaticum*, rank as the 4th most abundant species, which take up a total of 77% of the vegetated area. Below the plants, the green roof is composed of the growth medium, filter fabrics, water retention panels, a moisture retention mat, a drainage mat, root barriers, and waterproof membranes. The growth medium was made of the Utelite E-Soil® Root Zone Mix, which was mechanically blended by the Utelite Corporation (Salt Lake City, UT). It was volumetrically composed of 50% screened loamy topsoil, 25% Utelite 'fines' expanded shale, and 25% approved compost. The bulk density is 1.02 g/cm^3, while the saturated bulk density is 1.5 g/cm^3. From the test report of Utelite Corporation, the soil porosity is 0.58, pH is 7.1, organic matter is 8.0% by mass, and saturated hydraulic conductivity is 1.4 mm/min. The maximum water retention is also claimed to be 0.48. However, it was observed as 0.35 when the medium is fully saturated during the process of calibrating the soil moisture sensor for the study. The green roof was also installed with Rain Bird (Azusa, CA) rotating sprinklers controlled by a soil-moisture-based control system (triggered at a volumetric soil water content of 0.21) to irrigate the green roof at nights from May to early November.

The second site was located at the extensive green roof on top of the fifth floor of the Natural History Museum of Utah (Museum) (40.7639°N, -111.8227° W). It is composed of multiple green roof strips separated by solar panel arrays. The total vegetated area is approximately 1,115 m^2. The soil depth is 102 to 152 mm. Designed by Design Workshop Inc. (Salt Lake City, UT), the green roof is made of growth media, filter fabric, drainage mats, and a waterproof membrane. The medium type is Utelite E-Soil® 60/40 lightweight planting media, a product mechanically blended by Utelite Corporation (Salt Lake City, UT, U.S.A.), which is made up of 60% Utelite 'fines' expanded shale and 40% approved organic matter. It has a pH value of 7.0. The bulk density is 0.91 g/cm^3, while the saturated bulk density is 1.42 g/cm^3. Its water retention percentage is claimed to be 50%. The organic matter is 6.4% by mass. The vegetated herbaceous plants cover 22% of the roof area. Sedums cover the 44% of the planted area, which are a mix of Red Carpet Stonecrop, *Sedum spurium 'Red Carpet'*, Bailey's Gold Stonecrop, *Sedum floriferum*, Russian Stonecrop, *Sedum kamtschaticum*, Variegated Stonecrop, *Sedum lineare "Variegatum"*, and Spruce Stonecrop, *Sedum reflexum*. Tubes buried under the soil are used for drip irrigation, while the watering schedule was controlled centrally by the Ground Maintenance Department of the University of Utah.

2.3 Methods

2.3.1 Field Experiment

Weighing lysimeters offer a cost-effective and engineering-efficient way to combine multiple measurement missions together into one experiment, and they have been successfully applied to estimate ET for green roofs (DiGiovanni et al. 2013; Feller 2011) and stormwater bioretention systems (Denich and Bradford 2010). Moreover, they also account for advection effects, which most other meteorological measuring methods cannot capture. The mass changes of the container on certain time scales (e.g., hourly or daily) are measured and converted into ET volumes. The water balance of a weighing lysimeter can be written as follows to calculate ET:

$$ET = P + IR - O - Q - \Delta S, \quad (1)$$

where ET is evapotranspiration; P is precipitation; $Irrig$ is Irrigation; O is surface overflow, Q is discharge, and ΔS is the change of the storage. Surface overflow was neglected in this study since the selected soil was highly pervious and no runoff was observed. Precipitation and irrigation were measured with tipping bucket rain gauges modified for the task. During the periods when precipitation, irrigation, and discharge are zero, the ET can be measured directly by scales and the water balance becomes

$$ET = -\Delta S. \quad (2)$$

Mass changes were used to determine ΔS using a custom-built weighing lysimeter. The lysimeter was made of a rectangular acrylic container (1.22 m by 0.61 m by 0.36 m) sealed with adhesives and silicone. An underdrain made of a bulkhead fitting and a valve was installed at one side of the lysimeter on the bottom. The container contained a replica of the existing green roof, which from the top to the bottom includes layers of growth media, a filter fabric, and an AmerGreen (Thousand Oaks, CA) 50RS drainage mat. The type of growth media used in the lysimeters was chosen to match the actual green roof setting. The depth of the growth medium within the container was designed to be 254 mm.

On the Library site, three weighing lysimeters were built to observe the ET of the nonvegetated growing medium, sedums (Red Carpet Stonecrop, *Sedum spurium 'Red Carpet'* and Russian Stonecrop, *Sedum kamtschaticum*), and grass (Blue Grama grass, *Bouteloua gracilis*) (Fig. 2). Those plants were transplanted into the lysimeters in September, 2013, after several years of growth on site as part of the green roof installation. Rice Lake BenchMark HE weighing scales (Rice Lake, WI, capacity: 454 kg, resolution: 0.05 kg ~ 0.06 mm, tolerance: 0.08 kg ~ 0.10 mm) with IQ355 indicators were used to track the mass balances of the lysimeter. The lysimeters were placed near the green roof surface to reduce the temperature and wind differences from the surroundings. Decagon (Pullman, WA) 5TM sensors were buried at the surface and bottom layers of each lysimeter to measure soil moisture and temperature. Another pair of 5TM sensors was installed in the actual green roof garden at the surface and base as a comparison. Campbell Scientific (Logan, UT) TE525 tipping bucket rain gauges were modified and buried underground to measure the discharge from the underdrain of each lysimeter. Another three TE525 tipping buckets were placed near each corresponding lysimeter to measure the inflows, including precipitation and irrigation from the rotating sprinklers. A Campbell Scientific CR1000 datalogger linked with a Campbell Scientific AM16/32B multiplexer was used to record data from all the sensors mentioned above and to compute 5-minute averages. Data collection started on January 1, 2014 and ended on December 31, 2014, providing a complete year of observation. The 5-min ET data were summed to produce hourly and

Fig. 2. Experimental set-up on top of the Marriott Library on the University of Utah campus (surface covers of the three weighing lysimeters are indicated, with medium meaning nonvegetated).

daily values used in the study.

The year 2014 was relatively wet, with a total precipitation of 496 mm compared to the average annual precipitation of 409 mm. The green roof was also well watered by sprinklers during the summer months, with a total of approximately 442 mm of applied water (Fig. 3). Discharge from the lysimeter underdrains was observed after most irrigation events. ET rates measured in this study were assumed to represent the potential ET (PET) rates.

A micrometeorological weather station was deployed near the three lysimeters on the

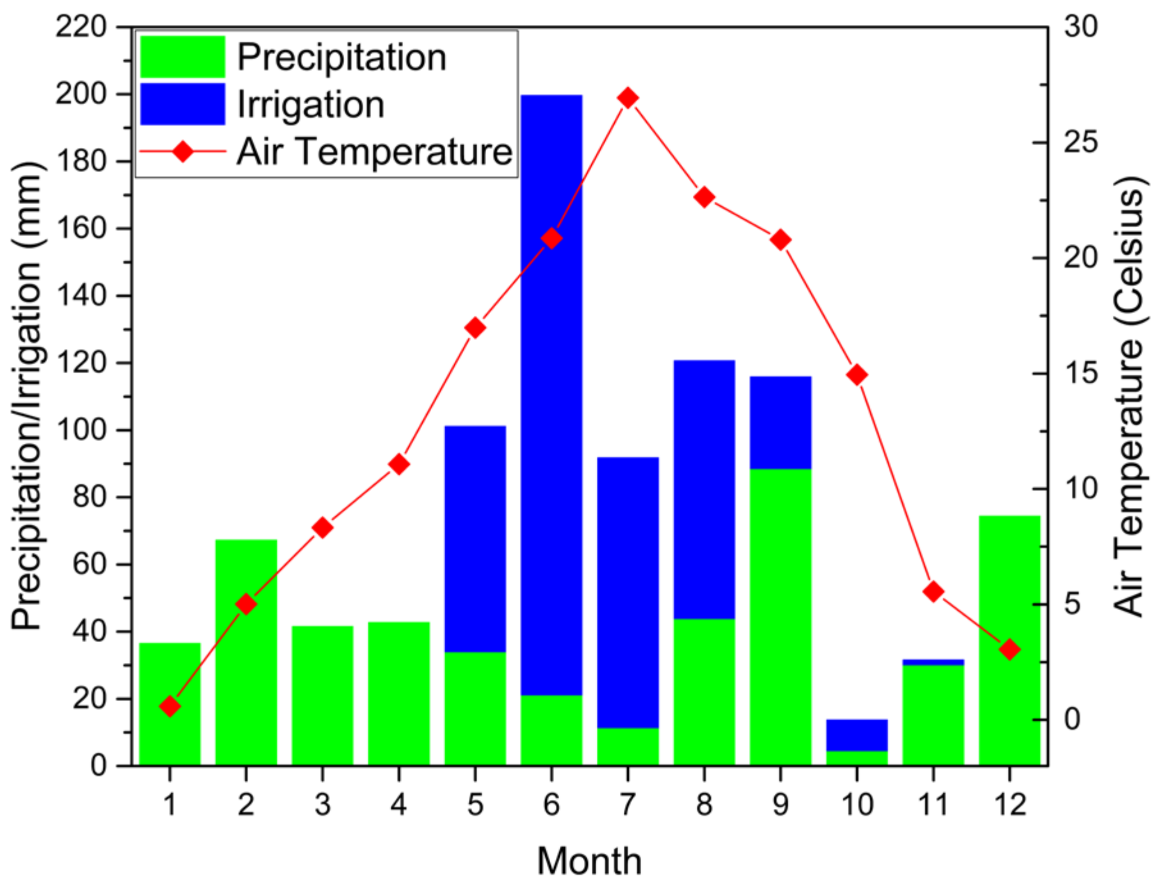

Fig. 3. Monthly averages of precipitation, irrigation, and 2-m air temperature of the Library site in 2014.

library site. The station was equipped with a Campbell Scientific CSAT3 sonic anemometer operating at 20 Hz to measure all three components of the wind speed at 2 m. Four thermocouple probes (Omega HTTC36-T-18G-6, Stamford, CT) were mounted at 0.12 m, 0.60 m, 1.50 m, and 2.50 m above the roof level to determine the local temperature profile. Temperature and relative humidity were measured at 2 m using a Campbell Scientific HMP155A probe (replaced by a Campbell Scientific CS210 humidity sensor after 09/22/2015 for the relative humidity). Kipp & Zonen CNR-1 (Delft, The Netherlands) net radiometer was mounted at 3 m above the roof level, which provided measurements of all four components of the short and longwave radiation budget. Data were recorded using a Campbell Scientific CR5000 datalogger. The weather stations (ID: WBB and MTMET) from a nearby building on campus operated by Mesowest (http://mesowest.utah.edu/) were used to fill the gaps caused by power failures and to distinguish precipitation events from irrigation events during summer nights. A Decagon SC-1 Leaf Porometer (Pullman, WA) was employed to measure the diurnal stomatal resistances of sedums and grass in the lysimeters on June 17, 2015 to provide data for corroborating observations of the lysimeters. The leaf area was estimated by sampling the branches and analyzing the images based on Easy Leaf Area (Easlon and Bloom 2014).

A weighing lysimeter of the same design with inner dimensions of 1.22 m by 0.61 m by 0.20 m was built and deployed at the Museum site. The same combination of growth media, filter fabric, and drainage mat was used as the green roof installation at the site. A mix of four sedum species was transplanted from the installed green roof at the site into the lysimeter in September, 2013. The medium depth was set as 102 mm. A Rice Lake BenchMark HE weighing scale (Rice Lake, WI, capacity: 227 kg, resolution: 0.05 kg ~ 0.06 mm, tolerance: 0.04 kg ~ 0.05 mm) and an IQ355 indicator were used to provide the weight observations of the lysimeter. Decagon (Pullman, WA) EC-5 sensors were buried at the surface and bottom layer of the lysimeter to measure soil moisture. The Campbell Scientific CR10X-PB datalogger was used to store 5-minute averages of weight measurement. A Campbell Scientific (Logan, UT) TE525 tipping bucket rain gauge was modified and placed below the lysimeter to measure the discharge from its underdrain. Another TE525 tipping bucket rain gauge was placed nearby to measure precipitation. A 12.7 mm-diameter Netafim tube was connected from the irrigation main line to the lysimeter through a hole drilled in the wall, which was used to provide the same irrigation schedule as the green roof on site. Although a plastic flow meter with a diameter of 12.6 mm and NPS threads (Adafruit, NYC) was used to measure the inflow, it did not track the irrigation amount accurately; hence, only days without irrigation events were used for analysis at this site. As irrigation was applied frequently in the middle summer and due to a power failure and datalogger malfunction in early 2014, only 117 days of data from late June to early December were available for analysis. Consequently, the museum site is primarily used to validate the ET model results based on data from the Library site.

2.3.2 Penman-Monteith Equations

The P-M equation (Monteith 1965) was adopted in this study in three forms to simulate ET rates for green roofs, as it has been proven to be reasonably applicable for a wide suite of landscapes and even green roofs (DiGiovanni et al. 2013). It is widely known in its ASCE format as follows (Allen et al. 2005):

$$RET = \frac{0.408\Delta(R_n - G) + \gamma\frac{C_n}{T+273}u_2(e_s - e_a)}{\Delta + \gamma(1 + C_d u_2)}, \tag{3}$$

where RET is reference ET (RET) corresponding to short grass (mm hr^{-1}), Δ represents the slope of the saturation-vapor-pressure temperature curve (kPa °C^{-1}), R_n is the net radiation (MJ m^{-2} hr^{-1}), G is the soil heat flux (MJ m^{-2} hr^{-1}), γ is the psychrometric constant (kPa °C^{-1}), C_n is the numerator constant (C_n=37 K mm s^3 Mg^{-1} hr^{-1}), T is the air temperature at 2 m (°C), e_s is saturation vapor pressure (kPa), e_a is actual vapor pressure (kPa), C_d is the denominator constant (s m^{-1}) (equal to 0.24 for daytime hours and 0.96 for nighttime hours), and u_2 is the mean hourly wind speed at the 2-m height (m s^{-1}).

This version of P-M equation implicitly fixes surface resistance at 70 s m^{-1} for daily estimation and 50 s m^{-1} and 200 s m^{-1} for nighttime estimation, which are often used for landscapes and green roofs (Berretta et al. 2014; DiGiovanni et al. 2013; Schneider et al. 2011; Wadzuk et al. 2013), though those parameters represent the surface resistance for the short, cool-season, well-watered reference grass. Therefore, RET values needed a correction to achieve the actual ET (AET) values for conditions different than those assumed. One correction is using a crop coefficient (K_c) to adjust the difference of PET rates between the reference grass and other species, which can be written as follows (Allen et al. 1998):

$$PET_{K_c} = K_c \times RET. \tag{4}$$

Since PET can be measured directly using lysimeters and RET rates are calculated based on meteorological data and fixed parameters from the ASCE scheme [Eq. (3)], the daily crop coefficients can be computed from Eq. (4). Monthly averages of daily crop coefficients were then generated from the observations and used to simulate hourly PET rates for different months (PET-K_c). As monthly coefficients are often difficult to obtain a priori, a yearly-average crop coefficient was used to simulate hourly PET rates (PET-K_c-Yearly) as a simplification.

Another correction is to use a water stress coefficient to adjust the difference between PET rates and AET rates due to the soil moisture content variation (Allen et al. 1998). A type of water stress coefficient, used for other green roofs (DiGiovanni et al. 2013; Stovin et al. 2013), was used in this study to convert hourly RET rates into hourly PET rates (PET-K_s) as follows:

$$PET_{K_s} = \frac{\theta_i - \theta_{wp}}{\theta_{fc} - \theta_{wp}} \times RET, \tag{5}$$

where θ_i is volumetric soil moisture content, θ_{wp} is soil wilting point (0.12 was used in this study) (DiGiovanni et al. 2013), and θ_{fc} is soil field capacity (0.35 was used). As the green roofs were all well watered during the experiment, the *PET-Ks* rates should be close to actual measurement.

Besides the FAO-56 version (Allen et al. 1998) and the ASCE version (Allen et al. 2005), the original P-M equation could also be parameterized via experiments to directly estimate AET with varying surface resistances as follows (Monteith 1965):

$$ET = \frac{\Delta(R_n - G) + \rho_a c_p \frac{(e_s - e_a)}{r_a}}{\lambda[\Delta + \Upsilon\left(1 + \frac{r_s}{r_a}\right)]}, \tag{6}$$

where λ is the latent heat of vaporization, which can be calculated as $\lambda=2.501\text{-}0.00237\times T$ (Stull 1988), but 2.45 MJ kg^{-1} was used in this study for simplicity, r_a is the aerodynamic resistance (s m^{-1}), and r_s is surface resistance (s m^{-1}).

There has been debate surrounding the value of surface resistance used in the P-M equation for reference grass (Allen et al. 2006), as it is determined by leaf area index varying across the growing stages and stomatal resistance, which is difficult to quantify and varies during the course of a day with solar radiation, leaf temperature, vapor pressure, leaf water potential, and carbon dioxide (Jarvis 1976). Little is known about the surface resistance for green roofs. As the actual PET rates are tracked by lysimeters and other meteorological data are recorded by the weather station, hourly surface resistances for each type of studied green roof can be back calculated using the P-M equation for each month [Eq. (6)] (Jones 1992; Schulze et al. 2005). The back-calculated hourly surface resistance values then could be useful for estimating ET for green roofs with similar settings and climates by using Eq. (6), and they were used in this study to estimate PET rates (*PET*-r_s) for comparison to observed values. As the hourly surface resistances for each month are also difficult to obtain a priori, the yearly-averaged surface resistance was also calculated as a simplification and used to predict PET rates (*PET*-r_s). The statistical analysis was made by OriginPro 9.1.0. One-way analysis of variance (ANOVA), Kruskal-Wallis ANOVA test and Mood's Median test were used to explore the differences in ET time traces amongst the three surface covers. Linear regression analysis with a fixed intercept at zero was used to determine the fit of PET simulations.

2.4 Results

2.4.1 ET Observations

A daily ET time series for the three green roof types studied on the Library site was determined from lysimeter measurements. The average ET rates over the one-year study period for the nonvegetated, sedum, and grass covers are 2.01±1.16, 2.52±1.79, and 2.69±1.69 mm/d, respectively. The ET rates from the three surface covers all show a unimodal temporal pattern over the course of the year with a peak in the summer, but their magnitudes tend to vary significantly from each other (Fig. 4). As the requirements of normality and equal variance for one-way analysis of variance (ANOVA) are not very well met based on the Shapiro-Wilk test and Levene's test, two nonparametric analysis methods were also conducted to test their differences. The p-values of the Kruskal-Wallis ANOVA test and Mood's Median test are 2.08E-4 and 1.53E-6, respectively, which both indicate that the populations among the three green roof lysimeters are significantly different at a significance level of 0.05.

During early summer months, ET rates generally decrease in sequence from the grass, to the sedum, and then to the medium covers. During the late summer months, sedums were observed to have higher ET rates than grass. During the winter time when plants were not active, soil evaporation tended to have higher ET rates than the plants. Mean monthly

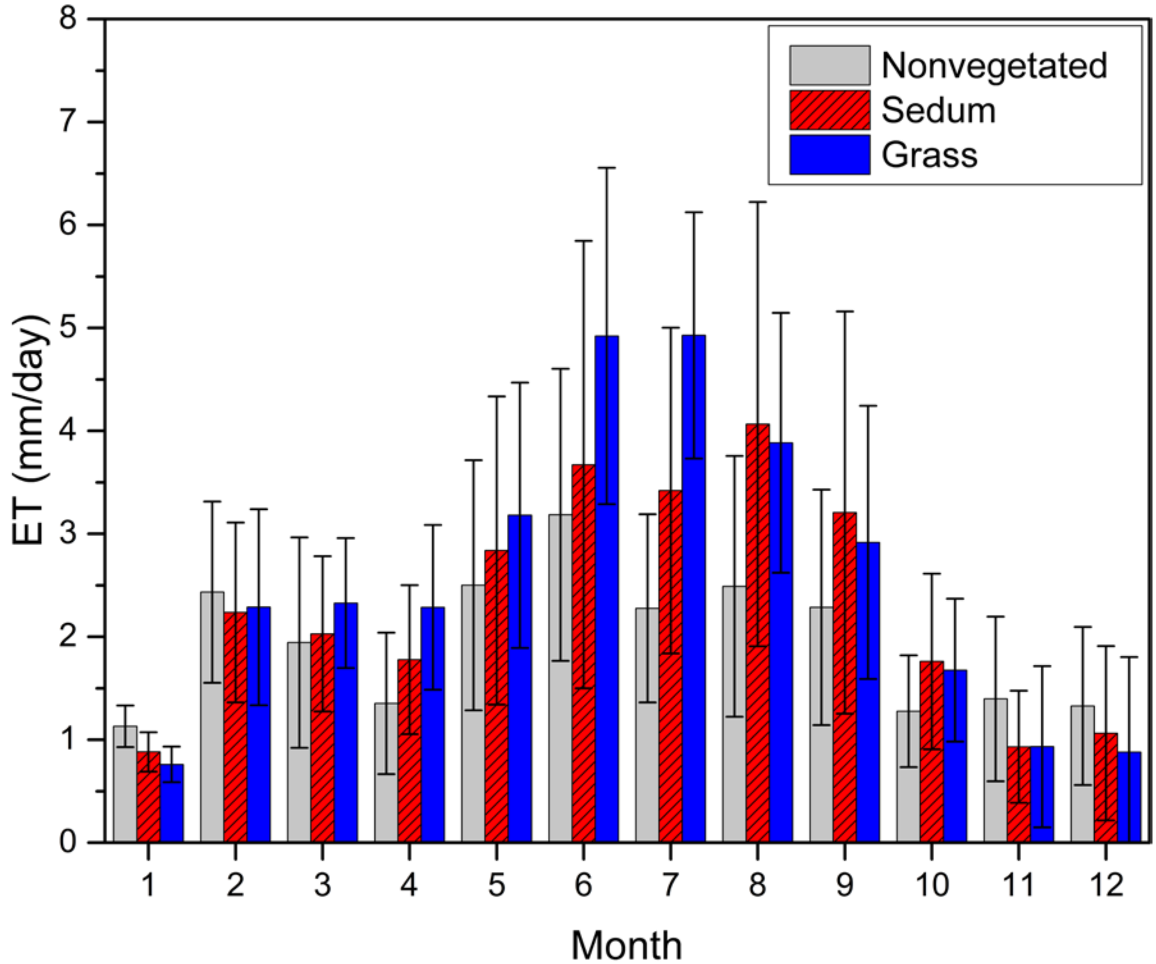

Fig. 4. Monthly averages of green roof ET rates of 2014 (error bars represent standard deviations).

ET rates were largest in June for grass (4.93 mm/d) and medium (3.18 mm/d), while the sedum had the maximum mean monthly ET rate of 4.07 mm/d later in August. Maximum daily ET rates were observed as 8.50 mm and 7.90 mm for sedums and grass in August, while that of the nonvegetated unit was observed in May (6.11 mm). The largest difference between the grass ET rates and the rates of the other two types of covers (nonvegetated and sedum) occurred in July and were 2.65 mm/d and 1.51 mm/d, respectively. The ET rates of the three green roof lysimeters tended to be close to each other during the winter months.

2.4.2 Crop Coefficients

Daily crop coefficients for the three types of green roofs were estimated by dividing the measured PET rates by the calculated RET rates each day [Eq. (4)]. Crop coefficients have similar temporal patterns as ET observations (Fig. 5). Although the three covers show different relative magnitudes across the year, their annual averages were close: 0.57±0.32 (grass), 0.57±0.36 (sedum), and 0.50±0.36 (nonvegetated).

2.4.3 Surface Resistances

The estimated surface resistances appear to vary widely among different covers, different hours, and different months (Fig. 6). Infinite values of the surface resistance correspond to zero ET conditions when the stomata close or the soil dries. Those infinite data have been removed from the plot (Fig. 6). Most nighttime values were computed as infinity, so the curves of yearly averages corresponding to nighttime were close to zero after those infinite values were removed (Fig. 6). Specifically, the nonvegetated cover had the highest surface resistance, and the sedum and then the grass covers followed accordingly; their annual averages are computed as 1707, 480, and 399 s m^{-1}, respectively. A diurnal measurement of stomatal resistance was made on June 17, 2015 for the sedum and grass in the same lysimeters. Their values were converted to surface resistance after leaf areas were measured, which were compared with the hourly averages of estimated values of June, 2014. In spite of one year’s difference, the estimated surface resistance still showed a relatively good match to the observed values for daytime hours (Fig. 6).

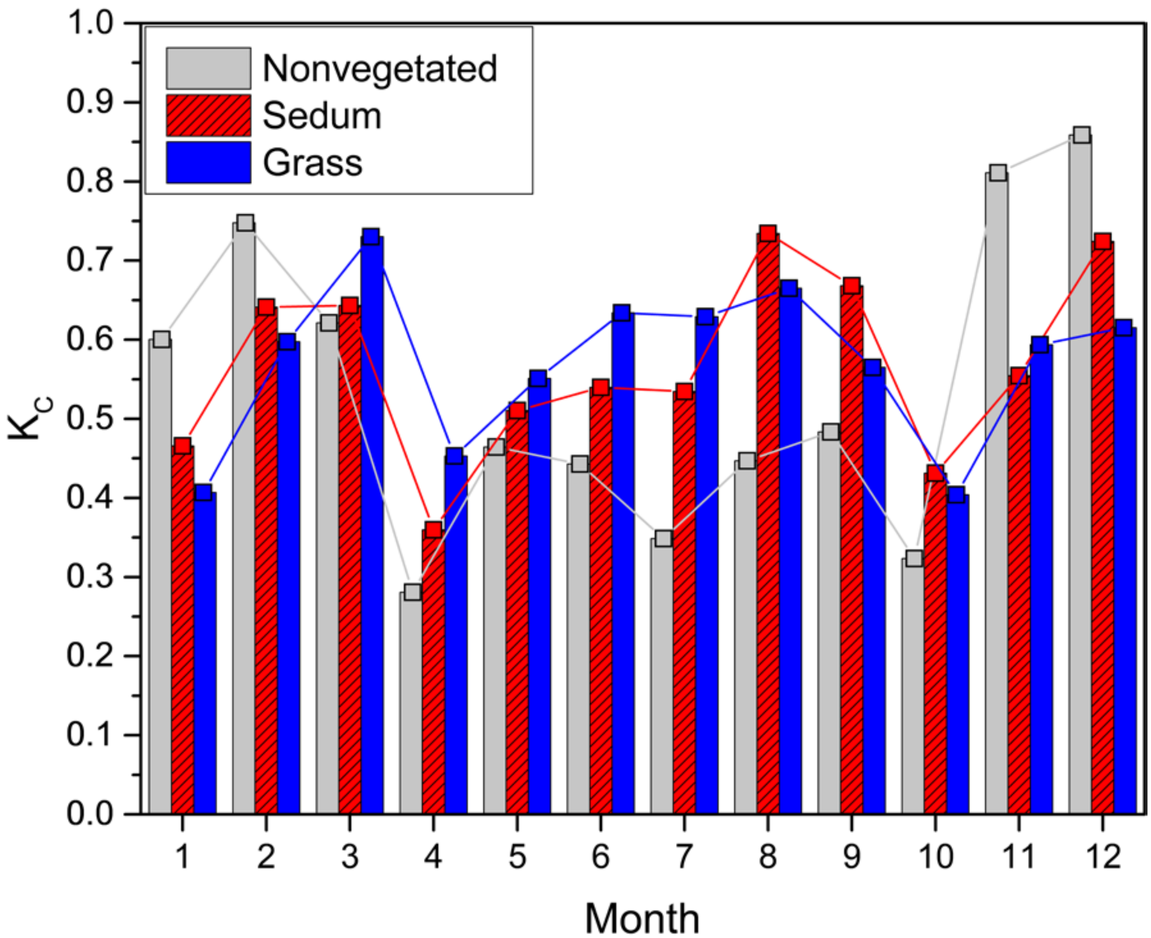

Fig. 5. Monthly crop coefficients for the three surface covers in 2014.

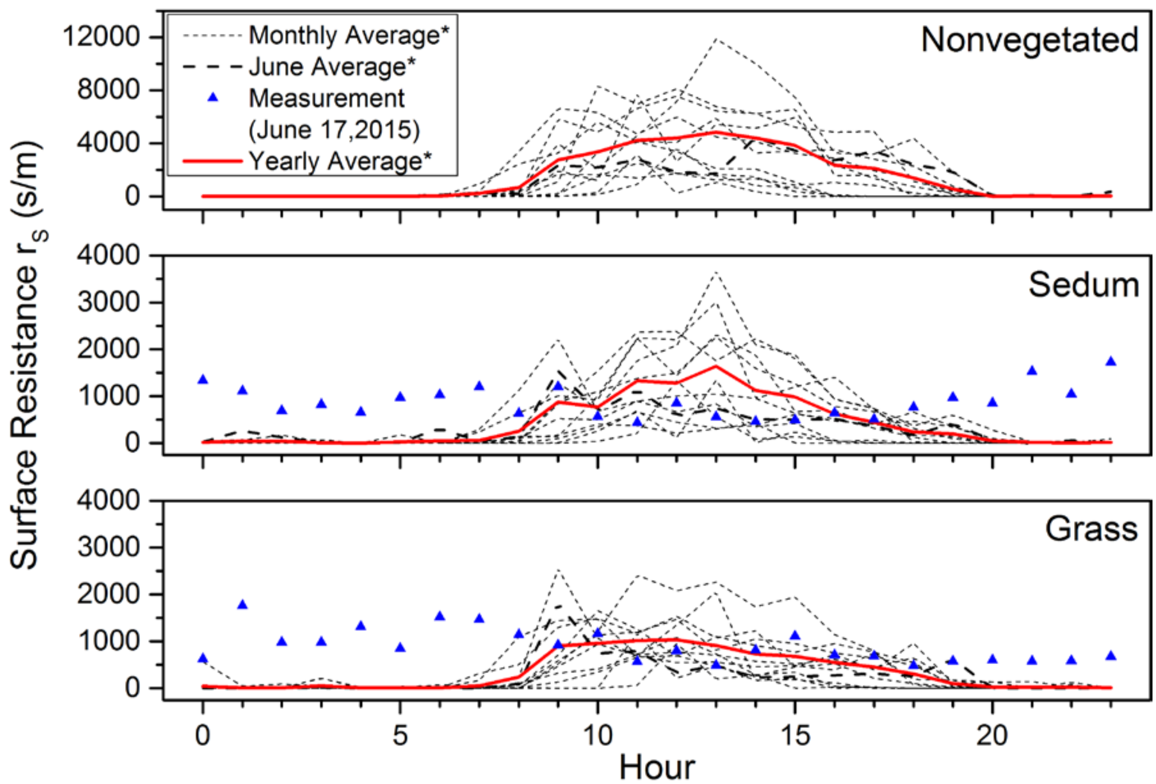

Fig. 6. Hourly averages of surface resistances of three types of green roofs. *Note: the infinite values calculated were removed; most of the nighttime values are infinity.

2.4.4 ET Simulations

The three variations of the P-M equation described above were used to simulate hourly ET rates based on meteorological measurements. The *goodness of fit* was determined by the comparison of the daily ET estimates and the daily ET observations for nonvegetated (Fig. 7), sedum (Fig. 8), and grass (Fig. 9) on the library site. The *PET-K_c* method tends to overestimate PET rates for the nonvegetated and sedum covers, while the *PET-r_s* method tends to slightly overestimate PET rates only for the sedum. They both show a good fit with observation for the grass. The *PET-K_s* method largely underestimates PET rates for all cases. The coefficients of determination (R square) show that the *PET-K_c* method has the best fit with the measured values, but the *PET-r_s* method also achieves a comparatively good fit which generates R square very close to the *PET-K_c* method (Table 1). However, the *PET-K_s* method seems to generate a less competitive fit compared with the other two. The three methods were also tested for the sedums at the Museum site. There, all three methods tended to overestimate PET rates (Fig. 10), and their R square values are all lower than the Library site (Table 1). Notably, both the *PET-r_s-yearly* and *PET-K_c-yearly* methods generate R square values close to the *PET-r_s* and the *PET-K_c* methods, which use more detailed monthly parameters (Table 1).

2.5 Discussion

2.5.1 ET Observations

The yearly-averaged ET rates observed in this study are higher than most rates observed in other studies in wetter climates. For example, 1.68 mm/d for sedums and 1.06 mm/d for medium with unlimited water supply (Rezaei 2005), 1.71 mm/d from sedums (Marasco et

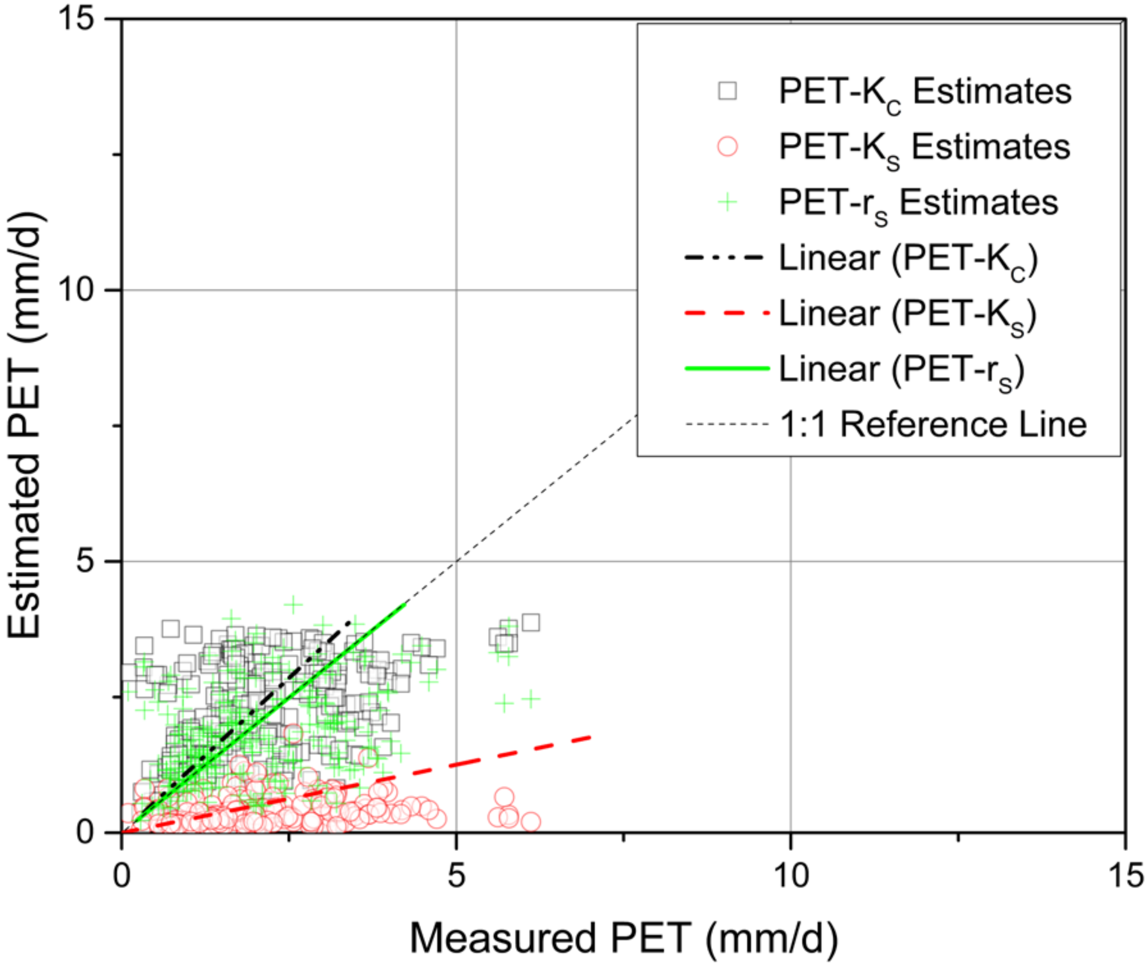

Fig. 7. PET estimates using the three methods plotted against PET observations for the nonvegetated green roof at the Library site.

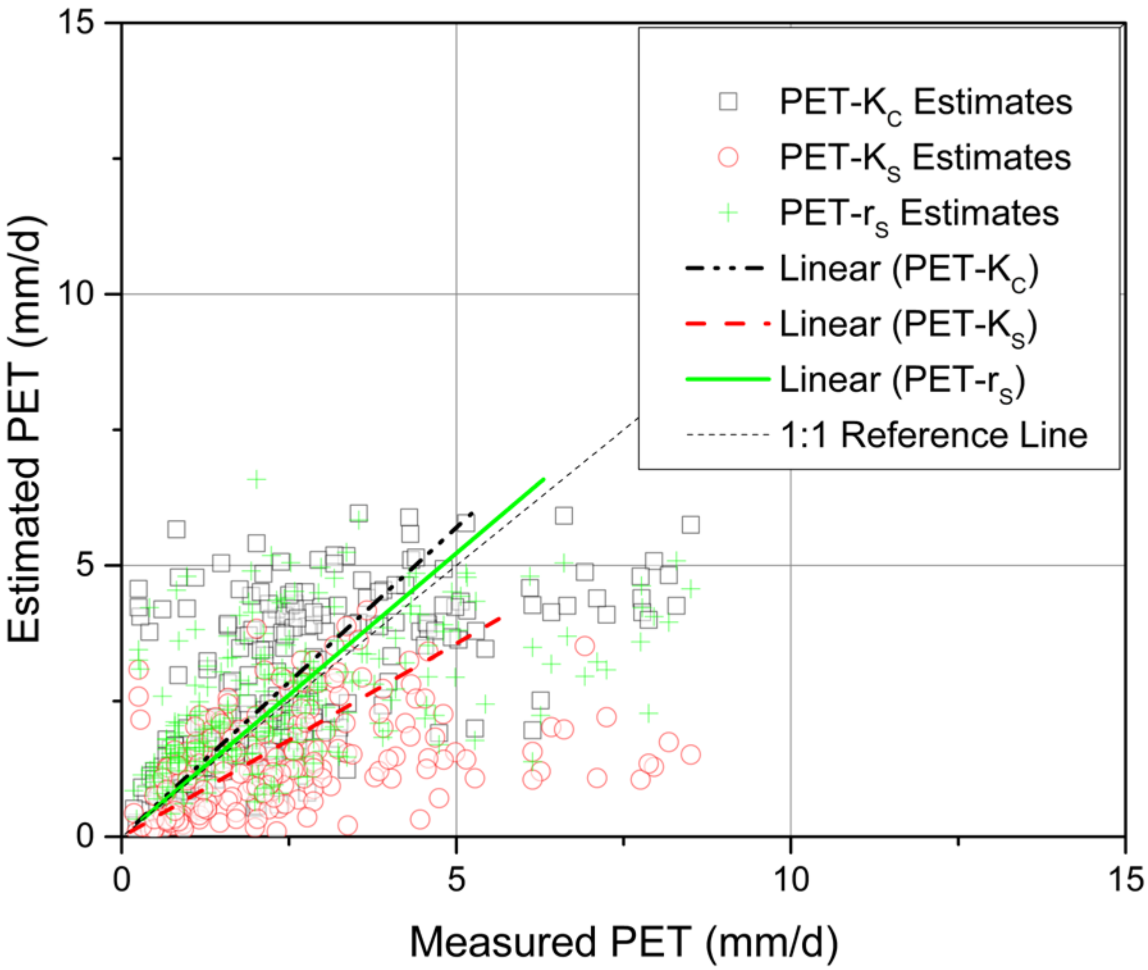

Fig. 8. PET estimates using the three methods plotted against PET observations for the sedum-covered green roof at the Library site.

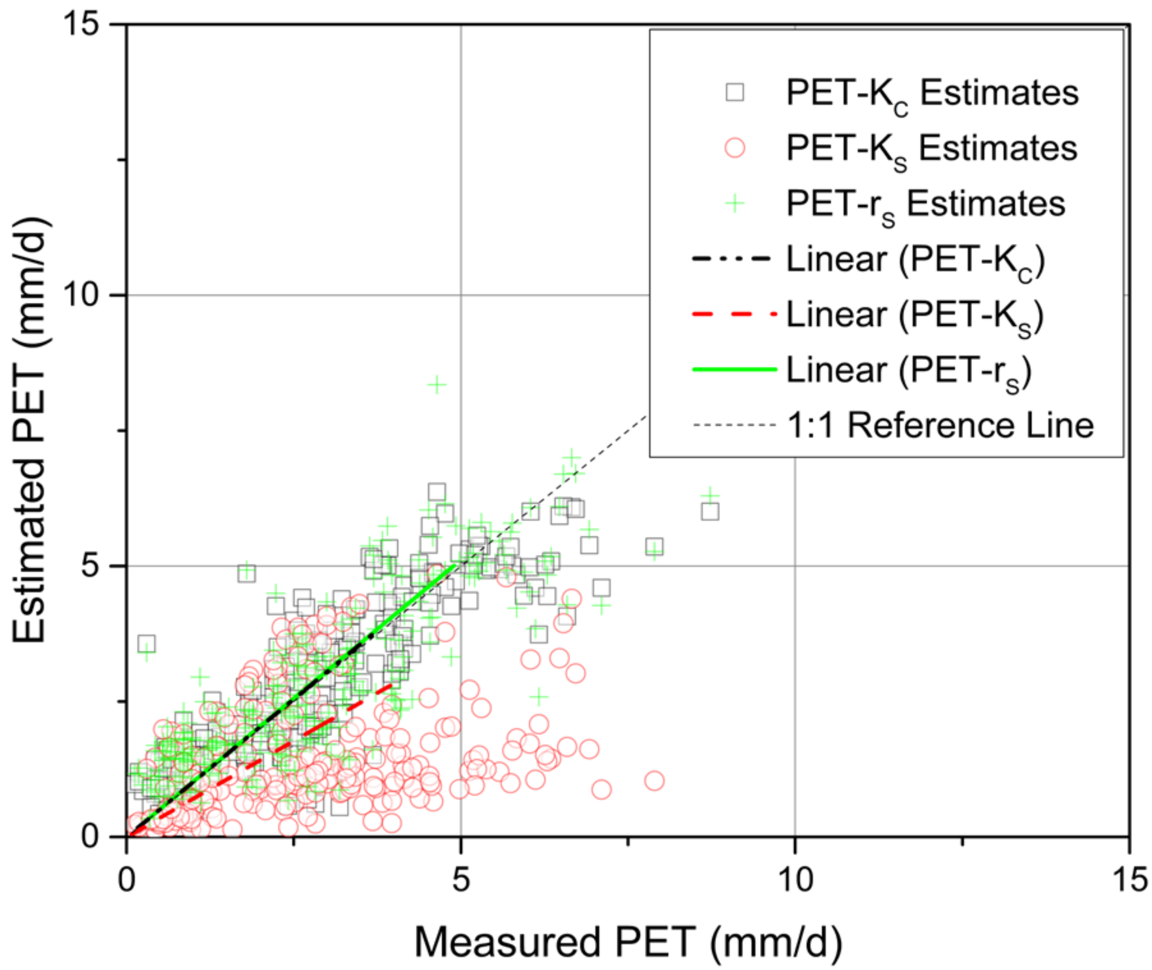

Fig. 9. PET estimates using the three methods plotted against PET observations for the grass-covered green roof at the Library site.

Table 1. Statistical performance measures of the three types of P-M equations for PET estimation.

	Library Site						Museum Site	
	Medium		Sedum		Grass		Sedum	
Linear Fitting	R^2	RMSE (mm/d)	R^2	RMSE	R^2	RMSE	R^2	RMSE
PET-K_C	0.79	1.07	0.78	1.47	0.93	0.89	0.59	1.84
PET-K_S	0.60	1.50	0.63	1.77	0.63	1.94	0.59	1.71
PET-r_S	0.75	1.17	0.74	1.58	0.91	1.00	0.58	1.87
PET-r_S-yearly	0.70	1.28	0.74	1.59	0.90	1.04	0.58	1.86
PET-K_C-yearly	0.75	1.17	0.75	1.54	0.91	0.98	0.61	1.80

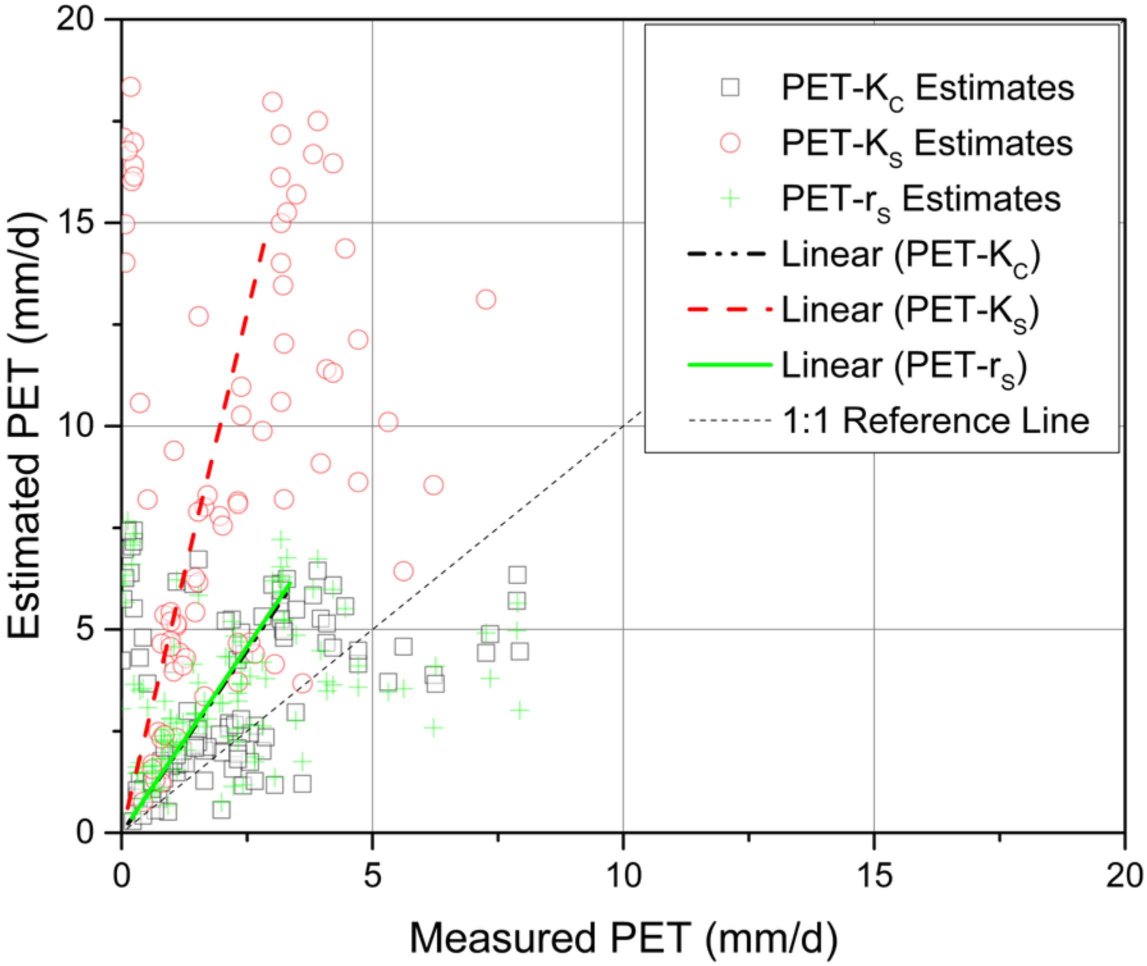

Fig. 10. PET estimates using the three methods plotted against PET observations for the sedum-covered green roof at the Museum site.

al., 2014), 2.27 mm/d from sedums (DiGiovanni et al. 2013), or 2.21 mm/d from sedums (Voyde et al. 2010). The averages of ET rates for sedums and grass from April to November are 2.74 and 2.94 mm/d (Fig. 4), which, however, are lower than the observation (3-4 mm/d) in a humid climate (Wadzuk et al. 2013). The summer averages of ET rates of sedums and grass from May to September are 3.52 and 3.91 mm/d, which are slightly higher than 3.4 mm/d observed in a wetter climate when moisture is adequate (Poë et al. 2015). The largest monthly averages of the sedum and grass are 4.07 and 4.92 mm/d, which are higher than the 1.24 mm/d from sedums in a semihumid climate (Sherrard and Jacobs 2012), but lower than 4.94 mm/d in a more humid climate (Marasco et al. 2014). The lowest monthly ET rates of sedums and grass in the winter are observed as 0.88 and 0.76 mm/d in this study, which are higher than 0.52 mm/d (Sherrard and Jacobs 2012) and 0.24-0.72 mm/d in wetter climates (Marasco et al. 2014). It appears that sedums and grass have higher ET rates than most studies conducted in more humid climates, even when the moisture is adequate as well. Not surprisingly, this indicates that climate can play an important role in determining the green roof ET rates as long as the moisture requirement can be met.

A principal component analysis was conducted for the normalized meteorological variables (air temperature, relative humidity, wind speed, air pressure, incoming solar radiation, and precipitation). Relative humidity, solar radiation, and air temperature, in descending importance, explain 81.22% of the variance. A partial least squares analysis, conducted based on the normalized daily ET measurements and the same set of meteorological variables, further confirms that the ET rates of the three roof covers are most sensitive to the same three variables yet in a different order. The variable importance in projection (VIP) scores of air temperature, solar radiation, and relative humidity are 1.46, 1.38, and 1.05, respectively, while VIP scores of other variables are lower than 0.8.

Although the daily time series of ET rates appear significantly different between sedum and grass, from the statistical analysis, the annual averages of these two roof covers are close to each other, with a difference of 0.17 mm/d (7%). The largest difference between them happened in June and July when the monthly averages of grass ET rates were 1.25 mm/d (34%) and 1.51 mm/d (44%) higher than sedums (Fig. 4). Their differences were lower during winter months, when average sedum ET rates are higher than grass ET rates by 0.18 mm/d (17%) at most. Sedums and grass showed significant difference in ET rates in the summer, but their ET rates were close on an annual basis. Their reversed relationship during winter may reflect that sedums have a higher cold tolerance than grass (Monterusso et al. 2005).

Evaporation from the nonvegetated cover tends to be lower than plants during most of the warm months, but becomes higher during the winter months. The largest average difference between sedum ET rates and nonvegetated evaporation rates happened in August, when the former is 1.58 mm/d (63%) higher than the latter, which is close to the finding of 58% under well-watered condition in a wetter climate (Rezaei 2005). The largest average difference between grass ET rates and nonvegetated evaporation rates happened in July when the former is 2.65 mm/d (117%) higher than the latter. During winter months, the nonvegetated evaporation rates, however, can be as much as 0.46 mm/d (33%) and 0.45 mm/d (34%) higher than sedum ET rates in November and grass ET rates in December, respectively. The reverse relationships between the summer and the winter may indicate that plants on the green roof could transpire much more than the nonvegetated roof for growth when plants are active and the moisture is unlimited, while plants can store water via interception and around roots when moisture is not comparable and the environment is not favorable for growth. The nonvegetated case, however, does not have such controls. For example, higher ET rates are reflective of a larger potential to restore a green roof's water storage capacity to capture more water from the next precipitation event (Stovin et al. 2013). It appears from this study that green roofs tend to have higher capacity to absorb stormwater in the summer, while the nonvegetated roof tends to have better potential to capture snowmelt water in the winter.

2.5.2 ET Simulations

The *PET-K_c* method achieved the best fit for almost all the cases (Table 1). The annual crop coefficients calculated by this study are very close to some other studies in wetter climates. For example, 0.53 was reported for a well-watered sedum canopy measured by a weighing lysimeter of the same area as the one used in this study (Sherrard and Jacobs 2012). An annual average of 0.51 was reported for different sedums estimated based on moisture contents of green roof platforms (1.31 m^2) larger than the area of this study (Starry 2013). Similarly, 0.35-0.52 was observed for well-watered sedums calculated based on an energy balance approach for a 1000 m^2 green roof (Lazzarin et al. 2005). However, the crop coefficients calculated in this study are lower than some studies in wetter climates. For example, Voyde (2011) reported 0.85 for well-watered sedum measured using 0.072 m^2 weighing trays, Schneider (2011) observed 1.0-1.7 for sedums measured with a 0.21 m^2 weighing lysimeter, and Rezaei (2005) observed 1.35 on average for sedums estimated from 0.56 m^2 indoor greenhouse plots. Although climate may affect the amounts of green roof ET, it appears that climate may not greatly affect the magnitude of crop coefficients for a green roof, at least not for sedums, as similar crop coefficients were calculated in different climates. Different experimental settings may exert a significant influence on determining the crop coefficient of green roofs. The lysimeters of small sizes with greater boundary effects and disturbances or a greenhouse environment without diurnal shifts, tend to generate larger crop coefficients. Hence, PET rates determined by such designs are more likely to reach the RET level that a reference crop can reach within that climate.

The *PET-K_s* method achieves a consistent series of coefficients of determination of around 0.6 (Table 1), which, however, are lowest for any case on the Library site, while close to other methods on the Museum site. The *PET-K_s* method underestimates the ET rates for the Library site (Fig. 7-Fig. 9), but overestimates the ET rates at the Museum site (Fig. 10). This may be because this method is highly sensitive to the input parameters (i.e., wilting point and field capacity (DiGiovanni et al. 2013), which are difficult to quantify.

The *PET-r_s* method compares well with the *PET-K_c* method at both sites (Table 1). To compare with other studies, the annual averages of surface resistances for sedums and grass calculated in this study were converted to the corresponding stomatal resistances for the middle growing stage (as leaf areas keep varying), which are 679 and 372 s m^{-1}, respectively. The calculated stomatal resistance of sedums is a little lower than the value of 750 s m^{-1} reported by Schneider (2011), which was not under well-watered conditions. However, the present results are much higher than the value of 250 s m^{-1} found by Voyde (2011). It has been recognized that sedums could have crassulacean acid metabolism (CAM) mechanisms, which have the ability to close stomata during the daytime to save carbon dioxide and water, and open stomata to resume metabolism during the cooler and wetter nights (Dvorak and Volder 2010; Jones 1992; VanWoert et al. 2005b; Voyde et al. 2010). However, ET rates measured at night were small, and sedums in this study did not show strong CAM mechanisms even when irrigation only happened at night, which is consistent with other observations (Starry 2013; Voyde 2011). This may be because the sedums were well watered in this study, which causes them to not switch from C_3 mode, with which plants usually open stomata during the daytime (Jones 1992), to CAM mode when moisture is adequate (Gravatt and Martin 1992; Kluge 1977; Starry et al. 2014). Accordingly, both plants' surface resistance became higher at night than during the day from either calculation (infinite surface resistances were removed from Fig. 6) or measurement (Fig. 6). But it appears that their stomata may still remain slightly open at night (Fig. 6), as Blue Grama Grass belongs to the C_4 plants (Waller and Lewis 1979; Williams 1974), which even have been shown to transpire 3-12% of their daily ET during the night when irrigated (Wang and Dickinson 2012), in spite of higher nighttime surface resistances (Schulze et al. 2005).

The nonvegetated medium has much higher surface resistances than both plants during most times (Fig. 6), as plants have a more efficient water uptake mechanism by exploiting moisture via roots, transporting it via xylems instead of dry top soil, and releasing it via stomata. The drying of the top soil layer could lead to a several-thousand (seconds per meter) magnitude surface resistance (Daamen and Simmonds 1996; Mahfouf and Noilhan 1991; van de Griend and Owe 1994). Thus, the nonvegetated green roof in this study had considerably higher surface resistances but a lower ET rate than plants. The sedum cover generally tended to have a lower ET rate than the grass cover, especially when they were active during the summer (Fig. 4), as the former has a stricter water use strategy and accordingly higher surface resistances than the latter (Fig. 6).

To investigate the availability of using more simplified parameters for a more general use, yearly averages of monthly crop coefficients (the *PET-*K_c*-yearly* method) and of monthly surface resistances (the *PET-*r_s*-yearly* method) were tested. Both methods surprisingly achieve comparably good fits with measurements, almost as well as their counterparts using the monthly parameters for both sites (Table 1). This indicates that using one single value of crop coefficient to convert RET, or using one single value of surface resistance in the P-M equation may achieve reasonable accuracy compared with using the monthly values for annual PET estimation. Furthermore, although crop coefficients of the three roof covers vary across the year (Fig. 5), their annual averages are close.

2.5.3 Water Budgets

As Utah overall has a semiarid climate and water is one of the most precious natural resources, the water demands of GI also need to be considered before implementing them in this climate. The annual water budgets of the three studied green roof covers were estimated based on measurements (Fig. 11). The gaps of ET measurements due to power outage or large precipitation were filled by the estimates by the PET-r_s method. The gaps of ET measurements due to power outage or large precipitation were filled by the estimates by the PET-r_s method. The gaps of drainage measurements were filled by a simple linear regression relationship established based on observations. It appears that ET dominates the outflow budget for all three roof covers. Higher ET fractions associated with plants compared to nonvegetated medium emphasizes the role of transpiration on green roofs for stormwater management. However, the total annual irrigation amounts were close to the annual precipitation amounts, which indicates the tradeoff between higher stormwater processing capacity by grass and the higher water demands in the dry climates.

2.5.4 Other Considerations

Moisture and temperature were also measured at the surface and bottom of the three lysimeters and the green roof garden at the Library site (Fig. 12). The moisture content at the bottom of each lysimeter is higher than the surface during most months. The annual average differences of soil moisture content fraction from the bottom to the surface of each lysimeter were 0.10, 0.17, and 0.14, corresponding to the nonvegetated, sedum, and grass covers. It reflects the fact that the plants can effectively deplete the storage via ET compared to the nonvegetated cover. As sedums have the highest difference, this may infer

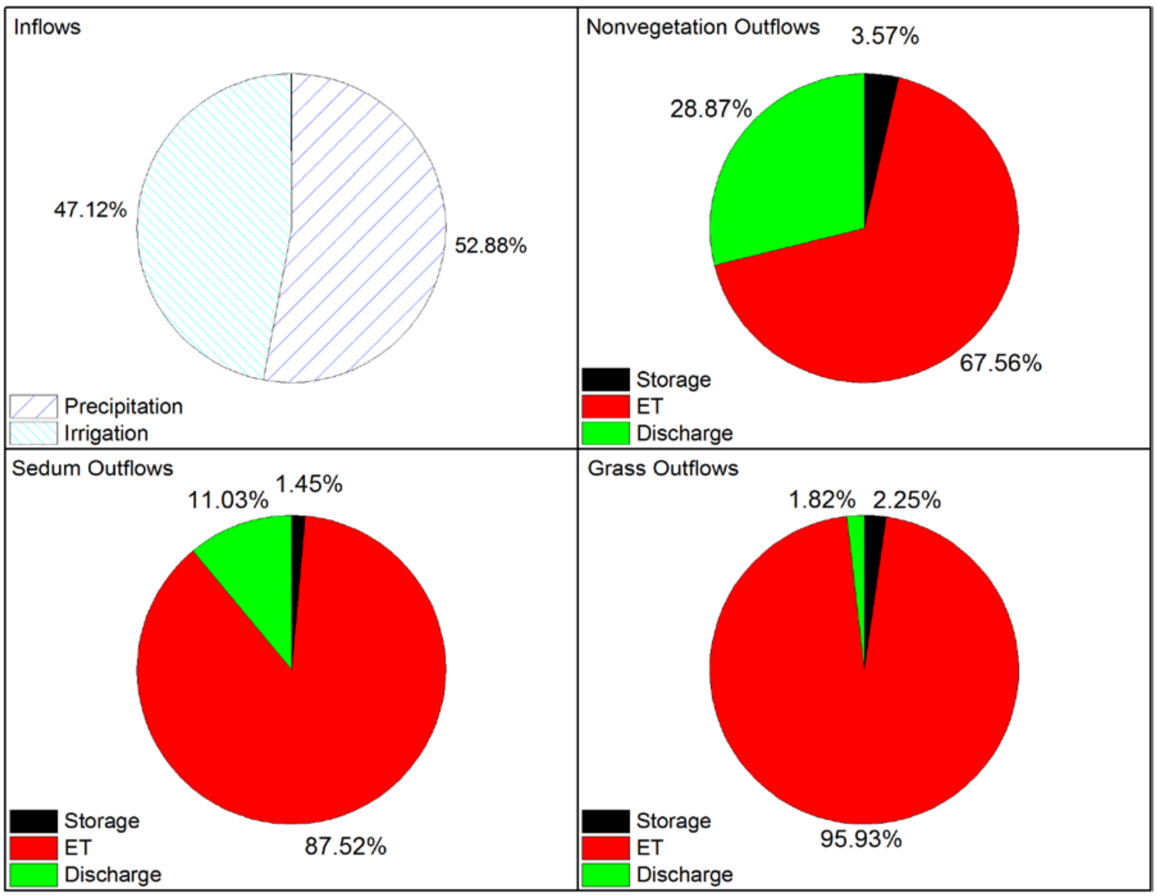

Fig. 11. Annual water budgets of three types of green roofs of 2014.

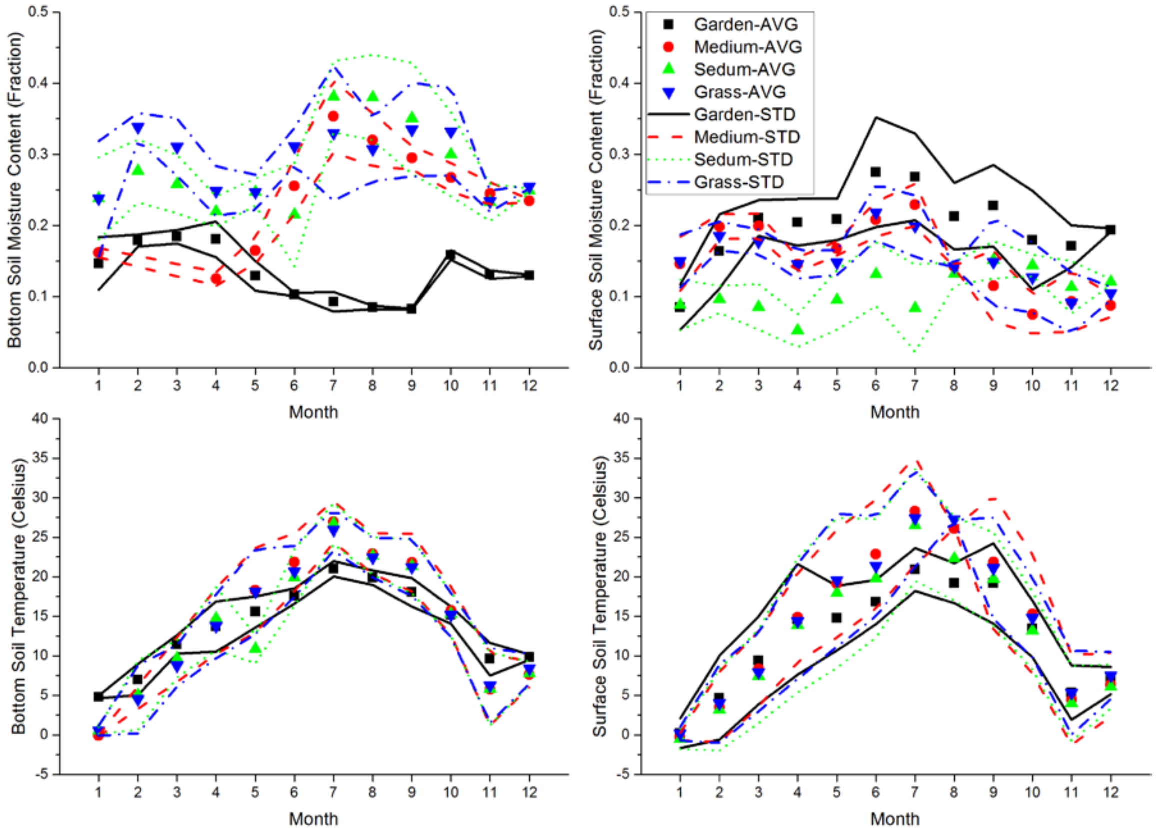

Fig. 12. Monthly averages of soil moisture and temperature at the surface and bottom of three lysimeters and the green roof garden. Data are missing for the medium (nonvegetated) lysimeter at the bottom level in February and March due to the broken sensors. The point symbols represent monthly averages, while the lines represent the monthly standard deviations.

that sedums make use of the surface moisture mostly due to its shallower root systems compared to grass. Although it is intuitive to have a deeper medium to store water for plants, especially in dry climates, this finding may indicate this may not be necessary if only sedums are to be planted on a green roof, even in a semiarid climate. However, compared to the lysimeter data, the green roof garden had much higher surface moisture content than at the bottom during most months of the year. As the moisture content should have a decreasing gradient from the surface to the bottom due to ET, this may indicate most irrigation and precipitation did not reach the lower part of the green roof before being lost via ET. This may further support the idea that a large medium depth on green roofs may not be necessary for growing grass (even in a semiarid climate), as most incoming moisture will be lost via ET near the surface on short time scales.

The moisture content at the bottom level of the lysimeters is larger than that in the green roof garden, where the moisture sensor was buried deeper. This may have been a result of ET not consuming the incoming precipitation and irrigation in the lysimeters. On the other hand, plants grown in the green roof garden with more horizontal flows and a more diverse plant community, including shrubs with deeper roots, have higher overall ET capacity than the lysimeters. Also, the surface moisture content of the green roof garden, on the contrary, is higher than that of lysimeters during most months. This may be due to the fact that the corresponding sensor to measure the garden's surface moisture was buried at a bare ground location in the garden without nearby roots to tap the moisture. On the other hand, a better ecosystem in the green roof garden with more interflows and shading by tall shrubs saves more moisture at the surface than the lysimeter environment. But, overall, the high moisture content at the green roof garden surface and the high moisture content at the lysimeter bottom indicate that both the garden and the lysimeters were well watered.

In addition, bottom medium temperatures of the green roof garden, grass lysimeter, and sedum lysimeter (in a reducing order) are higher than the bottom temperature of the nonvegetated lysimeter during most summer months. The opposite occurred during most winter months (Fig. 12). This is consistent with other findings about green roofs' energy benefits in terms of summer cooling and winter heating (Castleton et al. 2010). The green roof garden showed a similar temperature pattern compared to the nonvegetated lysimeter on the surface level. However, the sedum and grass lysimeters have lower and higher surface temperatures compared to the nonvegetated lysimeter for all months. Further studies are needed to explore this relationship.

2.5 Conclusion

To study the ET behavior and general hydrologic performance of green roofs in a semiarid mountain west U.S. climate and to explore a better way to model green roof ET, an experiment using weighing lysimeter units was conducted in Salt Lake City, Utah in 2014. The results confirmed the expectation that that green roof ET rates would vary significantly between different roof covers and in different months. The annual averages of ET rates of the studied grass, sedums, and nonvegetated covers under the well-watered condition in this climate were 2.69±1.69, 2.52±1.79, and 2.01±1.16 mm d^{-1}, respectively, which are higher than other studies in wetter climates. Three methods based on the P-M equation were tested to simulate the ET observations at two sites. Both the *PET-*K_c and *PET-*r_s methods achieved overall good fits for both sites. The yearly averages of crop coefficients for the nonvegetated, sedum, and grass covers were calculated as 0.50±0.36, 0.57±0.36, and 0.57±0.32, respectively; while averages over the study period of surface resistances were calculated as 1707, 480, and 399 s m^{-1}, accordingly. The sedum coefficients are close to other studies in wetter climates. Using the more simplified yearly-constant parameters (the *PET-*K_c*-yearly* and *PET-*r_s*-yearly* methods) to predict ET rates was shown to achieve an accuracy similar to the more detailed monthly parameters. The estimated water budgets indicate that ET is by far the largest outflow contributor for all three roof covers. In fact, ET represents more than 88% of water outflow for sedums and grass. Hence, they might not be able to sustain themselves healthily without irrigation in this climate.

CHAPTER 3

IMPROVING EVAPOTRANSPIRATION MECHANISMS IN THE U.S. ENVIRONMENTAL PROTECTION AGENCY'S STORM WATER MANAGEMENT MODEL

3.1 Introduction

Over the past two decades, stormwater green infrastructure (GI) has emerged as a leading recommended practice in new development and redevelopment. Through added water storage and vegetation facilitating infiltration and evapotranspiration (ET), GI can reduce and delay stormwater runoff in urban areas. The runoff reduction capacity of GI types has been summarized in various stormwater design guidelines and research studies (Hirschman D 2008). Bioretention, for example, has been noted to reduce discharge volume by 70% in the climate of the Midwest United States (U.S.) (Culbertson and Hutchinson 2004). In another study, simulations show that 38% runoff volume reduction from a watershed could be achieved by implementing bioretention on 3.9% of the impervious areas (Abi Aad et al. 2010). Of the volume reduction, a significant amount (20-50%) exits the bioretention unit by exfiltration and ET (Li et al. 2009). Green roof, another common type of GI, is not designed with the same storage capacity as bioretention, but can reduce storm peak discharges from rooftops by up to as much as 70% (Alfredo et al. 2010). A study in Auckland, New Zealand shows green roofs with a similar soil depth in a similar climate could retain a median of 82% of rainfall volume per rain event, and reduce a median of 93% peak flow (Voyde et al. 2010). More broadly, GI can also provide other ecosystem services like restoring the predevelopment runoff volumes and peak flow rates (Burian and Pomeroy 2010; Burns et al. 2012; DeBusk et al. 2011; Olszewski and Davis 2013; Petrucci et al. 2013), recharging groundwater (Dussaillant et al. 2004; Endreny and Collins 2009; Hamel et al. 2013; Stout et al. 2015), improving water quality (Brown et al. 2009; Davis et al. 2006; Li and Davis 2009), and cooling surfaces and near surface atmosphere (Coutts et al. 2013; Kumar and Kaushik 2005; Nakayama and Hashimoto 2011; Scherba et al. 2011; USEPA 2003; Wong et al. 2003).

Stormwater models have been developed to provide runoff estimates (James and Dymond 2012; Jia et al. 2002; Xiao et al. 2007; Young et al. 2011), but accurately modeling other components of the water balance, like infiltration and ET, has been less emphasized (Fletcher et al. 2013). However, a clear need has developed recently for stormwater models to better estimate GI effects on (1) runoff volume reduction, (2) groundwater recharge, and (3) ET and its connections to urban climate and energy consumption. In addition, most stormwater models are not equipped with the capacity to model spatial and temporal resolution of physically based processes in pervious areas and GI, such as ET, variable source areas, or macropore flow (Elliott and Trowsdale 2007; Fletcher et al. 2013). For example, two commonly used urban drainage infrastructure models, the U.S. Environmental Protection Agency (EPA) Storm Water Management Model (SWMM) (http://www2.epa.gov/water-research/storm-water-management-model-swmm, accessed 08/23/2015) and the System for Urban Stormwater Treatment and Analysis IntegratioN Model (SUSTAIN) (http://www2.epa.gov/water-research/system-urban-stormwater-treatment-and-analysis-integration-sustain, accessed 08/23/2015), only model daily ET by user input and assume a homogeneous potential ET (PET) distribution across the modeled watershed. On the other hand, urban hydrologic models can incorporate sophisticated processes for the water cycle, but they do not provide high fidelity representations of stormwater, water supply, and wastewater infrastructures (Dupont et al. 2006; Grimmond et al. 1986; Järvi et al. 2011; Lemonsu et al. 2007; Mitchell et al. 2008; Mitchell et al. 2001). Currently, there is a great need to provide an enhanced modeling capacity of ET to evaluate the hydrologic performance of GI and pervious areas in urban drainage infrastructure models.

The objectives of this paper are to demonstrate the incorporation of a well-established ET scheme into a widely used stormwater model, to test the modified model to validate its performance, and to apply the model to show the impact of inaccurate spatial and temporal representation of ET on stormwater runoff and water budget simulations. The basis of this paper is a new set of ET modules incorporated into an update of EPA SWMM to provide a strengthened ET modeling capacity with higher spatial and temporal resolutions for pervious landscapes and GI. Specifically, a Penman Monteith (P-M) ET framework (Allen et al. 1998) was adopted to estimate PET and was integrated with SWMM to estimate actual ET (AET). A case study of five hypothetical catchments in the environmental contexts of Salt Lake City, Utah was used to evaluate and demonstrate the updated model. The new model was applied to understand the importance of ET for the urban water budget and stormwater management.

3.2 Methods

3.2.1 Modifications to SWMM

The water budget with regards to urban landscapes and GI can be described as follows:

$$P + IR = Q + I + ET + \Delta S, \tag{7}$$

where P is precipitation, IR is irrigation, Q is surface runoff, I is percolation, ET is evapotranspiration, and ΔS is the change of the water storage in the water body, soil, or plants. As indoor water uses are not related to the water demand of GI and they have similar inflow and outflow amounts, they were excluded from the water balance here. As SWMM can simulate most of the components except irrigation in the above equation and it had a strong capacity in modeling precipitation-runoff-routing processes (Elliott and Trowsdale 2007) and urban drainage infrastructure components, EPA SWMM 5.0.022 was used as the platform in this study to model the water budget.

SWMM is limited to daily (or larger time increment) PET inputs with a hypothetically uniform distribution across the entire model domain (i.e., all subcatchments must use the single provided PET rate). Further, the plants' response to the soil moisture variation, reflected by AET rates, is ignored by SWMM. To better represent actual conditions, SWMM was reprogrammed in this study in three ways. First, changes were made to permit subdaily PET time series due to varied weather conditions to be loaded through an external text file. Second, modifications enabled up to six types of PET time series on behalf of various land covers to be imported via unique columns in the same text file. Lastly, a water stress coefficient was added to calculate AET by adjusting PET rates based on the soil moisture balance at each time step.

Six types of PET inputs considered in this study include water evaporation and PET of bioretention units, green roofs, residential landscapes, deciduous trees, and coniferous trees. The water evaporation was designed here to reflect the evaporation from ponding and canopy interception. PET from bioretention systems and green roofs was modeled to extract water from soil layers and storage layers/drainage mats, as represented in SWMM. Besides the water surface and GI, SWMM uses the aquifer module underneath the pervious areas to represent the soil layers of the landscapes and other green spaces. Only the unsaturated zone in the aquifer module was considered in this study, which is acceptable when the water table is deep, as is the case in the study area used in this paper. Notably, landscape types can vary across a single catchment; thus in the updated SWMM the pervious areas were classified into three types of land covers: vegetated landscapes, deciduous trees, and coniferous trees. As these three landscape types still need to share the same aquifer object within each subcatchment in SWMM, an area-weighted spatial average of three PET rates corresponding to these three classes was calculated and used as the overall PET for the pervious area within each subcatchment, although the three PET time series were input into SWMM separately.

The water stress coefficient was introduced into SWMM to simulate the plant response to the soil moisture variation. Besides weather factors which were taken into account during the PET estimation, soil moisture can also greatly affect the plants' stomatal control and thereby their ET rates, especially during dry spells. SWMM adopts the concept of the bucket model; however, this completely subtracts the absolute ET amounts from the storage without consideration of the influence of soil moisture. Therefore, the concept of the water stress coefficient was taken from the Penman-Monteith ET scheme (Allen et al. 1998) into SWMM to account for the plants' regulation of water use by adjusting PET rates from soil layers of bioretention units, green roofs, and landscapes to achieve AET rates. SWMM is able to provide the updated moisture balance of each subsurface layer at each time step, which makes it possible to calculate the moisture residual and the water stress coefficient. The specifics about the calculation procedures are described in the following section.

3.2.2 FAO-56 Penman Monteith ET Scheme

For this study, the ET estimation scheme of the Food and Agriculture Organization of the United Nations' Irrigation and Drainage Paper No. 56 (FAO-56) (Allen et al. 1998) was adopted to calculate ET for open water, pervious landscape areas, and GI, as it provides a well-organized collection of calculation procedures and a quantitative means to distinguish ET rates among different land covers, compared to other estimation methods, like the Hargreaves' equation (Hargreaves et al. 1985) included in SWMM, which is unable to reflect the species difference and the soil moisture effect. With some redefinition, the FAO-56 scheme can be divided into three parts. First, the reference ET (RET) is calculated based on the P-M equation parameterized for a reference grass cover at current meteorological conditions with unlimited water supplied; second, PET is calculated by multiplying RET by crop coefficients, which are used to reflect the ET difference between other species and the reference grass cover; third, AET is calculated by multiplying PET by the water stress coefficient. Although the concept of crop coefficients is widely used, they are highly empirical and dependent on the climatic conditions. So the crop coefficient was not adopted in this study, instead, PET rates from different land covers were directly calculated by plugging different parameters in the P-M equation (Table 2). Under the same

Table 2. Parameters used for the Penman-Monteith equation.

Types	Albedo	References	Height (m)	r_s_day	r_s_night	References
Water	0.08	(Stull 1988)	0.002[a]	0	0	(Oke 1988)
BR	0.305	(Stanhill et al. 1966)	0.50[b]	208	208	(Jones 1992)
GR	0.23	(Lazzarin et al. 2005)	0.12[c]	631	631	(Jones 1992)
LS	0.23	(Allen et al. 2005)	0.12[c]	50	200	(Allen et al. 2005)
DECI	0.16	(Stull 1988)	12.70[d]	307	307	(Jones 1992)
CONF	0.12	(Stull 1988)	14.35[d]	263	263	(Jones 1992)

Note: r_s_day = daytime surface resistance; r_s_night = nighttime surface resistance; Water = ponding and intercepted water; BR = bioretention; GR = green roof; LS = landscape; DECI = deciduous tree; CONF = coniferous tree (same for the following tables).
[a]Height of 2 mm was used for water surface (http://www.nc-climate.ncsu.edu/openwaterevap, accessed 03/17/2015).
[b]Height of standardized tall reference crop (Allen et al. 2005) was used to represent bioretention plants.
[c]Height of standardized short reference crop (Allen et al. 2005) was used to represent turf and green roof plants.
[d]Average tree heights were measured from Lidar data (http://gis.utah.gov/, accessed 03/17/2015) for the University of Utah campus.

meteorological conditions, choices of parameters in the P-M equation could reflect differences of species in leaf anatomy, stomatal characteristics, aerodynamic properties, and albedo (Allen et al. 1998). The parameters corresponding to turf grass were used here to represent vegetated landscapes (Allen et al. 2005), as the turf was generally the dominant species in landscapes in the urban study area. The averaged parameters corresponding to a range of common broad-leaved deciduous woody species and evergreen coniferous species living in semiarid regions were adopted for the deciduous and coniferous trees in this study (Jones 1992; Stull 1988). To make it easier to show the choices of parameters in the equation, the format of the American Society of Civil Engineers (ASCE) standardized P-M equation (Allen et al. 2005), with modifications, was used (3).

Specifically, the important parameters in Equation (3) can be calculated as follows,

$$C_n = \frac{3600\ \varepsilon}{1.01 \times R \times r_a}, \tag{8}$$

where ε is the ratio of molecular weights of water vapor versus dry air = 0.622, R is specific gas constant = 0.287 (kJ kg^{-1} K^{-1}), and r_a is the aerodynamic resistance for a certain type of reference land cover (s m^{-1}). And,

$$C_d = \frac{r_s}{r_a}, \quad (9)$$

where r_s is surface resistance (s m^{-1}). Both r_a and r_s were calculated in the traditional way (Allen et al. 1998). For open-water evaporation, r_s is set to zero and r_a=250/(1+0.536u_2) (Shuttleworth et al. 2009; Thom and Oliver 1977). The surface resistance of the standard cool-season grass was used for the vegetated landscape in this study (Allen et al. 1998). In the process of calculating surface resistances, the stomatal resistances of plants were assumed as the maximum reciprocals of their leaf conductances corresponding to the semiarid climate (Jones 1992). The leaf area indices of bioretention units and green roofs were estimated by a general equation for grass as $LAI=24h$, where h is the plant height (m) (Allen et al. 1998). And the averaged estimates from the direct measurements for a range of common species were adopted in this study as the leaf area indices for the deciduous and coniferous trees (Bréda 2003). The estimated surface resistances are listed in Table 2.

After PET is calculated, AET can be estimated as the product of PET and the water stress coefficient (K_s) under nonideal soil moisture conditions as described in Equation (10). The water stress coefficient reflects the influence of soil moisture deficit towards the plants functioning and ET rates. It was calculated as Equation (11) in this study instead of the original way (Allen et al. 1998), as the former better matched observations (Colaizzi et al. 2003). When the water supply is adequate in the soil and plants do not experience water stress, K_s is kept as 1 and the AET rate equals the PET rate (Allen et al. 1998).

$$AET = K_s \times PET, \quad (10)$$

$$K_s = \frac{ln\left[\left(1-\frac{D_r}{TAW}\right)100+1\right]}{ln(101)}, \tag{11}$$

where *TAW* is the total available soil water in the root zone (mm), which can be estimated from field capacity and wilting point (Allen et al. 1998), and D_r is root zone depletion (mm). D_r needs to be estimated by considering the full soil moisture balance at each time step, which is often difficult to determine and therefore K_s adjustment of PET has often been neglected. However, the water balance in the surface layer, soil layer, and storage layer of GI and in the aquifer of landscapes is updated at each time step in SWMM, which offers a way to calculate D_r. Therefore, the K_s-related equations were added to the SWMM code such that they can be updated in SWMM at every time step. In this way, hourly PET time series for different land covers were calculated outside SWMM using a Python code, and then they were imported into the updated SWMM and adjusted by the K_s to achieve AET rates (Fig. 13).

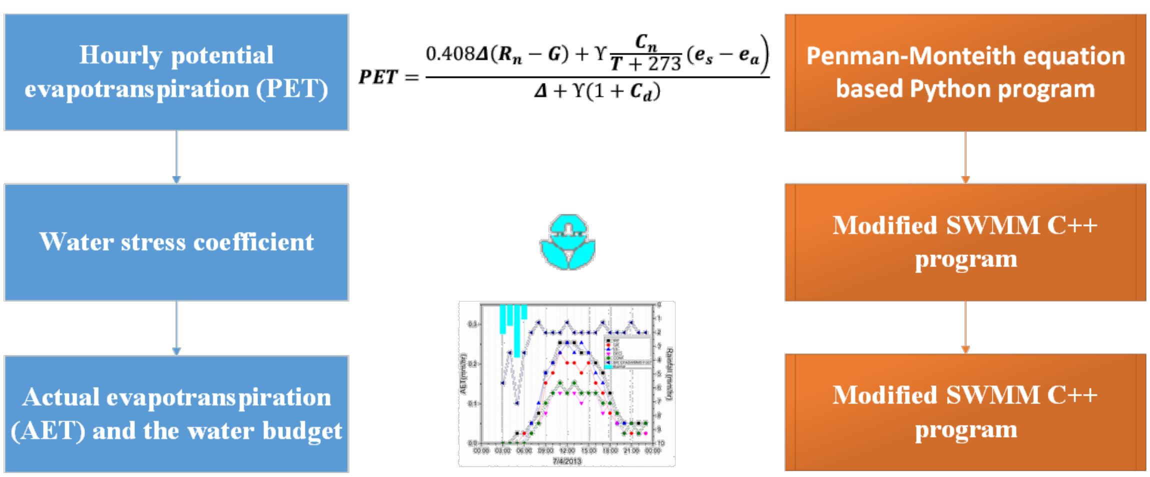

Fig. 13. Roadmap of incorporating FAO-56 ET scheme into SWMM.

3.3 Case Study

To test the updated ET scheme and its linkage to SWMM, five types of land covers were studied, including bioretention units, green roofs, turf-dominated landscapes, deciduous trees, and coniferous trees. Five hypothetical catchments (1 acre each) were created in the updated SWMM, and each of them was assumed to be completely covered by one of the studied land covers. The bioretention unit was parameterized to match instrumented test units on the University of Utah campus (Orr 2013) with approximately 0.6 m soil layer and 0.6 m gravel layer. Its bottom was lined with an underdrain, constraining release rate to 3.90 cm/hr to match measured infiltration rates. The green roof was specified in SWMM to match the green roof experiment on the Marriott Library on the University of Utah campus with a soil depth of 25.4 cm (Feng et al. under review). The drain coefficient of its underdrain was estimated as 35.24 cm/hr. The effective rooting depth for landscapes and trees was assumed to be 2 m (Allen et al. 1998). The PET calculation and SWMM model were parameterized based on the contexts of the University of Utah campus in Salt Lake City, Utah, which has a semiarid climate (Bailey 1979; Bair 1992; Eubank and Brough 1979; Russell and Cohn 2012) most of the time, but occasionally is classified as humid continental-hot summer (Brough et al. 1987). From 1981 to 2010 in Salt Lake City, the average annual precipitation was 408.94 mm and the average annual air temperature was 11.5 ˚C (NOAA 2013). From the Web Soil Survey (http://websoilsurvey.sc.egov.usda.gov/App/HomePage.htm, accessed 03/17/2015) operated by the U.S. Department of Agriculture (USDA)'s Natural Resources Conservation Service (NRCS), the primary soil type of the University of Utah campus was Bingham gravelly loam. Its hydraulic conductivity is approximately 0.889 cm/hr; and its porosity is 0.459, while its wilting point and field capacity are estimated at 0.148 and 0.288, respectively (Merrell 2013). From the U.S. Geological Survey (USGS) water level network, the water table in Salt Lake City at the selected location for the site was noted to be 38.26 m below the ground surface (U.S. Geological Survey 2015).

Besides rainfall data, other weather data were used in this study to calculate the PET time series. Year 2013 was selected as the major study period, which had the most complete time series data. 5-min rainfall and meteorological data (air temperature, solar radiation, relative humidity, wind speed, and air pressure) were acquired for two weather stations on the University of Utah campus (WBB, 40.76623N, 111.84755W; MTMET, 40.766573N, 111.828211W) by downloading from the Mesowest website (http://mesowest.utah.edu/, accessed 03/17/2015) operated by the Department of Atmospheric Science of the University of Utah. The average of the values before and after missing data was used to fill the gaps, although the missing data were small, 4% of the rainfall time series and 3.6% for other meteorological data. If the gaps of meteorological data were longer than one day, the available data on the former day were used to cover the gap.

3.4 Results and Discussion

3.4.1 Validation

Based on the P-M equation and meteorological data, the hourly PET time series for the water and the five land covers were calculated for the entire year of 2013. The daily averages of annual PET rates in 2013 from water, landscapes, bioretention, coniferous trees, deciduous trees, and green roof (listed in decreasing order) were 5.97, 4.80, 3.32, 2.92, 2.57, and 2.04 mm/day, respectively. Although there were no published PET comparisons between GI and traditional landscapes, this order of PET rates was consistent with existing research in terms of the relative magnitudes of landscapes and trees (Feng et al. under review; Litvak et al. 2014; Peters et al. 2011). The evaporation from water surfaces during summer months (June-August) was calculated as 10.75 mm/d, which matched the historical average of pan evaporation amounts during summers in Salt Lake City at Saltair Salt Plant (Western Regional Climate Center 2015).

The validation of the AET calculation for bioretention units and green roofs is based on previous field observations, while the validation for other land covers is based on comparisons with published data. The AET of a bioretention unit with a 220-m^2 drainage area and native plants grown inside (Orr 2013) was simulated using this updated SWMM model. Its AET was measured via LI-6400 (LI-COR, U.S.A.) on a total of 29 days within the period between 05/08/2012 to 11/04/2012 (Orr 2013). The mean of the simulated AET rates during those days is 3.25±2.37 mm/d, compared to the mean of observed values, 3.66±2.06 mm/d. The coefficient of determination (R^2) between the two time series is 0.57 (Fig. 14). The AET of a green roof consisting of vegetation from the sedum species (Feng et al. under review) was simulated to further test the updated SWMM model. The green roof AET was measured via a weighing lysimeter (Feng et al. under review). 112 days from 06/18/2014 to 11/02/2014 were picked and their daily AET amounts were used to validate the SWMM model. The mean of the simulated AET rates is 2.39±0.86 mm/d, while the mean of the observed values is 2.82±1.53 mm/d. The coefficient of determination (R^2) between these two time series is 0.78 (Fig. 14). Based on this comparison, the P-M scheme with the updated SWMM model generates reasonable AET estimates for bioretention and green roof systems in Salt Lake City. There is one observation that is

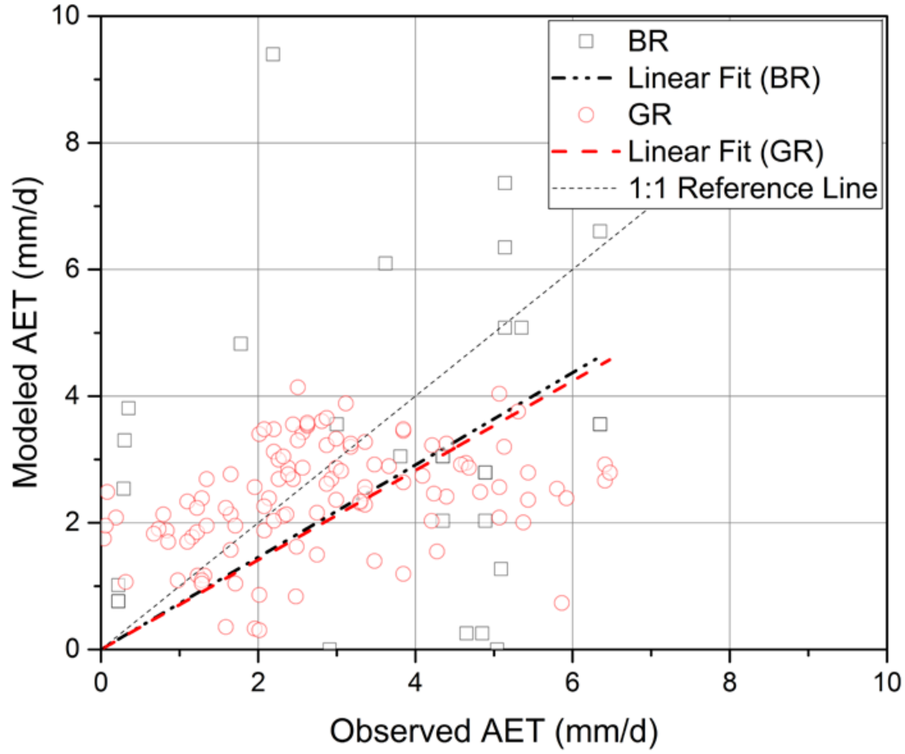

Fig. 14. Comparison of AET estimates by the updated SWMM with the observations for a bioretention site (BR) and a green roof (GR).

important to note. The simulated values tend to be lower than the measured values, which may be due to the reference values of the stomatal/surface resistance used in this study. The values selected might be too large for the specific plants in the bioretention unit and the green roof, especially during daytime. Further efforts to directly measure and improve the accuracy of the coefficients will be needed to continue to improve the modified SWMM.

Besides field experiments, published data were also used to validate AET estimates from the modified SWMM for the study period of 2013. During a period of eleven days in summer 2013, AET rates of bioretention ranged from 2.62 to 0.05 mm/d, and AET rates of green roof ranged from 2.18 to 0.51 mm/d (Fig. 15). Similarly, the simulated AET rates of landscapes ranged from 2.51 to 0.23 mm/d, which were lower than an observation (10.4±1.3 mm/d) in a similar climate but with irrigation (Litvak et al. 2014). Yet, the model results were close to other measurements for unirrigated turf grass, 2.99 mm/d (Peters et al. 2011). And simulated AET rates of deciduous trees and coniferous trees ranged from 2.06 to 0.61 and from 2.16 to 0.56 mm/d respectively, which were slightly lower than measurements in forest environments, 2.7±0.6 and 2.6±0.6 mm/d (Pataki et al. 2000), but were close to observations in an urban environment, which were measured to be less than 1 mm/d (Litvak et al. 2014). The comparisons of the modified SWMM with observations show the relative accuracy of the improved model. In addition, the comparison helps to identify future work that can be undertaken to continue to improve the modified SWMM.

3.4.2 Species Differences in ET and Water Budget

After validation, the rainfall-runoff events of the case study corresponding to the settings at the University of Utah campus in 2013 were simulated to demonstrate the improvements of the updated SWMM. Only precipitation without irrigation was considered in this simulation. A dry period after a 4-day rain event was selected to illustrate the AET differences among land covers (Fig. 15). The AET temporal variations during different rainy days reflected the influence of variable climate conditions. Although there were comparative rainfalls on July 6th, 2013 as on July 5th, 2013, the solar radiation and other meteorological factors did not favor a high ET. After the rainfall event concluded on July 8th, 2013, besides climatic influence, the AET amounts were more influenced by vegetation response to the soil moisture change, which was simulated by the water stress

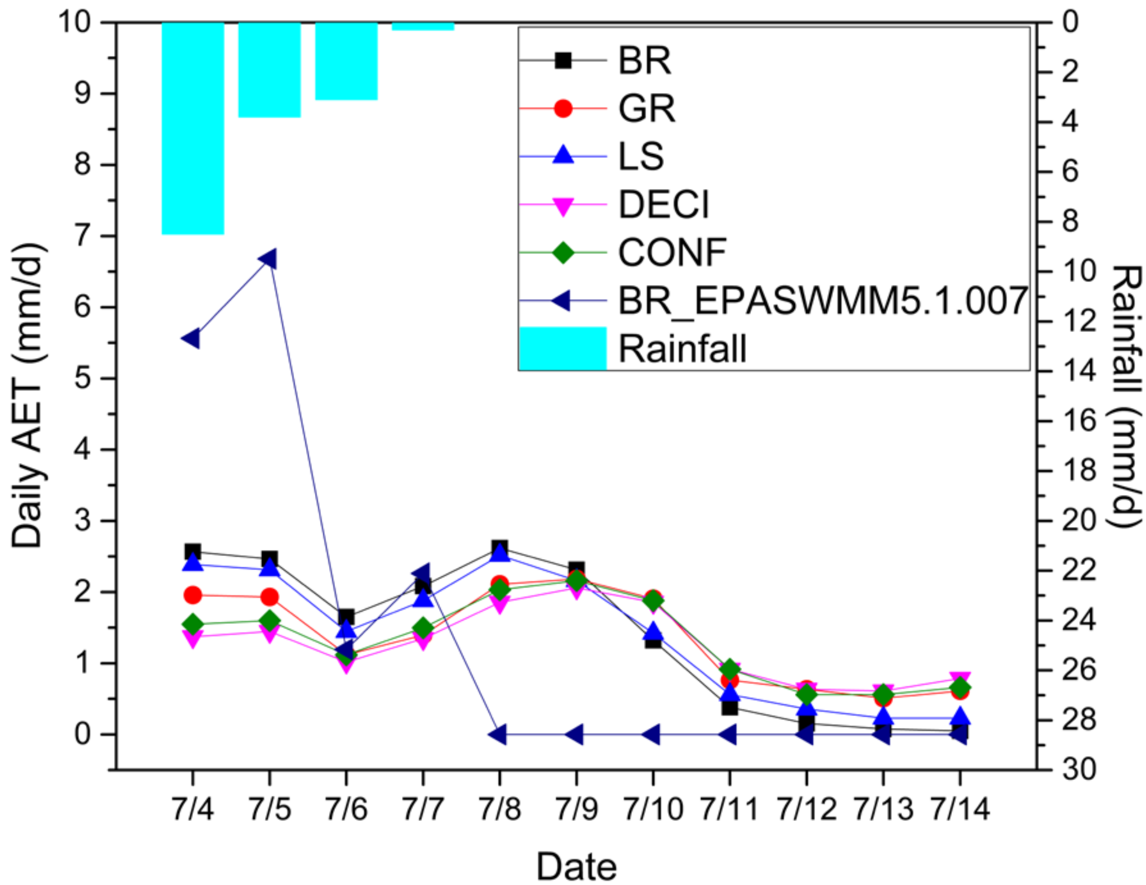

Fig. 15. Daily AET modeled by the updated SWMM from July 4th, 2013 to July 14th, 2013. BR = bioretention; GR = green roof; LS = landscape; DECI = deciduous tree; CONF = coniferous tree; Water = ponding and intercepted water (same for the following plots).

coefficient scheme. A nonmodified version of SWMM, EPA SWMM 5.1.007, was also used to simulate the ET pattern from the same bioretention setting for comparison. As it only allows PET inputs at a minimum of daily basis, the calculated hourly bioretention PET time series was summed up to the daily time series as its PET input. Without the water stress coefficient equipped, EPA SWMM 5.1.007 did not reflect the plant control of water flux to the atmosphere. Instead, it rapidly expelled the soil moisture during rainy days, leaving no water for ET in subsequent days.

A noteworthy observation of the SWMM results shows that even though having lower PET rates than landscapes, a bioretention unit could still have the highest AET amounts among land covers during rainy days and two days after. This was caused by the stronger infiltration capacity of bioretention, and more importantly, by the relatively high moisture fraction in the shallower depths of both the soil layer and storage layer of the bioretention, compared to the relatively low moisture fraction in the thicker unsaturated aquifer zone attached to the landscapes. Similarly, the green roof with lower PET capacity could also have higher AET amounts than trees during the first two rainy days, as its storage was relatively full compared to the ones supporting trees. Thus, a relatively larger water stress coefficient of the bioretention and the green roof was multiplied by their smaller PET values, which overall resulted in larger AET amounts than landscapes and trees. In this regard, these results also emphasize the importance of incorporating the water stress coefficient into the ET modeling, which can significantly modify the PET in GI, especially. To further assess the benefit of the updated SWMM, the annual water budgets of five land covers for 2013 were analyzed (Table 3). It is important to note that because the bioretention and the green roof were lined in this study, their percolation was zero. The bioretention and the green roof turned out to be able to infiltrate all the received rainfall without producing any surface runoff for the storm event sequence represented in the 2013 record, while discharge volumes from their underdrains were actually counted as runoff by EPA SWMM as defaults which were approximately equal to 50% of their inflows (Table 3). Although landscapes and trees generated surface runoff, most of their inflows were released via ET. This difference in the way of utilizing the stormwater may be due to the undersizing of the storage spaces of the bioretention and the green roof, compared to the

Table 3. Simulated annual water budgets in 2013 (mm/yr).

Budget	BR	GR	LS	DECI	CONF
Rainfall	466	466	466	466	466
ET	253	233	427	389	393
Percolation	0	0	9	47	44
Runoff	213	233	29	29	29

Note: BR = bioretention; GR = green roof; LS = landscape; DECI = deciduous tree; CONF = coniferous tree.

larger soil storages of landscapes and trees. However, this observation was partly driven by the selection of the characteristics of the GI, and other GI characteristics might lead to different observations; for example, the outflows from the green roof can be designed to be directed into landscapes or underground irrigation tanks instead of being directed into the drainage system, as assumed in this study. But the impact of the design variation on water budget characteristics is not the focus of this particular study.

PET rates were noted to vary among different land covers in the study period, and landscapes and the green roof were mostly the highest boundary and lowest boundary of all types of PET time series corresponding to different land covers, if the water evaporation was excluded (Fig. 16). To show the effects of the misuse of one PET rate for another on the estimation of AET rates and the water budget, a boundary value analysis was conducted (Copeland 2004; Reid 1997). Each of the three uniform PET hourly time series was applied for all five land covers in one of three rounds of tests. The three types of uniform PET hourly time series included the hourly average of five land covers' hourly PET rates, with the PET time series of the green roof as the lower end, and the PET time series of landscapes as the higher end. The results from the three tests were compared with the basic condition shown in Table 3, and the changes of magnitudes of the water budgets were evaluated (Fig. 17). Although the landscapes and the trees had the equivalent magnitudes

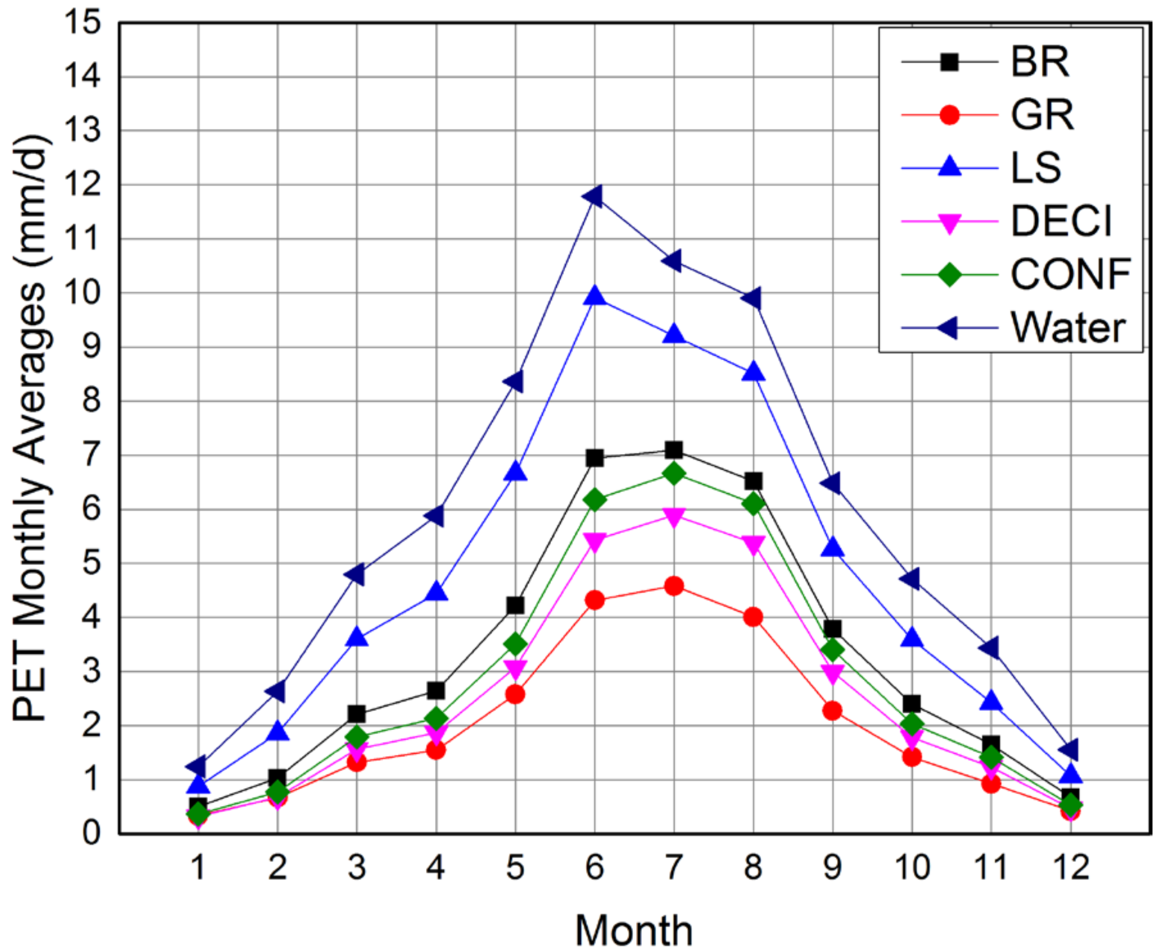

Fig. 16. PET monthly averages in year 2013.

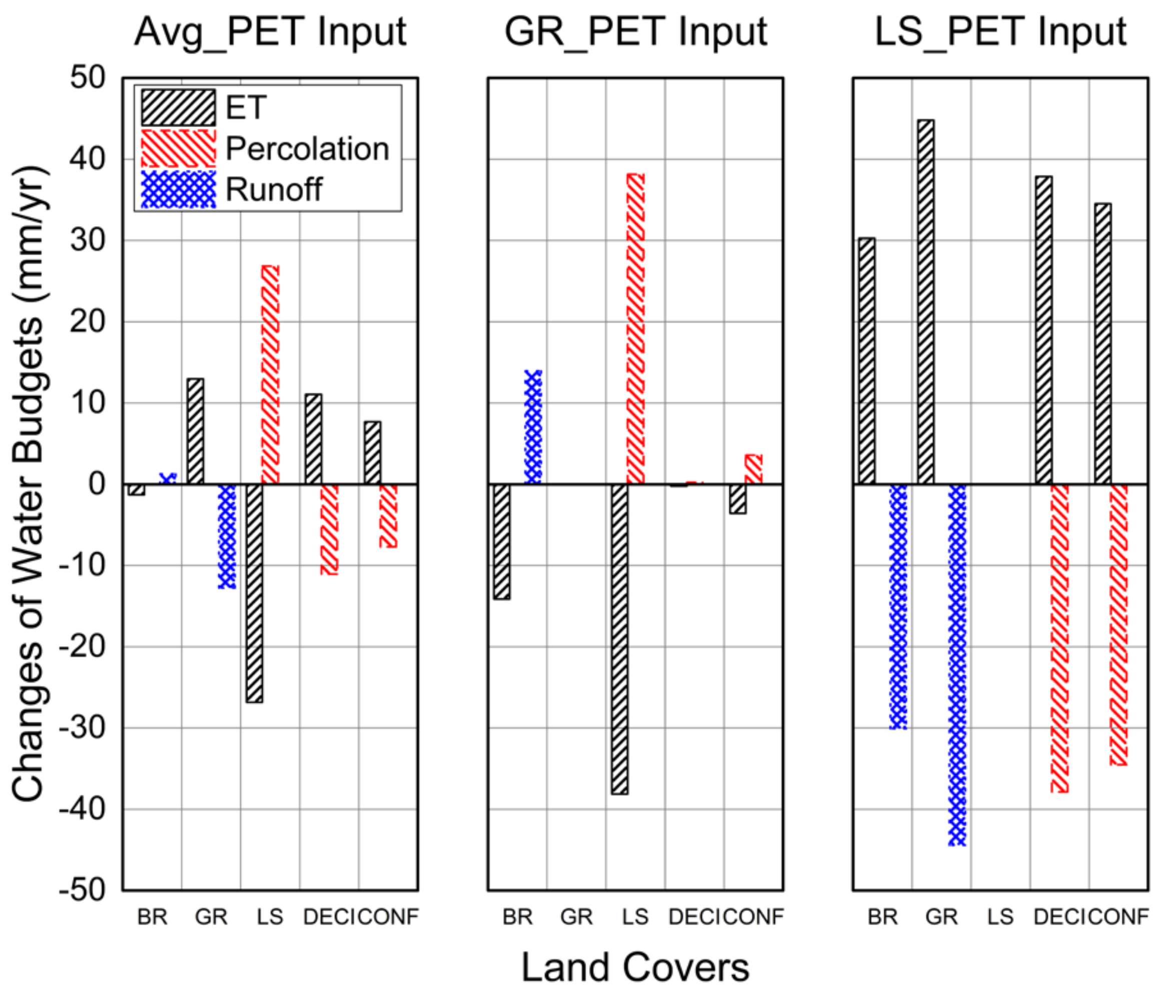

Fig. 17. Changes of the water budgets of five land covers due to different choices of PET inputs. Avg_PET Input = hourly averages of PET time series of five land covers as the PET input for each land cover, GR_PET Input = hourly PET time series of the green roof as the PET input for each land cover, and LS_PET Input = hourly PET time series of landscapes as the PET input for each land cover.

of the changes between percolation and AET in the opposite direction, the percent of the change with regards to percolation was much higher than that of AET, as the baseline amount of percolation was much smaller than its baseline comparative of AET; so the percolation was the most sensitive component in the water budgets of landscapes and trees to the incorrect estimate of the PET inputs (by assuming a different land cover). Specifically, if either the hourly average PET rates or the green roof PET rates were inaccurately represented as the PET input for the landscapes, the landscapes' annual percolation could experience an increase from 9.40 to 36.25 mm/yr or to 47.52 mm/yr, which was equivalent to an increase of 26.85 mm/yr (286%) or 38.13 mm/yr (406%). As explained above, the increases in percolation were coupled with the equal decreases of AET amounts, which were equivalent to 6.28% and 8.92% decreases of AET amounts for the two cases, respectively. Again, the result that the percent of changes of AET amounts was lower than that of percolation was because the baseline AET magnitude was larger than the baseline percolation magnitude. In contrast, if the higher landscape PET rates were used to represent deciduous and coniferous trees, the latter's percolation experienced a decrease of 37.87 mm/yr (80%) and 34.52 mm/yr (79%), respectively. Also, notably, if the landscape PET rates were used as the PET rates of GI, which is quite common for stormwater modeling, the AET rates of the bioretention and the green roof were overestimated by 30.28 mm/yr (12%) and 44.81 mm/yr (19%), with an underestimation of runoff by 30.07 mm/yr (14%) and 44.42 mm/yr (19%), respectively. Thereby, the inaccurate representation of PET rates for different GI and other land covers could generate large errors for runoff and water budget estimates. The assumption of a uniform PET spatial distribution can clearly be problematic if the catchment incorporates a variety of landscapes and GI.

3.4.3 Subdaily ET Patterns

Two days were selected to show the subdaily AET patterns simulated by the updated SWMM (Fig. 18, Fig. 19). The wet day was July 4th, 2013, which had 8.51 mm of rainfall in the early morning, and the dry day was July 8th, 2013, which was the next day after four

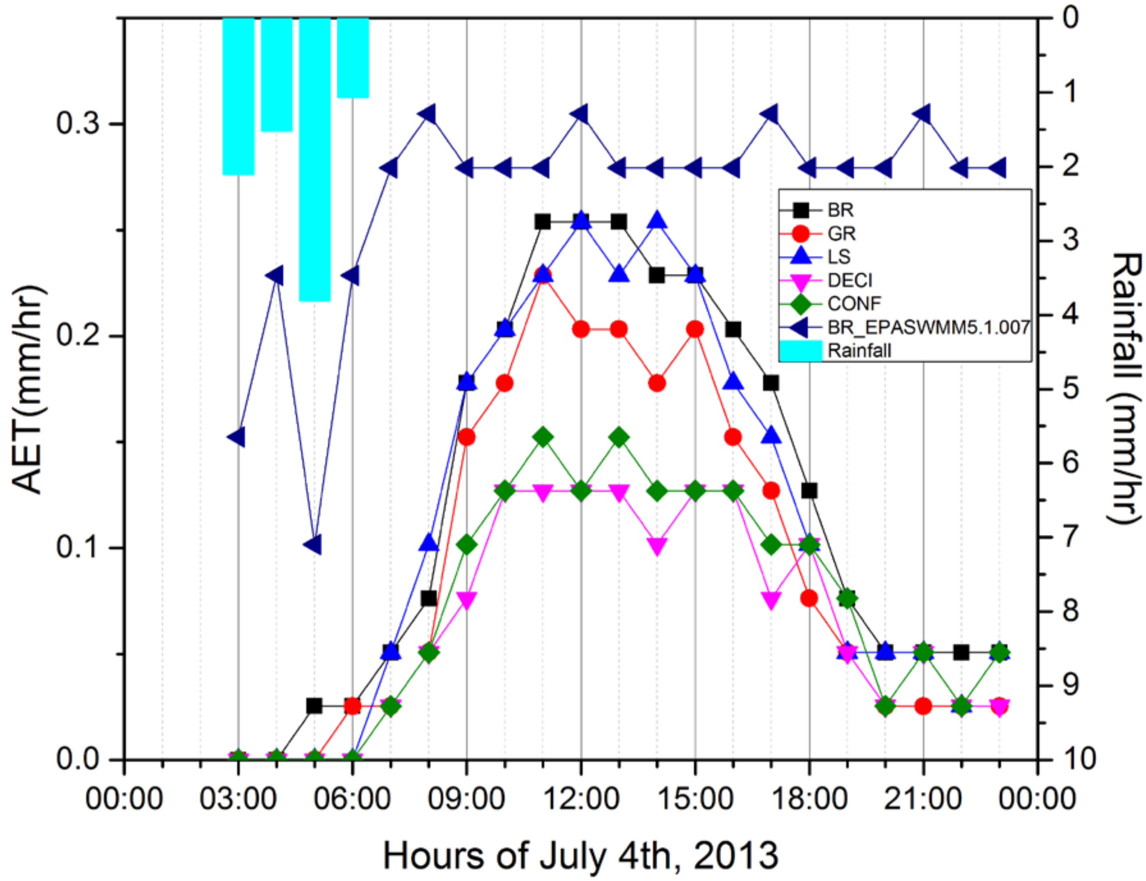

Fig. 18. Subdaily ET patterns on July 4th, 2013.

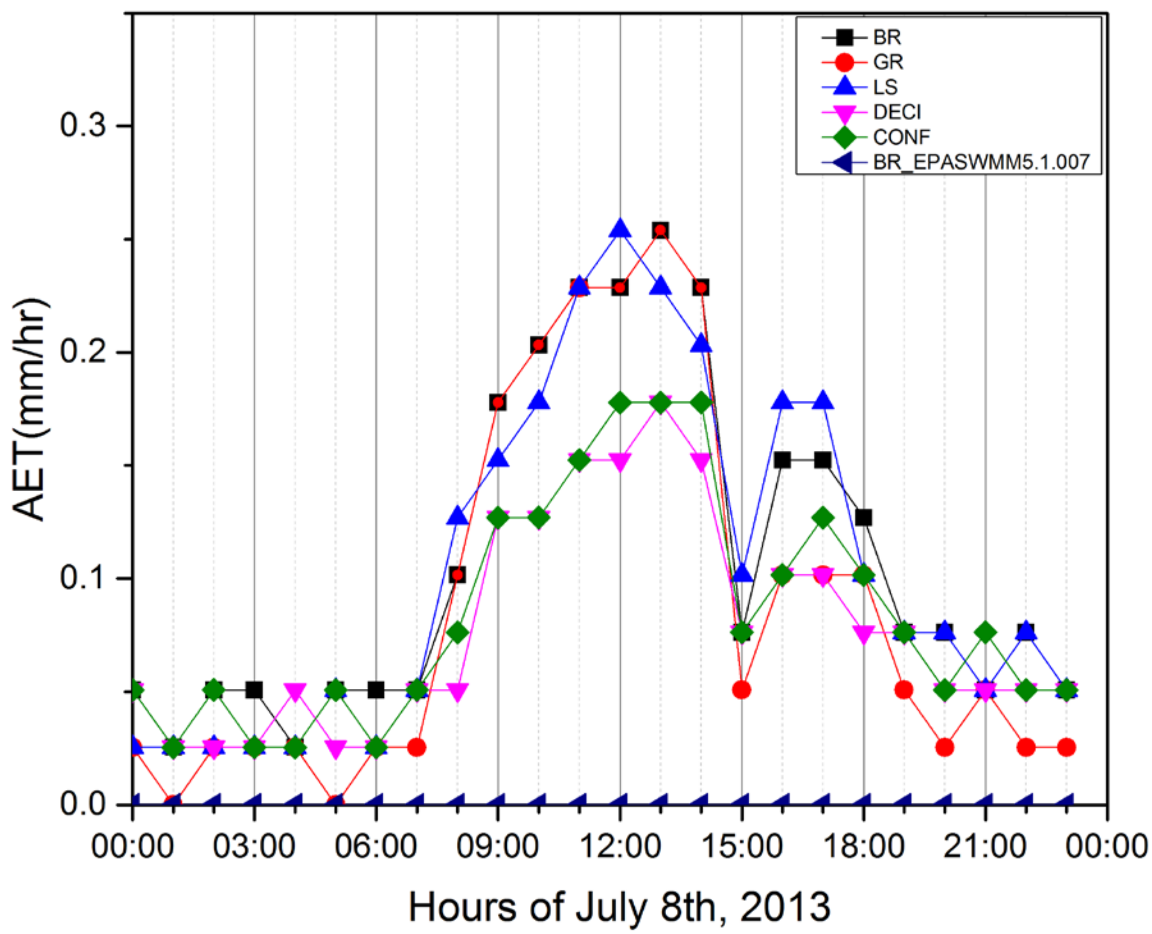

Fig. 19. Subdaily ET patterns on July 8th, 2013.

wet days having a total 15.70 mm of rainfall. All types of land covers showed a peak of AET rates within either day around noon. The only exception was the AET pattern of the same bioretention setting simulated by the EPA SWMM 5.1.007 (not modified with the updated ET modules), which simulated the highest AET rates compared with the other cases during all hours on the wet day, even during the rainy night hours. The fluctuation of its AET during rainy night hours was because ET was set to be zero during rainfall time steps. As EPA SWMM 5.1.007 only allows PET inputs at a minimum of a daily step, its imported daily PET amount is assumed to be evenly distributed at any subdaily step, so its

PET rates are the same across a 24-hour period in a given day, even between the daytime and the nighttime. This partly explains the discrepancy between the two models. Also importantly, there is no further limit to control the PET rates in EPA SWMM, so the high PET capacity consumed available stormwater quickly so that no water in the bioretention was available for further ET on the dry day. And in turn, the available storage capacity for next rain events is incorrectly calculated to be larger than it actually is. However, AET rates should vary across a day, depending on the weather conditions, and AET should typically occur for at least one day after four days of rainfall due to the plants' regulation of soil moisture. Thereby, the current EPA SWMM does not reflect a realistic ET temporal pattern at the subdaily scale and during the dry periods after rain events, but the updated SWMM can generate a more accurate pattern. Notably, the green roof accompanied with the bioretention and landscapes had higher AET rates than trees during noon hours even until the dry day, although trees had higher initial PET rates than the green roof. This indicates that the green roof may also allow high AET to happen only if there are adequate moisture and favorable weather.

3.4.4 Irrigation Effect

The landscape irrigation schedule of the University of Utah campus was used to simulate the effect of irrigation on the water budget. Intensive irrigation on the campus generally occurs from May to October. The irrigation amounts were controlled by a PET-based operating system, in which RET rates were calculated based on the weather data collected by an automatic weather station and the irrigation amounts at different vegetated zones were determined by the PET rates computed as the products of RET rates and crop coefficients. One vegetated zone was watered at nights only if the accumulative daily irrigation requirements (PET) exceeded a designated threshold value, otherwise the current day's PET amounts would be rolled over to the next day. As this study was designed to test out the new model instead of simulating the campus irrigation, a constant crop coefficient (0.6) was used, which was the minimum of the crop coefficients used on campus; and the catchments in this study were forced to be irrigated only during the hour between midnight and 1:00 am everyday instead of having the more sophisticated schedule used on campus. The landscape PET time series calculated in this study was used to estimate irrigation amounts. As GI was not supposed to be irrigated, only landscapes and trees were tested in this section.

If all the calculated PET time series were correctly used for their matching land covers and the simulated irrigation was added, which exceeded the annual rainfall total in 2013, then annual ET was raised by 185%, 130%, and 153% from landscapes, deciduous trees, and coniferous trees, respectively, while percolation was increased by 52%, 605%, and 439% (Table 4), compared to the baseline condition (Table 3). Notably, the fraction of ET within the water budget of landscapes rises from 92% in the rain-fed condition (Table 3) to 97% in the rain-and-irrigation-supplied condition (Table 4). However, the fractions of ET within the water budget of the deciduous trees and coniferous trees decreases from 83% and 84% in the rain-fed condition (Table 3) to 71% and 79% in the rain-and-irrigation-supplied conditions (Table 4), respectively. This indicates that the response of adding irrigation to the role of ET within the water budget may vary according to the PET magnitudes. When water supply is not enough, the water demand of the plants having high PET capacity, like the landscape in this study, is not fully met. When extra irrigation is

Table 4. Simulated annual water budgets with irrigation added in 2013 (mm/yr).

Budget	LS	DECI	CONF
Rainfall	466	466	466
Irrigation	794	794	794
ET	1216	897	993
Percolation	14	334	237
Runoff	29	29	29

supplied, those plants could exploit more water of the water budget until they reach their PET limits. However, the plants having lower PET capacity, like trees in this study, may (almost) reach their PET limits even when only rainfall is supplied. Further addition of irrigation will not be significantly used by those plants, and therefore the extra water supply will lose via other pathways, like percolation shown in this study (Table 4). Runoff shows no increase after irrigation was applied, due to the pervious surfaces of each type of the studied land covers, so the fractions of runoff within the water budgets are getting lower from 6% in the rain-fed condition (Table 3) to 2% in the rain-and-irrigation-supplied condition (Table 4).

Next, this water budget driven by the added irrigation and the matching PET inputs was regarded as the new baseline condition for the following comparisons. By contrast, if the lower green roof PET time series was used as the PET inputs, the AET amounts of landscapes, deciduous trees, and coniferous trees were underestimated by 494 mm/yr (41%), 175 mm/yr (20%), and 272 mm/yr (27%), while their percolation amounts were overestimated by the similar magnitudes but by the larger ratios as 3452%, 53%, and 115%, respectively. If the higher landscape PET time series was used as PET inputs for trees, the AET amounts of deciduous trees and coniferous trees were overestimated by 319 mm/yr (36%) and 223 mm/yr (22%), while their percolation amounts were underestimated by the similar magnitudes but by higher ratios as 96% and 94%. Therefore, as irrigation will significantly raise the amounts of ET and percolation in the semiarid or drier climates, the assumption of the homogeneous PET distribution, which mismatches PET rates with their corresponding species, can cause larger errors in the estimation of the water budget. Notably, the runoff was not changed significantly in these tests; it might be because the whole catchment was assumed as a pervious surface, which absorbed all the rainfall and irrigation inputs, as the irrigation was given smoothly through one hour at an even and low intensity. And although not quantified, the miscalculated storage available, given irrigation inputs to pervious areas, is expected to cause errors in the existing EPA SWMM estimates of long-term runoff volume reductions from GI implementations.

3.5 Conclusion

Targeting to improve the ET mechanism in stormwater modeling for a better evaluation of water budget changes due to GI implementations, this study incorporated a revised FAO-56 ET estimation framework (Allen et al. 1998) into a modified EPA SWMM model. The SWMM model was then upgraded to be able to accept an input of the heterogeneous and subdaily PET time series data, which other simple methods like monthly averages or Hargreaves equation cannot represent. Based on the available data observed in Salt Lake City, Utah and other places with similar climates, the results of the ET estimates from the modified SWMM were reasonable. The updated ET routine was validated for bioretention (R^2: 0.57) and green roof (R^2: 0.78) areas by comparing with direct measurements. The importance of having matching PET inputs for the corresponding vegetated land covers was presented. If the lower green roof's PET time series was used as the PET input for landscapes, the latter's annual percolation and AET amounts were overestimated and underestimated by 38.13 mm/yr and 38.13 mm/yr, respectively, which were equivalent to a 406% increase of percolation and a 9% decrease of AET amounts; but if the PET inputs for the bioretention and the green roof were replaced by the higher landscape PET input, their annual AET amounts could be overestimated by 30.28 mm/yr (12%) and 44.81 mm/yr (19%) with an underestimation of runoff by 30.07 mm/yr (14%) and 44.42 mm/yr (19%), respectively. And EPA SWMM currently can only accept PET inputs on a daily basis and does not take into account the moisture stress, so it was shown to give unrealistic estimates of ET at a subdaily scale and during the dry periods after the rain events. Furthermore, irrigation activities in dry climates will greatly increase both the ET and percolation amounts, which could further magnify the errors mentioned above with regards to the simplified modeling of the ET process in SWMM. Therefore, the oversimplified representation of the ET process can generate large errors for the water budget estimation, especially in dry climates. It is thus recommended that the more physically based ET mechanism with higher spatial and temporal resolutions proposed in this study is necessary.

This method mainly requires four weather data, including solar radiation, air temperature, wind speed, and relative humidity. The parameters provided by this study could be used for other places with similar climates. Lack of observational data for several key parameters, like the stomatal conductance or resistance, for the species may lead to uncertainty in application of the modified model; therefore more studies are required to continue to improve the P-M scheme for GI in the modified SWMM. Although not common in the urban water engineering literature, studies applying approaches and measuring instruments from biology can be effectively used to fill in this gaps.

CHAPTER 4

POTENTIAL OF GREEN INFRASTRUCTURE TO RESTORE PREDEVELOPMENT WATER BUDGET OF A SEMIARID URBAN CATCHMENT

4.1 Introduction

Understanding the water cycle in cities is critical to plan and design systems providing water services (e.g., supply, stormwater management, flood control) and supporting energy generation, food security, recreation, public safety, and economic development. Urbanization alters the water budget due to vegetation removal, creation of impervious surfaces, changes in water use and water diversions (Claessens et al. 2006). Such significant land use changes lead to a complicated mix of hydrologic responses. Surface runoff in a watershed may increase by 75% or more when a watershed is completely urbanized (Haase 2009). Subsurface recharge may be increased or decreased, depending on the circumstances (He and Hogue 2012; Hibbs and Sharp 2012).

Evapotranspiration (ET) may be the major factor in determining how much water is available for infiltration, and as such represents the controlling component of the urban water balance profile (Ellis 2013). The range of ET ratios within the water budgets in urban areas tend to be variable due to a complex combination of factors. For example, urbanization may increase or decrease vegetation compared to predevelopment conditions in different environments, which can raise or dampen ET amounts. Change of land use and the related micrometeorological conditions may promote higher potential ET (Balling and Brazel 1987), partly caused by increased vapor pressure deficit due to urban heat island (Gwenzi and Nyamadzawo 2014). Also, imported water can provide more water supply for vegetation transpiration within the city boundary (Bijoor et al. 2014), which has been evaluated as a way to mitigate the urban heat island effect (Gober et al. 2009; Gober et al. 2012; Shashua-Bar et al. 2009). To respond to these changes and uncertainties, quantifying urban impacts on spatiotemporal water budget responses remains an area of great need, especially in the planning and design that guides the configuration and operation of water systems in cities.

As a remedy to alterations in the runoff component of the water budget, stormwater management goals historically sought to reduce the impact of urbanization on runoff peak discharge, and thus reduce downstream flooding. This goal has evolved to incorporate runoff volume reduction, water quality restoration, public health protection, and stream erosion control. So-called gray infrastructure approaches (i.e., pipes, channels, or storage tunnels) have been the typical stormwater management solution (Muller et al. 2015). In the past two decades, a second-generation approach called low-impact development (LID) (Coffman et al. 1999) emerged as an alternative to gray infrastructure approaches. However, most of those second-generation systems, like best management practices (BMPs) or sustainable drainage systems (SUDS) primarily driven by runoff control, may not provide a fully sustainable surface water management approach for urban catchment planning. Green infrastructure (GI), on the other hand, is more generally linked with ecosystem services and can be advocated to be more effective to achieve broader sustainability goals on site and catchment scales (Ellis 2013).

GI uses a range of decentralized, often vegetated practices to control stormwater through infiltration and treatment mechanisms close to its point of generation. As the goal of stormwater management has been inherently runoff control, most of the studies of GI have focused on its capacity to reduce stormwater runoff volume and delay the flow event (Alfredo et al. 2010; Brown et al. 2009; Burian and Pomeroy 2010; Culbertson and Hutchinson 2004; DeBusk et al. 2011; Fassman and Blackbourn 2010; Li et al. 2009; Trinh and Chui 2013; Voyde et al. 2010). However, in accordance with the concept of the integrated ecosystem management/stewardship (Chapin III et al. 2009; Falkenmark and Rockström 2004), there remains a need to consider GI in a more comprehensive water budget framework, and to re-evaluate the effect of GI in terms of restoring all components of the water budget (Burns et al. 2012; Fletcher et al. 2013; Olszewski and Davis 2013).

Components of the water budget other than surface runoff may also be largely affected by applying GI, and those effects are much less understood. For example, overcompensation of stormwater infiltration may result in a rise in the groundwater surface (Göbel et al. 2004). Besides restoring the natural hydrological processes in urban areas, GI may also contribute positively to other ecosystem services, like mitigating the human and ecological stress of the UHI (Endreny 2008). This need also responds to an emerging international trend of near-natural stormwater management, which has become established in Germany alongside traditional urban drainage goals (Göbel and Coldewey 2013; Göbel et al. 2004), and aims at replicating the quasi-natural local water balance so as to preserve the local ecosystem's integrity (Keßler et al. 2012).

However, comprehensive analysis of GI impacts on the urban water budget is limited by the lack of appropriate modeling tools for the water cycle components, especially ET, which is arguably the most difficult component of the water budget to estimate in cities and can lead to significant uncertainty in water budget accounting (Pataki et al. 2011). Further, there is a lack of definitions and corresponding planning and design metrics that can be applied to evaluate the effects of GI in restoration of the catchment's water budget and ecosystem health, and to estimate the efficiencies to compare different GI solutions.

This paper describes a modeling approach used to assess the implications of GI on the water budget of an urban catchment in the semiarid climate of Salt Lake City, Utah, USA. A modified version of the United States Environmental Protection Agency (EPA) Storm Water Management Model (SWMM) was used to investigate water budget changes affected by GI implementation compared to an existing condition and a simulated natural condition. The study was performed for three types of 1-year periods – an average precipitation year, a dry year, and a wet year. Two coefficients are proposed to evaluate the efficiency of restoration of the natural hydrology by GI. The updated SWMM and the proposed evaluation coefficients are expected to serve as guidance tools in the design process for watershed scale stormwater management plans. The research results of this study will also help guide future water budget studies for GI applications.

4.2 Methods

4.2.1 Modeling Framework

EPA SWMM 5.0.022 was selected as the modeling platform for this study because of its ability to simulate the urban water budget. SWMM is able to simulate a water budget for both natural and urban environments, and it is one of the few models with the flexibility to simulate multiple types of GI (Elliott and Trowsdale 2007). Another advantage of EPA SWMM is its open-source policy, which allows modifications to its code to achieve specific requirements.

A modified form of SWMM (Feng and Burian under review) was used in this study for improved simulation of ET. Specifically, compared to the original EPA SWMM with homogenous ET representation and daily ET inputs, the improved version of SWMM allowed for a heterogeneous ET representation for up to six types of land covers, including ponding water, bioretention, green roofs, landscape, deciduous trees, and coniferous trees. The modified form also permitted subdaily ET time series inputs and corrections based on calculated water stress coefficients to account for soil moisture variations (Allen et al. 1998).

4.2.2 Case Study

A small urban catchment (0.11 km^2) located on the campus of the University of Utah in northeast Salt Lake City (SLC), Utah, USA was selected as the case study of this research (Fig. 20). SLC has a semiarid climate (Bailey 1979; Bair 1992; Eubank and Brough 1979; Russell and Cohn 2012). Its average annual precipitation is 409 mm and the average annual air temperature is 11.5˚C, from 1981 to 2010 (NOAA 2013). From the Web Soil Survey (http://websoilsurvey.sc.egov.usda.gov/App/HomePage.htm, accessed 03/17/2015) operated by U.S. Department of Agriculture (USDA)'s Natural Resources Conservation Service (NRCS), the primary soil type of the catchment is Bingham gravelly loam. Its hydraulic conductivity is about 0.899 cm/hr; and its porosity is 0.459, while its wilting point and field capacity are 0.148 and 0.288, respectively (Merrell 2013). The

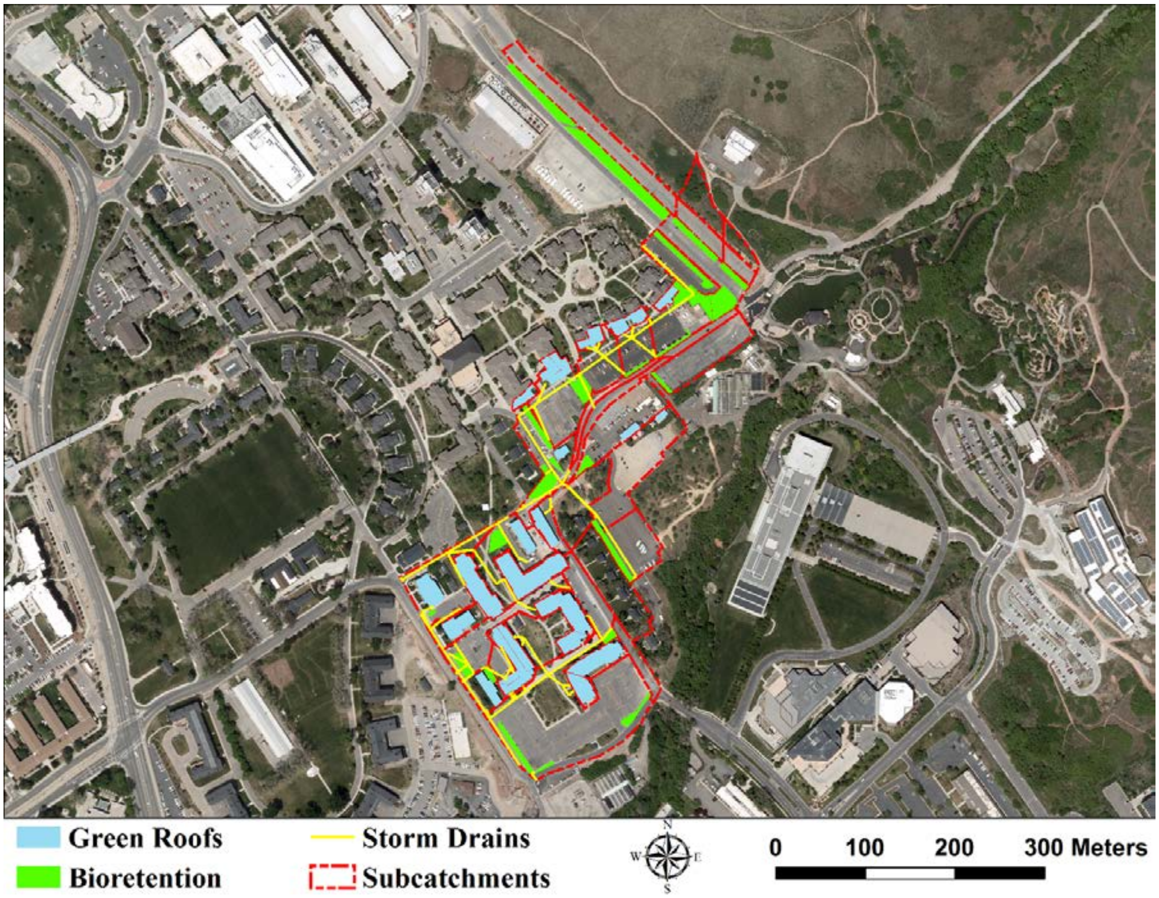

Fig. 20. Delineated catchment.

water table was measured as 38.26 m below the land surface by a U.S. Geological Survey (USGS) groundwater station near the study site (U.S. Geological Survey 2015). The average thickness of the local valley fill aquifer was estimated as 823 m (Arnow and Mattick 1968).

ET contributions from the deep groundwater were not considered in this study. Adding the deep percolation in SWMM will generate a very small (1.07 mm) difference in ET for the predevelopment condition for the year 2014 (simulated by EPA SWMM 5.1.007). This is equivalent to 0.22% of the total annual precipitation.

Meteorological data of 5-min interval from two weather stations operated by the

Department of Atmospheric Science at the University of Utah were used for this study; they were downloaded from the Mesowest website (http://mesowest.utah.edu/, accessed on 03/17/2015). The Mountain Met (MTMET) weather station (40.766573N, 111.828211W) located within the study catchment was used to represent meteorological conditions from July 3, 2012 to December 31, 2014. Meteorological data before that period (starting in 2011) was obtained from the nearby William Browning Building (WBB) weather station (40.76623N, 111.84755W), which is 1.66 km from the MTMET station. Except the precipitation, other raw data were summed up to hourly amounts.

Spatial distribution and fractions of current land cover within the watershed were determined by manually interpreting 1-foot-resolution orthophotography images downloaded from Utah Automated Geographic Reference Center (AGRC, http://gis.utah.gov/, accessed 06/08/2015) and verifications by site visits. The average building height in the catchment was estimated at 10.66 m based on 1m and 1.25m Lidar data collected by Utah AGRC (http://gis.utah.gov/, accessed 06/08/2015) in 2006. Similarly, the average heights of the deciduous and coniferous trees were estimated as 12.70 m and 14.35 m, respectively.

A separate storm drainage system services the catchment, and directs runoff into Red Butte Creek. The drainage catchment was delineated and subdivided based on terrain, locations of storm drains, and other local terrain features (e.g., curb and gutters). Several site visits were made to identify the locations of storm drain inlets and outfalls. A 2150 Area Velocity Flow Module (Teledyne Isco, USA) was installed in the storm drain at the outlet of the catchment to measure the flow rate at 1-minute increments.

The SWMM model corresponding to the baseline (developed) condition was manually calibrated by comparing the flow rates modeled by the updated SWMM with the flow rates measured by the flow meter from late February to early June 2014 (Fig. 21). Width, slope, imperviousness percentages, Manning's roughness coefficients, depression storage, and infiltration parameters (Green-Ampt method was used) of subcatchments, and size, length, and slope of stormwater pipelines were adjusted during the calibration. Trial-and-error method was used until no improvement of the modeling accuracy was reached compared to the observed peak flows. As the flow peaks happen immediately after rainfall events, evaporation and ET were assumed to be negligible for calibration. The outflow time series modeled by the updated SWMM did not pass the Kolmogorov-Smirnov normality test at the significance level of 0.05. The nonparametric Spearman correlation coefficient was then used to measure the strength of the relationship between the time series of the modeled and observed flow rates, calculated as 0.59 at a significance level of 0.05. A linear regression analysis with a fixed intercept at zero was used to test the strength of a linear relationship between two time series, which yielded an R-Square of 0.56. These indicate that the baseline model has been reasonably calibrated for the purpose of this study.

4.2.3 Scenarios

Three scenarios were simulated: baseline (BL), green infrastructure (GI), and natural hydrology (NH). In the NH scenario, the catchment was modeled as being covered with native grasslands like wheatgrass and bluegrass (Ehleringer et al. 1992), as the open meadow is the dominant landscape at the foothill environment right next to the study catchment. Green roofs and bioretention applied to the catchment in the GI scenario. The

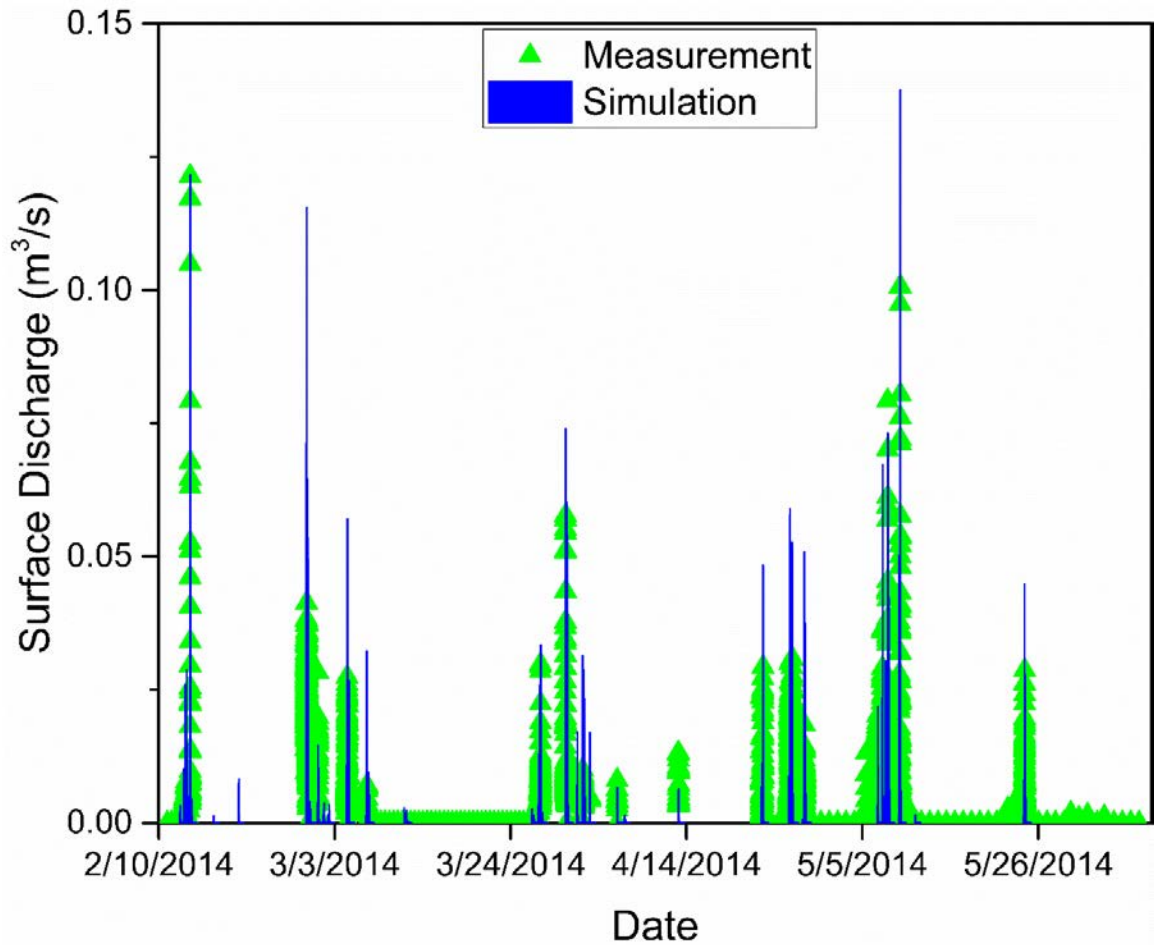

Fig. 21. A comparison of modeled flow data with observations.

numbers and the sizes of green roofs and bioretention were determined by designing them to reduce 1-year stormwater runoff volumes by 80%, which corresponds to an amount consistent with goals for stormwater quality management plans (Horner et al. 2004; Sullivan et al. 2010). SWMM simulations were executed with green roofs and bioretention iteratively added to the BL scenario until the 80% runoff reduction was achieved. The green roofs were placed on flat roofs and bioretention was placed on open ground areas. These units were configured to match the existing designs and recommendations for the climate of SLC (Houdeshel et al. 2012; Houdeshel and Pomeroy 2014).

The bioretention units were parameterized in SWMM to match instrumented test units on the University of Utah campus (Orr 2013) with a 0.6 m soil layer and a 0.6 m gravel layer. Bottoms were lined with underdrains with a 1.5 cm/hr drain rate. Green roofs were designed to mimic a green roof and the green roof experiment on the Marriott Library on the University of Utah campus, which has a soil depth of 25.4 cm (Feng et al. under review). Green roof underdrains were assumed to have the same drainage rate as the bioretention units.

Three years were selected for the simulations used in this study, based on the availability of the data and the relative magnitudes compared to the annual average precipitation (409 mm). It is difficult to find three years having total precipitation depths perfectly distributed around the annual average while not having missing weather observations at the study site. Therefore, in spite of the precipitation depth being close to the annual average, 2012 (371 mm) was assumed as a relatively dry year for this study. Although the yearly precipitation depth is higher than the annual average, 2014 (482 mm) was assumed as an average year for this study. 2011 (688 mm) was assumed as a relatively wet year for this study. SWMM simulations were independently executed for the calendar years (January 1 to December 31) for each of these three years at a 1-minute time step. Annual water budgets were summed and compared among different scenarios and different years. Results of the average precipitation year (2014) were then used to further explore the water budget variations among different scenarios at the monthly, daily, and hourly time scales. The average precipitation year (2014) was also used to test two indices proposed by this study.

4.3 Results

4.3.1 Annual Water Budgets

Across all three studied years, the BL scenario always had the largest stormwater discharge volume (Fig. 22). Specifically, the surface runoff increased from 41 mm in the NH scenario to 155 mm in the BL scenario (274%) in the dry year, from 20 mm in the NH scenario to 175 mm in the BL scenario (769%) in the average year, and from 30 mm to 260 mm (778%) in the wet year. The GI scenario, however, had a closer surface discharge amount to the NH scenario, which was a sum of surface runoff and part of the discharge

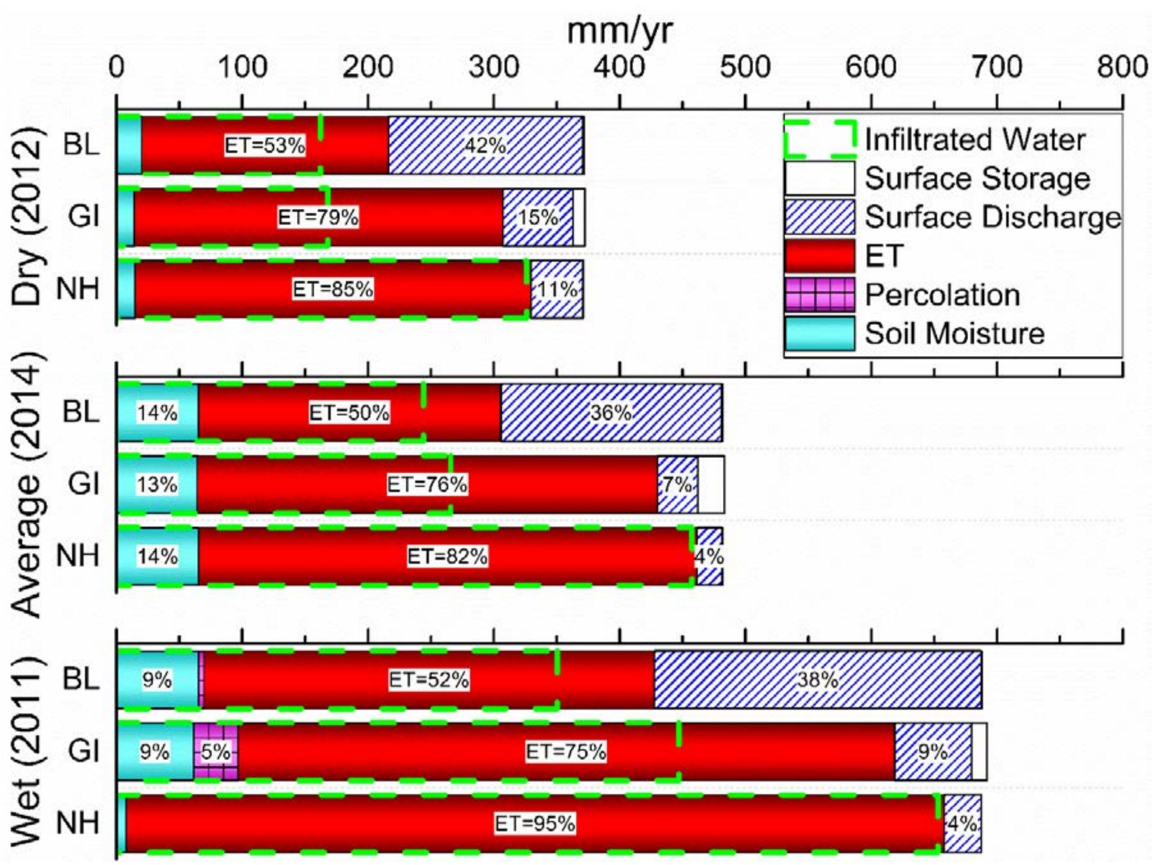

Fig. 22. Water budgets of the baseline scenario (BL), the green infrastructure scenario (GI), and the natural hydrology scenario (NH) in dry (2012), average (2014), and wet (2011) years.

from the underdrains of GI units that directly flows into the drain system. From the NH scenario, the surface discharge of GI scenario only increased by 15 mm (35%) in the dry year, 12 mm (58%) in the average year, and 32 mm (107%) for the wet year. Compared to the BL scenario, GI reduced surface runoff by 99 mm (64%) in the dry year, 143 mm (82%) in the average year, and 199 mm (76%) in the wet year.

ET was the dominant component of the water budget, as it accounted for 50-53%, 75-79%, and 82-95% of the water budget for the BL, GI, and NH scenarios, respectively (Fig. 22). The BL scenario had the lowest ET amounts compared to the other two scenarios (Fig. 22). Specifically, ET decreased from 315 mm in the NH scenario to 196 mm in the BL scenario (38%) in the dry year, from 396 mm in the NH scenario to 240 mm in the BL scenario (39%) in the average year, and from 650 mm to 358 mm (45%) in the wet year. The GI scenario, however, had closer ET amounts compared to NH scenario, with ET only being higher than the NH scenario by 22 mm (7%) in the dry year, 30 mm (8%) in the average year, and 128 mm (20%) in the wet year. Compared to the BL scenario, GI restored annual ET amounts by 97 mm (49%), 125 mm (52%), and 164 mm (46%) for the dry, average, and wet years, respectively.

Overall, development raised surface runoff annually by 274%, 769%, and 778% in dry, average and wet years, and reduced ET amounts by 38%, 39%, and 45%, respectively. GI restored annual surface runoff by 64%, 82%, and 76% and annual ET amounts by 49%, 52%, and 46% in the dry, average, and wet years, respectively, compared to the BL scenario.

Rainwater that was not stored on the surface or discharged from the surface (infiltrated water) had three destinations: storage in unsaturated soil, deep groundwater via percolation when the soil is saturated, and the air via ET. Most infiltrated water was lost through ET, and the majority of the remaining infiltrated water was stored in the soil layer (Fig. 22). Very small amounts percolated into the deeper soil. Within each water year, there was no significant difference in the soil moisture storage and percolation amounts between scenarios. The only exception was the GI scenario in the wet water year when percolation accounted for 5% of the total water budget.

4.3.2 Monthly Soil Moisture Balance Variations

The monthly water budgets between different scenarios were also compared to evaluate the monthly soil moisture balance (Fig. 23). The average year (2014) was evaluated for this purpose. Each component of the water budget is shown (Fig. 23), except for soil moisture storage (note: percolated water volumes are too small to be seen at the bottom of each stack). Monthly soil moisture storage is reflected by the difference between the precipitation and the sum of the other components (Fig. 23). Of note is that SWMM accounts for moisture stored within GI as surface storage by default. Therefore, when the sum of percolated water, ET, and discharge exceeds the precipitation, the storage space of GI is gaining moisture. For example, conversely, in March, August, and November, the soil located beneath GI was losing moisture because the sum of the percolated water, ET, discharge, and surface storage was greater than the total precipitation (part of the incoming precipitation goes into the GI instead of filling the soil moisture storage).

In this regard, the gross soil moisture storage of three scenarios increased in January, February, September, and December, while it decreased in April, May, and October. The three scenarios showed varying soil moisture patterns in other months. The BL scenario

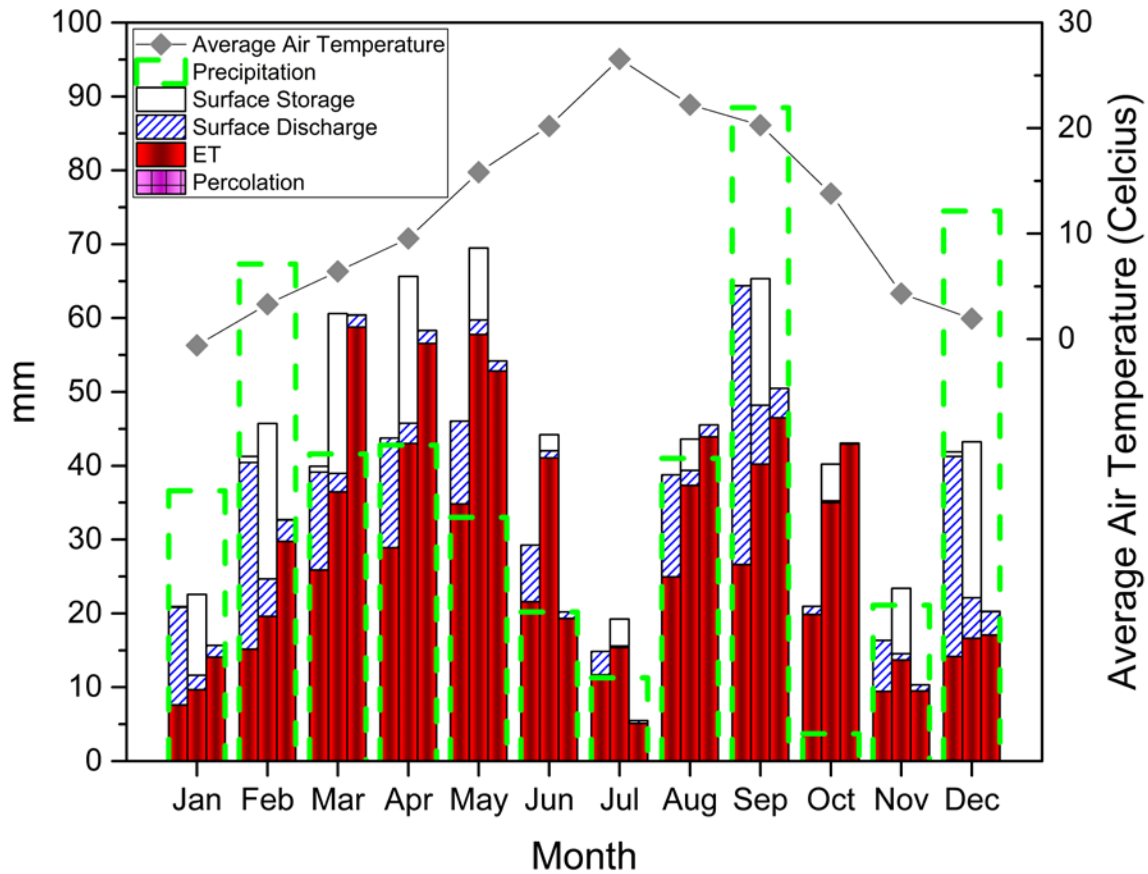

Fig. 23. Monthly water budgets simulated for the average water year (2014). The three stacked columns of each month from left to right represent BL, GI, and NH scenarios, respectively.

was gaining soil moisture, while the GI and NH scenarios were losing soil moisture in March and August. In the hot May, June, and July months, the GI scenario had the largest ET amounts. This happened after the soil moisture storage of the NH scenario decreased from March onward, when the GI scenario had surface storage. In the colder month of November, the GI scenario also had higher ET amounts than the NH scenario, which happened after a dry October when the soil moisture of the NH scenario was mostly drained.

The NH scenario had the largest ET amounts of any months when all three scenarios received larger precipitation than the sum of ET and surface discharge. The surface discharge amount of the BL scenario was always greater than the other two scenarios in any month.

The GI scenario always had the largest surface storage compared to the other two scenarios in any month (Fig. 23). The extent of surface storage of the GI scenario increased during the months when the precipitation exceeded the sum of surface discharge and ET, while vice versa happened when the precipitation was lower than the sum of surface discharge and ET. The only exception was July, when the surface storage was larger than the previous month but ET was higher than the precipitation.

4.3.3 Daily and Hourly Soil Moisture Balance Variations

The water budget simulations for the average year were further analyzed at the daily and hourly scales to explore the mechanism of GI and the potential to restore the catchment's water budget after precipitation events. A period of 48 days (June 17th to August 3rd) experiencing two major rain events was selected for evaluation (Fig. 24). The BL scenario retained the least total moisture (sum of soil and surface storages) compared to the other two scenarios. The soil moisture storage of the GI scenario was also less than that of the NH scenario. But by adding the surface storage capacity, the sum approximately matched the soil moisture storage of the NH scenario, both in magnitude and temporal pattern.

Correspondingly, the BL scenario had the lowest ET amounts because much of the rainfall was converted to surface discharge. The GI scenario, however, produced an ET temporal pattern that more closely matched the NH scenario in magnitude and variation.

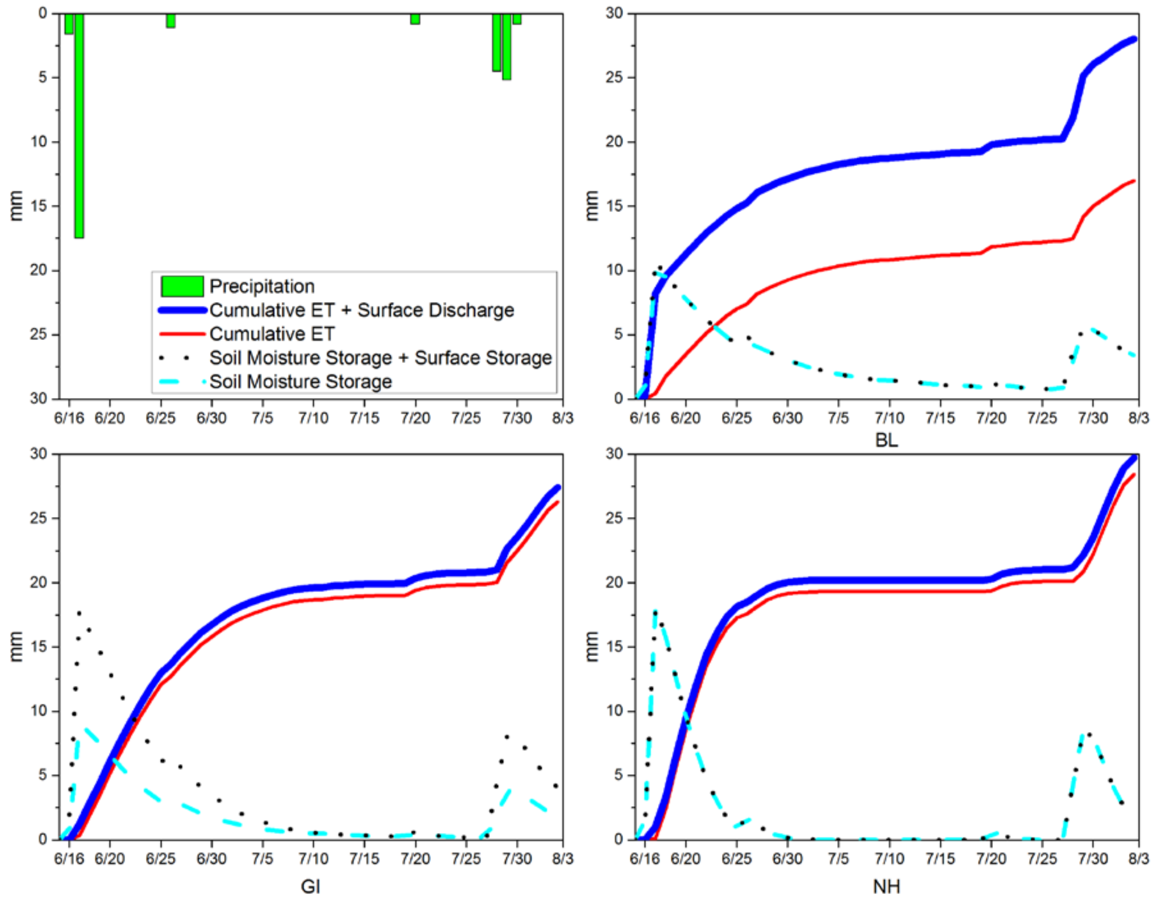

Fig. 24. Daily soil moisture storage, surface storage, ET, and surface discharge simulated by the updated SWMM for the average precipitation year (2014).

Both GI and NH scenarios have much lower surface discharge amounts compared to the BL scenario. Notably, the GI scenario had the largest final moisture storage incorporating both the surface and the soil layers, followed by the BL scenario and then the NH scenario. Conversely, the NH scenario had the largest sum of ET and surface discharge, followed by the BL scenario and then the GI scenario.

Similar relationships among the three scenarios were found when results were summed to the hourly scale (Fig. 25). The BL scenario had the least total stored rainwater. Adding the surface storage, the GI scenario had a comparatively close total moisture storage to the

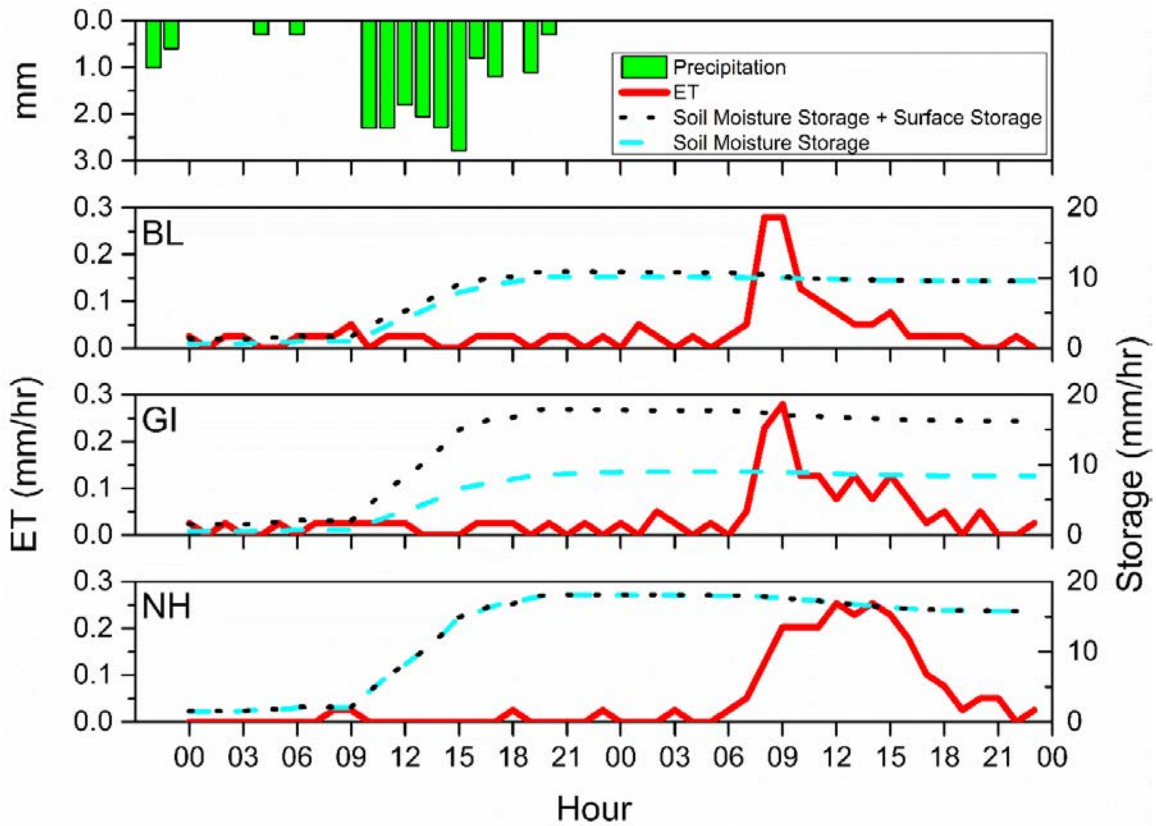

Fig. 25. Hourly surface storage, soil moisture storage, and ET rates simulated by the updated SWMM from June 17th to June 18th in the average precipitation year (2014).

NH scenario, in terms of magnitudes and temporal variations. Notably, the NH scenario had larger ET amounts during the second day (June 18th). However, unlike the NH scenario, the ET volumes in the GI scenario reached a peak then decreased. Also, the magnitude of ET volumes in all three scenarios are much less than those of the final storage volumes.

4.4 Discussion

4.4.1 The Role of ET within the Water Budget

As shown from the results, ET is the dominant component of the water budget in this semiarid climate case study, which is consistent with previous studies in different climates.

Specifically, in the NH scenario, ET accounts for 82-95% of the total water budgets, which is higher than the averages of the previous studies in natural watersheds that were mostly conducted in wetter climates (Table 5). However, in a similar semiarid climate with an annual precipitation of 331-477 mm, ET was estimated about twice as the annual precipitation on land (Mariotti et al. 2002).

In the BL scenario, the ET fraction of the total water budgets is 50-53%. It is hard to directly compare this result with other studies, as the range of urban ET ratios often vary

Table 5. A summary of ET fraction of water budget from previous studies.

ET Fraction of Water Budget	Reference
62-73% (Global Means)	(Douville 1998)
65-74% (Three Watersheds in Illinois, U.S.)	(Arnold and Allen 1996)
61-63% (Illinois, U.S.)	(Yeh et al. 1998)
57-74% (Four Watersheds in Arctic Region)	(Su et al. 2006)
76% (Average of Three Watersheds in South America)	(Su and Lettenmaier 2009)
73-89% (Taihu, China)	(Zhao et al. 2013)
25-66% (11 Major Drainage Areas of Canada)	(Wang et al. 2014)
>60% (Five Basins of China)	(Yao et al. 2014)
44-91% (Northern Eurasia)	(Liu et al. 2015)
64% (Ohio, U.S.)	(Chenevey 2013)
56% (Australia)	(Barron et al. 2013)
59% (Iowa, U.S.)	(Holman-Dodds et al. 2003)

widely due to a complex combination of local factors, including climate, imported water, urban heat island effect, plant cover, city density, designs of buildings and street canyons, and etc. But it still falls within the general range of the previous studies (Table 6).

In the GI scenario, annual ET amounts are 75-79% of the total water budget, which, similar to the NH scenario, is higher than other estimates in the wetter climates. For example, 32% is the fraction of ET within the total water budget after GI is applied in Malmö, Sweden (Villarreal et al. 2004), 71% after applying the modified household infiltration-based infrastructure and retention ponds in Trier, Germany (Keßler et al. 2012), 44% in an infiltration-based GI applied catchment in Iowa City, Iowa (Holman-Dodds et al. 2003), and 48% predicted for a GI scenario in Cincinnati, OH (Chenevey 2013).

To sum up, ET is the dominant component of the water budget in the climate of the study area. This is consistent with a previous study (Trinh and Chui 2013), in which ET was also found to be the greatest contributing factor for all scenarios including predevelopment, postdevelopment, and a restoration scenario using green roofs and bioretentions, even if ET was only measured as 5% of its own total water balance of a plot-scale green roof. However, the ET fractions of the water budget in this study are mostly higher than other studies in wetter climates. ET accounts for 50-53%, 75-79%, and 82-95% of the water budgets in the BL, GI, and NH scenarios, which also matches other findings that an increase in vegetation coverage tends to raise the ET fraction of the urban water budgets (Yao et al. 2014; Yeakley 2014).

Table 6. A summary of ET fractions of urban water budget from previous studies.

ET Fraction of Water Budget	Reference
38% (Vancouver, Canada; oceanic climate)	(Grimmond and Oke 1986)
38% (Sweden; oceanic, humid, or subarctic climates)	(Grimmond and Oke 1986)
71% (Mexico City, Mexico; subtropical climate)	(Grimmond and Oke 1986)
34% (Hong Kong, China; humid subtropical climate)	(Grimmond and Oke 1986)
49% (Sydney, Australia; temperate climate)	(Grimmond and Oke 1986)
57% (Moscow, Russia; humid climate)	(Grimmond and Oke 1986)
61% (Canberra, Australia; oceanic climate)	(Mitchell et al. 2003)
66% (Perth, Australia; Mediterranean climate)	(Mitchell et al. 2003)
55% (Sunninghill, South Africa; subtropical climate)	(Mitchell et al. 2003)
33-66% (Rezé, France; oceanic climate)	(Dupont et al. 2006)
40% (Rezé, France; oceanic climate)	(Rodriguez et al. 2008)
34% (Nantes, France; oceanic climate)	(Rodriguez et al. 2008)
58-75% (Four cities in Australia; oceanic climate)	(Kenway et al. 2011)
36% (San Luis Potosí, Mexico; semiarid climate)	(Martinez et al. 2011)
48% (Cincinnati, OH, U.S.; humid climate)	(Chenevey 2013)
2-56% (Perth, Australia; Mediterranean climate)	(Barron et al. 2013)

4.4.2 Water Budget Changes Due to Development

Although multiple factors have a mixed influence in increasing or decreasing urban ET ratios, the overall effect of urbanization is to typically decrease ET amounts due to vegetation reduction. For example, in a study of 51 eastern U.S. watersheds (Dow and DeWalle 2000) ET was estimated to drop by 31% (compared to annual average) with 100% urbanization, by 25% (compared to predevelopment condition) in Leipzig, Germany (Haase 2009), by 22% (compared to predevelopment condition) with a 9% increase in imported water in the Mill Creek watershed in Cincinnati, OH (Chenevey 2013), and by 23% (compared to predevelopment condition) in the Qinhuai river basin, China (Hao et al. 2015). In this study, the BL scenario experienced a decrease of ET by 38-45% from the NH scenario, which is slightly higher than estimates by the above studies.

Furthermore, the BL surface runoff in this study was 274-778% greater than the surface runoff in the NH scenario. This increase is greater than most studies in wetter climates, like the 182% increase demonstrated in Leipzig, Germany (Haase 2009) and the 128% increase in Cincinnati, Ohio (Chenevey 2013). This may indicate that the water budget may experience more severe changes after development in semiarid climates than in wetter climates.

4.4.3 Water Budget Restoration by GI

The potential to restore the natural water budget depends on the type of GI selected. Different GI types can target different parts of the water budget. For example, in an Iowa City, Iowa study (Holman-Dodds et al. 2003), infiltration-based GI was estimated to raise a catchment's groundwater recharge by 100%, making it close to that of predevelopment conditions. However, the GI in the Iowa City study was only able to increase ET by 8% compared to the developed condition, which was an increase of 3% of the total water budget (from 41% to 44%). In a study of a catchment in Cincinnati, OH (Chenevey 2013), a GI plan incorporating rain barrels, porous pavement, and green roofs was estimated to raise ET amounts by 19% compared to the developed condition, but the ET fraction (48%) of the total water budget did not show a significant change compared to the developed scenario without GI (40%).

In this study, the GI selected increases annual ET amounts by 46-52% compared to the BL scenario, which is 23-26% of the water budget. As the GI selected were bioretention and green roofs, which both have significant ET potential, the results suggest GI designs can effectively raise the catchment's ET level to predeveloped levels. This is also supported by a study in Malmö, Sweden, which used green roofs, open channels, and detention ponds that have even higher ET potential. The GI was found to raise the catchment's annual ET amounts by 74% compared to the developed condition (Villarreal et al. 2004).

Overall in this study, the GI scenario showed a water budget closer to the NH scenario than the BL scenario for all types of precipitation years (dry, average, wet). The GI scenario reduces annual runoff by 64-82%, which is higher than other studies, like 19% in Malmö, Sweden (Villarreal et al. 2004), 33% in Cincinnati, OH (Chenevey 2013), or 35% in Iowa City, IA (Holman-Dodds et al. 2003). These may indicate that GI can more significantly affect the water budget in a semiarid climate than wetter climates.

4.4.4 Storage Space of GI

With storage space in soil and GI, moisture collected in winter months (December to February) can be stored to be used in the subsequent warmer months (Fig. 23). ET rates determine how fast the storage capacity will be recovered for the next precipitation events (Krebs et al. 2016; Voyde et al. 2010). When precipitation is larger than the loss, the storage of both GI and natural landscapes begins to be refilled. But if ET demand does not regenerate the storage space fast enough, the net stored moisture may accumulate even when precipitation is low, as in July (Fig. 23).

During the months having enough incoming precipitation in the study area (January, February, September, December) or when soil moisture storage is still large enough (March, April, August, October), the natural plants with highest ET potential, simulated by the NH scenario, generate higher ET amounts than the BL scenario with least vegetation cover and the GI scenario with lower ET potential (Fig. 23). On the other hand, the plants in GI will deplete the soil moisture storage faster, which allows them to take in more stormwater in subsequent precipitation events. During the months when soil moisture storage was not enough to support high ET rates (May, June, July, November), the storage layers of GI, especially the gravel layer in bioretention and the drainage mat in green roofs, provide extra stored moisture to maintain ET amounts even higher than the NH condition (Fig. 23).

GI can add extra storage space on the impervious surface in urban catchments to match the total storage space of the predevelopment condition (Fig. 24, Fig. 25). The extra storage of GI can create ET patterns very close to the predevelopment condition (NH scenario) at the event scale. Due to the lack of storage space in the developed condition (BL scenario), the ET amounts were 35% and 40% lower than GI and NH scenarios after 48 days (Fig. 24), and 8% and 26% lower than GI and NH scenarios after 2 days (Fig. 25). This indicates the importance of the storage space in determining the performance of GI.

To sum up, in restoring the water budget, one of the most important features of GI might be providing concentrated storage to match the soil storage of the predevelopment condition, as GI can significantly raise the annual ET amounts close to the level of the natural landscapes (Fig. 23, Fig. 24, Fig. 25) although the former's ET capacity tends to be lower than the latter (Fig. 23, Fig. 25).

4.4.5 Restoration Efficiency Index

The pervious areas in the BL, GI, and NH scenarios occupy 33%, 47%, and 95% of the corresponding total catchment areas, respectively. With the replacement of the impervious surfaces by pervious areas among the three scenarios, the water budgets change accordingly in the average precipitation year (2014) (Table 7). From the BL to NH scenario, the pervious areas replaces 62% impervious surfaces of the total catchment, which causes a 32% ET increase and a 32% decrease of surface runoff. The ratio of the percent of change for one water budget component versus the percent of impervious surfaces to be replaced is calculated. This ratio is proposed as a restoration efficiency index (REI) to evaluate the efficiency of different types of GI to restore the water budget to the predevelopment condition, which can be written as follows:

$$REI_i = \frac{(h_i' - h_i)}{Imp - Imp'} \ or \ \frac{(h_i - h_i')}{Imp - Imp'}, \tag{12}$$

where i represents the water budget component (ET, surface discharge, etc.), REI_i is the restoration efficiency index corresponding to a certain water budget component, h_i is

Table 7. The restoration efficiency indices for ET and runoff by the increase of pervious areas within the catchment in the average precipitation year (2014).

Transitions	Percent of Replaced Impervious Area of the Total Catchment	Percent of ET Increase of the Total Water Budget	ET Restoration Efficiency	Percent of Runoff Decrease of the Total Water Budget	Q Restoration Efficiency
BL->NH	62%	32%	0.52	32%	0.52
BL->GI	14%	26%	1.86	29%	2.07

the percent of the component within the water budget of the development condition (BL scenario in this study), and h_i' is the percent of the component within the water budget of the restoration condition (GI and NH scenario in this study), *Imp* is the imperviousness percentage of the development condition (BL scenario in this study), and Imp_i is the imperviousness percentage of the restoration condition. To make *REI* positive, $(h_i' - h_i)$ is used when the percent of water budget component (like ET) of the development condition is lower than that of the restoration condition; $(h_i - h_i')$ is used when the percent of water budget component (like discharge) of the development condition is higher than that of the restoration condition. In this study, the development condition (BL scenario) was assumed to have lower ET and higher discharge than the restoration conditions (GI and NH scenarios).

From the BL to the GI scenario, a 26% increase of ET and a 29% decrease of runoff of the total water budget, however, only requires 14% impervious surfaces replaced within the total catchment. Therefore, the REI of GI in terms of restoring ET and runoff patterns from the development condition are 1.86 and 2.07. This means that replacing 1% of impervious areas by GI within the total catchment could bring in 1.86% increase of ET and 2.07% decrease of runoff within the total water budget from the development condition.

If assuming the natural landscape as one type of GI, then the native vegetation has an REI of 0.52 to restore ET and surface runoff patterns from the developed condition

(Table 7). This means that replacing 1% impervious areas by natural landscape within the total catchment could bring in a 0.52% increase of ET and a 0.52% decrease of runoff of the total water budget. This indicates that to gain the extra restoration effect by choosing the traditional landscape may require retrofitting a much larger built city space.

Therefore, the proposed restoration efficiency index can be used as a way to evaluate the restoration efficiency of different GI choices per unit area. To replace 1% impervious areas within the catchment, GI is 260% and 301% more efficient than traditional landscaping in terms of the restoration of ET and surface runoff in this climate.

A sensitivity analysis was conducted to test the sensitivity of REI to the imperviousness percentages of the BL condition. The ratio between the REI of the GI scenario and the REI of the NH scenario was calculated for different levels of imperviousness percentages of BL scenario (Fig. 26). When the imperviousness percentage of BL scenario is lower than 10%, the surface discharge of the BL scenario of this study can be lower than that of the NH scenario, and ET of the BL scenario can be higher than that of the NH scenario, which lead to negative REI (not shown in Fig. 26). This may be due to the higher infiltration capacity and surface storage created by certain existing vegetation swales built in the BL scenario, compared to the grassland setting in the NH scenario. When the imperviousness percentage of the BL scenario is larger than 10%, the ratios of REI of GI scenario to REI of NH scenario are consistently higher than 2.32 for restoring/increasing ET, higher than 3.81 for restoring/reducing discharge. This demonstrates the advantage of using GI to restore water budget in terms of converting less impervious areas, compared to using natural landscapes. This advantage of reducing discharge by GI is the most significant (above 6 times larger than the average) when the imperviousness percentage of BL scenario is as low as 10%.

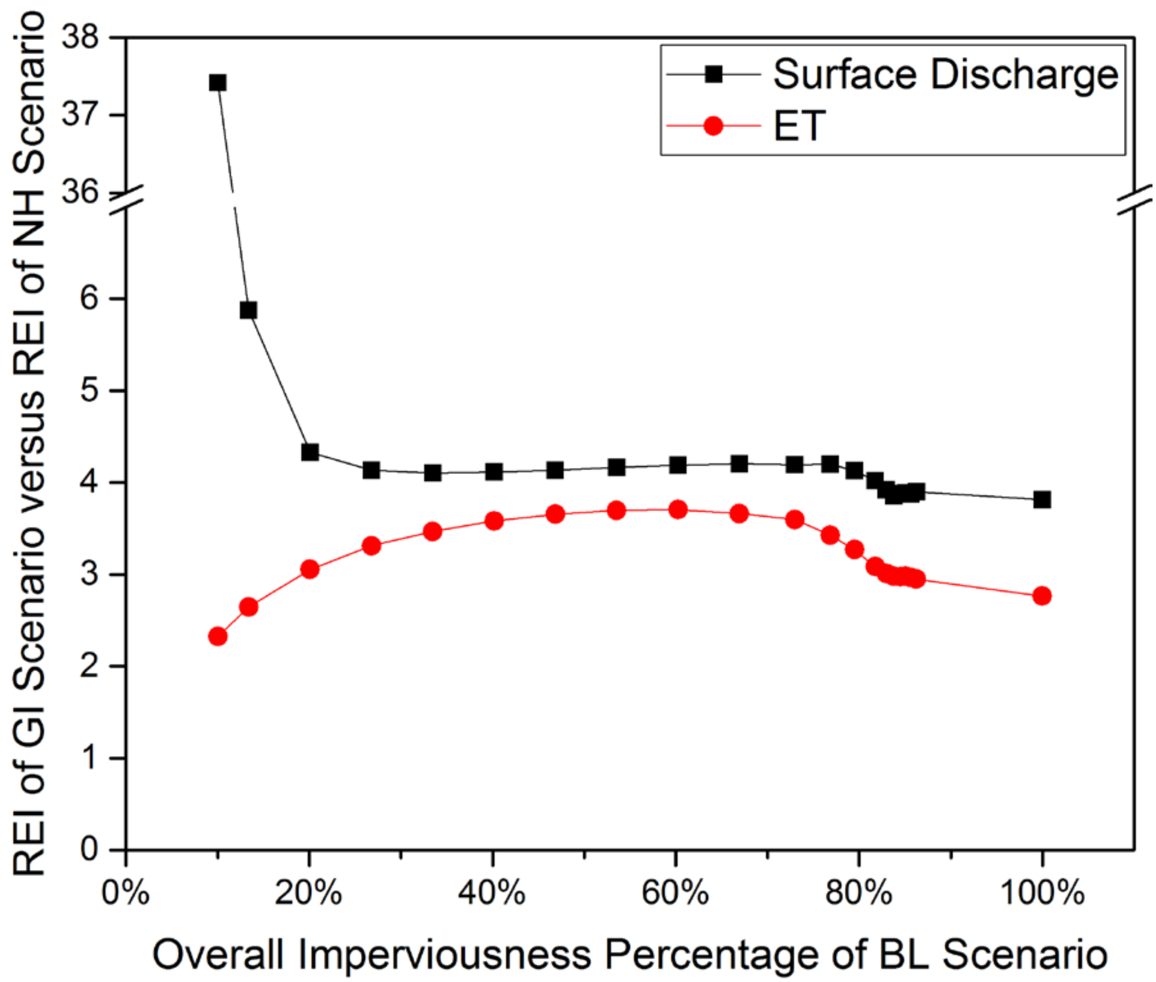

Fig. 26 The ratio of REI of GI scenario versus REI of NH scenario for the yearly ET and surface discharge, plotted against the overall imperviousness percentage of BL scenario.

This might be because when the catchment is highly pervious, adding more natural landscape will not reduce the surface discharge significantly anymore; but GI can further reduce the discharge by a larger degree than the natural landscape, which leads to a much larger REI in GI scenario than in NH scenario. After the imperviousness percentage of BL scenario exceeds 20%, GI shows a stable advantage over natural landscapes in terms of the necessity to convert less impervious areas in the catchment.

4.4.6 Water Budget Restoration Coefficient

A water budget restoration coefficient (WBRC) is proposed to evaluate the overall restoration effect of different GI plans in restoring the water budget, which is calculated as follows:

$$WBRC = 1 - \sum(f_i|f_i' - f_i|), \tag{13}$$

where f_i is the percent of the component within the water budget of the predevelopment condition (NH scenario in this study), and f_i' is the percent of the component within the water budget of the studied condition. The range of this coefficient should be above zero and equal or less than one. Larger values mean that the studied water budget is closer to the predevelopment water budget, and that the infrastructure or land covers have a high overall restoration effect.

The WBRC of the BL, GI, and NH scenarios for the dry, average and wet precipitation years were calculated using Equation (13), assuming the NH scenario was the predeveloped condition (Table 8). The GI scenario has coefficients more close to one than the BL scenario in any water year, which is consistent with the previously reported results indicating GI's potential to restore the natural hydrology of the catchment. More generally, the coefficient can be used to determine the degree of the water budget restoration by GI. As an example, the WBRC were calculated for three other studies based on their published data (Table 8). In the GI scenario, WBRC of the dry and average years in this study were higher than WBRC of all other studies, which indicates that the GI applications in these two cases reached better restoration effects. From BL scenario to GI scenario, WBRC of the dry, average, and wet years of this study have increased by 36%, 31%, and 41%, respectively, compared to 4% and 9% in the last two studies, respectively. This further

Table 8. The water budget restoration coefficients of three scenarios in three water years and estimated values for other studies.

	BL	GI	NH
Dry Year	0.69	0.94	1.00
Average Year	0.72	0.94	1.00
Wet Year	0.58	0.82	1.00
Villarreal et al.,2004	-	0.85	1.00
Chenevey, 2013	0.84	0.87	1.00
Holman-Dodds et al.,2003	0.81	0.88	1.00

proves that GI has higher potential in restoring the water budget in the studied climate than other wetter climates. After GI was applied, the water budgets have achieved 94%, 94%, and 82% of the predevelopment state in the dry, average, and wet years, respectively.

WBRC can also be used to evaluate how far the urban water budget can be changed away from the predevelopment condition due to urbanization. In the BL scenario, the WBRC of this study are all lower than other studies in wetter climates. This may further support the inference as above that the water budget may vary more severely after development, and GI may be more effective to restore the water budget in semiarid climates compared with wetter climates. But when the precipitation becomes higher, the restoration effect may become closer to the wetter climate, like the case of the GI scenario in the wet year (Table 8).

4.5 Conclusion

A modified version of EPA SWMM was used to explore the effect of GI in restoring the urban water budget to predeveloped conditions in the semiarid climate of Salt Lake City, Utah. Three scenarios (BL, GI, and NH) were compared in three types of precipitation years (dry, average, and wet). Water budget variations among scenarios and water years were presented to analyze the effect and mechanism of GI to restore the catchment's natural water budget. Two coefficients were proposed and used to evaluate the efficiency of the watershed GI plan.

Based on the results, the GI scenario was shown to produce a water budget closer to the NH scenario than the BL scenario in all three types of precipitation years. ET is the dominant factor of the water budgets of all scenarios in all water years, as found in previous findings for other climates. In this study, ET accounts for 50-53%, 75-79%, and 82-95% of the water budget for the BL, GI, and NH scenarios, respectively, which are relatively higher than the ratios found by other studies in wetter climates. After development, the water budget results indicated a more severe change than other studies in wetter climates. Compared to the predeveloped condition, the surface discharge of the developed condition is raised by 274%, 769%, and 778% in the dry, average, and wet precipitation years, respectively; ET was reduced by 38%, 39%, and 45%, correspondingly. The results of the present study showed GI to be more effective in restoring the water budget in the semiarid climate than other climates. Compared to the predeveloped condition, GI annually reduces surface runoff by 64%, 82%, and 76% and restores ET amounts by 49%, 52%, and 46% for the dry, average, and wet precipitation years, respectively. Adding the extra storage on the urban surface might be one of the most important features of GI to restore the water budget in a semiarid climate, as it creates comparable total storage space as the natural landscape. Then the difference of ET rates due to the capacity of various plant species controls the time to recover the storage space for subsequent storms.

The proposed restoration efficiency index can be used as a way to evaluate the restoration efficiency of different GI choices per unit area. To increase 1% of the area within the catchment, GI can be 260% and 301% more efficient than traditional landscaping in terms of the restoration of ET and surface runoff in a semiarid climate. The proposed water budget restoration coefficient (WBRC) further supports the hypothesis that GI can restore the urban water budget close to the natural hydrology. Based on the proposed WBRC, the water budgets have been restored due to GI applications by 36%, 31%, and 41% to approach the predevelopment condition in the dry, average, and wet years, respectively. In other words, after GI was applied, the water budgets have achieved 94%, 94%, and 82% of the predevelopment state in the dry, average, and wet years, respectively. Comparison with other studies indicates the water budget may vary more severely after development, and GI may be more effective to restore the water budget in a semiarid climate than wetter climates. But when the precipitation amount becomes closer between the climate types, the restoration effect becomes less different. The restoration efficiency index and the water budget restoration coefficient provide a useful metric to evaluate GI plans for the goal of achieving the near-natural hydrologic response of a watershed.

CHAPTER 5

SUMMARY AND PERSPECTIVES

This dissertation presented the results of a study investigating the overarching question of GI impacts on the urban water budget in a semiarid climate. Field experiments and hydrologic modeling were conducted in a semiarid city, Salt Lake City, Utah, U.S.

This work created, for the first time, an ET observation dataset for the semiarid intermountain west of the U.S. Empirical parameters for Penman-Monteith ET methods including crop coefficients and surface resistances for green roofs were identified and calibrated for this region, also for the first time. Their values can be directly used for ET modeling of green roofs in similar climates. From the comparisons, the yearly-averaged parameters have achieved estimates of similar accuracy as the month-averaged parameters. This could reduce the difficulty of identifying monthly parameters and make these methods easier to use for estimating green roof ET rates. By comparisons with the observation dataset, the Penman-Monteith methods based on the yearly-averaged parameters can reach acceptable estimates with the coefficients of determination (R^2) as 0.58-0.91 for different surface covers.

The EPA SWMM was modified to be able to represent spatially heterogeneous ET rates in one catchment for up to six types of land covers, including bioretention, green roof, landscape, deciduous trees, coniferous trees, and water surface. This creates an updated platform for urban water budget modeling, as it removes the previous barrier that traditional hydrological models are not equipped with GI modules, and stormwater models can only have one type of ET representation across a catchment. Also, the EPA SWMM was modified to be able to operate using subdaily ET time series input, for the first time. The updated model was validated by comparing the ET estimates with the existing observation datasets with the coefficients of determination (R^2) as 0.57 for bioretention and 0.78 for green roof.

With the updated model, the last part of this work demonstrates the potential of restoring the predevelopment urban water budget by adopting GI strategies in a semiarid climate. Based on the proposed water budget restoration coefficient (WBRC), the water budgets have been restored due to GI applications by 36%, 31%, and 41% to approach the predevelopment condition. In other words, after GI was applied, the water budgets have achieved 94%, 94%, and 82% of the predevelopment state. The comparisons of water budgets among different scenarios indicate that the urban water budget can be altered significantly due to development, which suggests that the semiarid regions are much more sensitive to development than humid regions. Similarly, the results also indicate that GI can more effectively restore the urban water budget in this climate than humid regions. Therefore, careful planning and designs for future development are needed in semiarid urban areas. Although the urban water budget may be restored by GI from the quantity perspective based on this study, certain ecosystem components, processes, and functions may be seriously disturbed by urbanization and the changes of the water budget. To make sure those can be restored or the impacts of the disturbances can be mitigated, a development planning with a comprehensive evaluation of the major ecosystem responses is needed before the development starts, especially in semiarid regions.

The specific conclusions are summarized in the following three sections.

5.1 Green Roof's ET and Water Budget Observations

To study the hydrologic performance of green roofs in Utah's climate and to explore a better way to model green roof ET, an experiment using weighing lysimeter units was conducted in 2014 in Salt Lake City, Utah. The results confirmed that green roof ET rates can vary significantly between different roof covers and in different months. The annual averages of ET rates of the studied grass, sedums, and nonvegetated covers under well-watered conditions in this climate were found to be 2.69±1.69, 2.52±1.79, and 2.01±1.16, respectively, which are higher than studies in wetter climates. Three P-M-equation-related methods were tested to simulate the ET observations at two sites, while two methods based on crop coefficients and surface resistances achieved overall good fits. The annual averages of crop coefficients for the nonvegetated, sedum, and grass covers were calculated as 0.50±0.36, 0.57±0.36, and 0.57±0.32, respectively; while the annual averages of surface resistances were calculated as 1707, 480, and 399 s m^{-1}. The values for the sedums are close to other studies in wetter climates. Using the simplified annual-constant parameters to predict ET rates was proven able to achieve similar accuracy as using the more detailed monthly parameters. The estimated water budgets have shown ET as the major component for all three covers. As the ET fraction of the water budget is more than 88% for sedums and grass, they might not be able to sustain themselves without irrigation in this climate.

5.2 An Improved Water Budget Model and Its Meaning to the Semiarid Region

Targeting to improve the ET mechanism in stormwater modeling for a better evaluation of water budget changes due to GI implementations, a revised FAO-56 ET estimation framework (Allen et al. 1998) was merged into a modified EPA SWMM model. The SWMM model was then upgraded to be able to accept an input of the heterogeneous and subdaily PET time series data, which other simple methods like monthly averages or Hargreaves equation cannot represent. Based on the available data observed in Salt Lake City, Utah and other places with similar climates, the results of the ET estimates from the modified SWMM were reasonable. The updated ET routine was validated for bioretention (R^2: 0.57) and green roof (R^2: 0.78) areas by comparing with direct measurements. The importance of having matching PET inputs for the corresponding vegetated land covers was presented. If the lower green roof's PET time series was used as the PET input for landscapes, the latter's annual percolation and AET amounts were overestimated and underestimated by 38 mm/yr and 38 mm/yr, respectively, which were equivalent to a 406% increase of percolation and a 9% decrease of AET amounts. But if the PET inputs for the bioretention and the green roof were replaced by the higher landscape PET input, their annual AET amounts could be overestimated by 30 mm/yr (12%) and 45 mm/yr (19%), with an underestimation of runoff by 30 mm/yr (14%) and 44 mm/yr (19%), respectively. EPA SWMM currently can only accept PET inputs on a daily basis and does not take into account the moisture stress, so it was shown to give unrealistic estimates of ET at a subdaily scale and during the dry periods after the rain events. Furthermore, irrigation activities in dry climates will greatly increase both the ET and percolation amounts, which could further magnify the errors mentioned above with regards to the simplified modeling of the ET process in SWMM. Therefore, the oversimplified representation of the ET process can generate large errors for the water budget estimation, especially in dry climates. It is thus recommended that the more physically based ET mechanism with higher spatial and temporal resolutions proposed in this study is necessary.

This method mainly requires four weather datasets including solar radiation, air temperature, wind speed, and relative humidity, which are often available from weather stations. The parameters provided by this study could be used for other places with similar climates. Lack of observational data for several key parameters, like the stomatal conductance or resistance, for the species may lead to uncertainty in application of the modified model; therefore, more studies are required to continue to improve the P-M scheme for GI in the modified SWMM. Although not common in the urban water engineering literature, studies applying approaches and measuring instruments from biology can be effectively used to fill in the gaps identified in this study.

5.3 The Impact of GI on Restoring the Urban Water Budget Model

To explore the effect of GI in restoring the urban water budget to the predevelopment hydrology in a semiarid climate, a case study was conducted in an urban catchment in Salt Lake City, Utah using the modified SWMM. Three scenarios (BL, GI, and NH) were compared in years with three different precipitation amounts (dry, average, and wet). The water budget variations among scenarios and types of years were studied to analyze the effect and mechanism of GI to restore the catchment's natural hydrology. Two coefficients were proposed and used to evaluate the efficiency of this process.

Based on the results, the GI scenario showed a water budget closer to the NH scenario than the BL scenario in all years. ET is the dominant factor of the water budgets, as found in previous studies and for other climates. ET accounts for 50-53%, 75-79%, and 82-95% of the water budget for the BL, GI, and NH scenarios, respectively, which are higher than the ratios found in wetter climates. After development, the water budget showed a more severe change than other studies in wetter climates. Compared to the predevelopment condition, the surface runoff of the postdevelopment condition is raised by 274%, 769%, and 778% in the dry, average, and wet years, respectively, while ET was reduced by 38%, 39%, and 45%, correspondingly. GI was shown to be more effective in restoring the water budget in this climate than other climates. Compared to the predevelopment condition, GI annually reduces surface runoff by 64%, 82%, and 76% and restores ET amounts by 49%, 52%, and 46% for the dry, average, and wet years, respectively. Adding the additional storage on the urban surface might be one of the most important features of GI to restore the water budget in this climate, as it creates comparable total storage space as the natural landscape. Then the difference of ET rates due to the capacity of various plant species governs the turnover time of recovering the storage space for the next storms.

The proposed restoration efficiency index can be used as a way to evaluate the restoration efficiency of different GI choices per unit area. To increase 1% of the area within the catchment, GI can be 260% and 301% more efficient than traditional landscaping in terms of the restoration of ET and surface runoff in this climate. The proposed WBRC further supports the hypothesis that GI can restore the urban water budget close to the natural hydrology. Based on the proposed WBRC, the water budgets have been restored due to GI applications by 36%, 31%, and 41% to approach the predevelopment condition. In other words, after GI was applied, the water budgets have achieved 94%, 94%, and 82% of the predevelopment state. It has been calculated for other studies using the published data. The comparison further demonstrated the water budget varies after development, and GI may be more effective to restore the water budget in the semiarid climate than wetter climates. But when the precipitation amount becomes closer, the restoration effect in dry climates may become closer to the wetter climate. The WBRC can be expected to be used in other cases to determine the restoration effect of GI applications for restoring the watershed natural hydrology.

5.4 Limitations and Future Work

This study serves as one of the first explorations of water budget response to GI applications in semiarid climates, and the first to investigate the potential for GI to restore the urban water budget. Based on the experience accumulated through the experiments and modeling, certain points can be improved in future work.

The average ET ranges of green roofs of the studied climate have been identified by this work. Field experiments can be further conducted to determine the appropriate species, irrigation amounts, and storage depth for this climate. A controlled experiment composed of several repeats with varying levels of those factors can be conducted to answer this question.

A better groundwater model for updating the soil moisture balance in the unsaturated soil zone is needed. The SWMM modified in this study can be mainly used for dry regions where the water table is deep enough that it will not significantly interact with the surface recharge. But for the mesic environments with shallower water tables, the contribution of

ET from the deeper saturated soil zone will be significant. A three-layer groundwater model to separately represent the rooting zones for ET estimation (using Penman-Monteith framework), the unsaturated soil zone, and the deeper saturated soil zone seems necessary.

This work mainly considers rainfall as the major source of water supply, to separate the influence of the climate from the anthropogenic controls to the water budget. Although irrigation may significantly raise the ET rates for an urban site (Spronken-Smith et al. 2000), its impact to the fraction of ET within the water budget at a city scale is still unclear. An experiment in Chapter 3 shows that the response of the ET fraction within the water budget to irrigation addition may vary depending on the PET capacity of different plant types. Therefore, to study the irrigation impacts, there is a further need to incorporate a new irrigation module in SWMM, which should only allow irrigation on the pervious areas.

The weather data used for Chapter 3 and 4 were assumed as homogenously distributed for the whole catchment, as the area of the catchment is small. But difference may still exist for most meteorological variables due to blocking and topography. New measurement tools or micrometeorological models are needed to better reflect the spatial variability of those variables and improve the ET estimates spatially.

Another study comparing the tradeoffs of selecting different combinations of types, amounts, locations, and designs of GI is necessary for guiding the future planning of GI at the watershed scale for different climate regions.

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