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Potential of green infrastructure to restore predevelopment water budget of a semi-arid urban catchment

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ABSTRACT

This paper presents a study of the potential for green infrastructure (GI) to restore the predevelopment hydrologic cycle in a semi-arid urban catchment. Simulations of stormwater runoff from a 0.11-km^2 urban catchment in Salt Lake City, Utah, USA for predeveloped (Natural Hydrology, NH), developed (Baseline, BL), and developed with GI (Green Infrastructure, GI) conditions were executed for a one-year period. The study was repeated for a relatively dry year, wet year, and an average year based on precipitation amounts in the year. Bioretention and green roofs were chosen for the GI plan. Results showed that the water budget of the catchment with the GI plan implemented more closely matches the NH water budget compared to the BL scenario, for all three years (dry, wet, average). The BL and GI scenarios showed more significant modifications to the water budget than what has been found by studies in humid climates. Compared to the BL condition, GI annually reduces surface runoff by 35%, 45%, and 43% and restores evapotranspiration by 18%, 19%, and 25% for the dry, average, wet years, respectively. Based on the introduced water budget restoration coefficient (WBRC), the water budget of the study catchment was restored by the GI plan to 90%, 90%, and 82% of the predevelopment state in the dry, average, and wet years, respectively. By comparing the WBRC estimated for other studies, it is further inferred that the water budget is more significantly affected by development and GI restoration in semi-arid than humid climates, but the differences lessen as the precipitation amount increases.

1. Introduction

Urbanization alters the water budget due to the removal of native vegetation, alteration and compaction of soils, building of impervious surfaces, changes in water use, and introduction of water diversions (Whitford et al., 2001; Pauleit et al., 2005; Shuster et al., 2005; Claessens et al., 2006; Powell et al., 2008; Scalenghe and Marsan, 2009; Jacobson, 2011; Guan et al., 2016; Yao et al., 2016). Such changes lead to a complicated mixture of modifications to the hydrologic cycle across a range of spatial scales. Surface runoff in most watersheds is observed to increase with urbanization (Rose and Peters, 2001; Weng, 2001; Lee and Heaney, 2003; Haase, 2009; Boggs and Sun, 2011; Zhang et al., 2013; Wu, 2015), while changes to other water budget components have been reported to typically be reduced, such as precipitation (Rosenfeld, 2000; Shepherd, 2006; Kaufmann et al., 2007; Hand and Shepherd, 2009), groundwater recharge (Lerner, 1990, 2002; Foster et al., 1994; Rose and Peters, 2001; Zhang and Kennedy, 2006; He et al., 2009; Jeppesen et al., 2011; He and Hogue, 2012; Hibbs and Sharp, 2012; Barron et al., 2013), baseflow (Brun and Band, 2000; White and Greer, 2006; Jacobson, 2011; Nie et al., 2011), and evapotranspiration (ET) (Oke, 1979; Grimmond and Oke, 1986; Balling and Brazel, 1987; Dow and DeWalle, 2000a; Rose and Peters, 2001; Dimoudi and Nikolopoulou, 2003; Gober et al., 2009; Haase, 2009; Jeppesen et al., 2011; Ramier et al., 2011; Shields and Tague, 2012; Wijesekara et al., 2012; Barron et al., 2013; Bijoor et al., 2014; Gwenzi and Nyamadzawo, 2014). However, the magnitude and direction of the water budget component modifications are difficult to predict given the complexities of the urban system (Burian and Pomeroy, 2010).

Such alterations to the hydrologic cycle can negatively impact the urban ecosystem and downstream areas. Increased runoff, for example, is directly connected to a wide array of environmental stressors (Hasse and Lathrop, 2003), such as flood risk (Liu et al., 2006; Haase, 2009; Du et al., 2012; Rutland and Dukes, 2012; Wijesekara et al., 2012), sediment erosion and transport (Nie et al., 2011), stream quality degradation (Interlandi and Crockett, 2003; Foley et al., 2005; Astaraie-Imani et al., 2012; Zgheib et al., 2012), aquifer pollution (Lerner and Barrett, 1996; Chisala and Lerner, 2008; Hibbs and Sharp, 2012), waterborne diseases (Vörösmarty et al., 2000; Narain, 2012), acidification of water

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bodies (Kelly et al., 2011; Xiao et al., 2012), and aquatic species loss (Gillies et al., 2003). 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 guide the configuration and operation of stormwater management systems in cities.

To develop plans to mitigate the adverse ecosystem impacts due to urbanization, a new international trend of pursuing the goal of 'near-natural' stormwater management has emerged (Göbel et al., 2004; Göbel and Coldewey, 2013). The concept of 'near-natural' aims to replicate the quasi-natural local water balance so as to preserve the local ecosystem's integrity (Keβler et al., 2012; Walsh et al., 2016). This trend is consistent with the efforts of using green infrastructure (GI) to restore the predevelopment hydrologic cycle, promoted by the United States Environmental Protection Agency (EPA) (USEPA, 2000), and aligns with sustainable design goals incorporated into the Envision™ sustainable infrastructure rating system (Envision) (http://sustainableinfrastructure.org/, last accessed on July 2nd, 2016) and the Leadership in Energy and Environmental Design (LEED) rating system (http://www.usgbc.org/leed, last accessed on July 2nd, 2016).

For stormwater management, GI is designed to reduce the quantity and improve the quality of runoff by adding storage (often pervious) with the capacity to capture, evapotranspire, and infiltrate stormwater. Compared to predevelopment landscapes, the vertical storage capacity of GI in cities compensates for the lost area of natural surface storage. By expanding storage in the vertical direction and incorporating water conservation, GI seeks to efficiently (in terms of land area) achieve stormwater runoff management and environmental benefits of natural landscapes. Most GI-related studies and applications have focused on runoff (Booth et al., 2004; Culbertson and Hutchinson, 2004; Simpson, 2007; Brown et al., 2009; Li et al., 2009; Alfredo et al., 2010; Burian and Pomeroy, 2010; Fassman and Blackbourn, 2010; Voyde et al., 2010a; DeBusk et al., 2011; Petrucci et al., 2013; Trinh and Chui, 2013; Ellis and Viavattene, 2014; Loperfido et al., 2014; Zahmatkesh et al., 2014; Ambrose and Winfrey, 2015; Jarden et al., 2016; Guan et al., 2015a, 2015b; Wella-Hewage et al., 2016) and groundwater recharge (Shuster et al., 2007; Moglia et al., 2010; Kidmose et al., 2015). A critical, yet often overlooked, water budget component addressed by GI is ET, because (1) ET controls the amount of available water for percolation (Ellis, 2013), and therefore affects runoff volumes and peak rates (Boggs and Sun, 2011; Sun et al., 2013; Walsh et al., 2016; Wong and Jim, 2015; Yang et al., 2016); (2) ET affects the urban heat island (UHI) intensity (Sailor, 1995; Alexandri and Jones, 2008; USEPA, 2008; Gober et al., 2009, 2012; Shashua-Bar et al., 2009; Krayenhoff and Voogt, 2010), and in turn the cooling costs and related energy consumption (Barrio, 1998; Kumar and Kaushik, 2005; Lazzarin et al., 2005; Levallius, 2005; Getter and Rowe, 2006; Takebayashi and Moriyama, 2007; Alexandri and Jones, 2008; Mitchell et al., 2008; USEPA, 2008; Fioretti et al., 2010; Gartland, 2010; Ouldboukhitine et al., 2011; Saadatian et al., 2013); (3) ET from green roofs generates cool air, which may give rise to strengthened street canyon flow and improved air quality near roads (Baik et al., 2012); (4) green spaces (increasing ET) provide space for plants and increase carbon sinks, especially in arid regions (Sun et al., 2011), and improves biodiversity (Currie, 1991); and (5) ET enhances atmospheric moisture, which may lead to enhanced precipitation under certain circumstances (e.g., semi-arid climates) (Eltahir, 1998; Schär et al., 1999; Shepherd and Burian, 2003; Koster et al., 2004; Burian and Shepherd, 2005; Jung et al., 2010; Seneviratne et al., 2010; Aragao, 2012; Spracklen et al., 2012; Taylor et al., 2012).

In accordance with the concept of integrated ecosystem management/stewardship (Falkenmark and Rockström, 2004; Chapin et al., 2009), there remains a need to evaluate the effect of GI in terms of recreating the near-natural water budget (Burns et al., 2012; Fletcher et al., 2013; Olszewski and Davis, 2013). Restoring ET and infiltration to predevelopment levels is a critical part of this goal, and is now represented in stormwater management design criteria specified in Envision and LEED. Therefore, the objective of this paper is to evaluate the potential of GI to restore the water budget of a developed area to an estimated predevelopment level in a semi-arid region.

2. Methods

2.1. Modeling framework

EPA SWMM 5.0.022 was selected as the modeling platform for this study, as it 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). The bioretention unit in EPA SWMM 5.0.022 was used to model both bioretention systems and green roofs for this study. The bioretention model in SWMM is composed of surface, soil, storage, and drainage layers. The storage layer was assumed to represent the drainage mat layer for green roofs. For both bioretention and green roofs, the layers were parameterized with appropriate hydraulic properties following guidance in the SWMM Users Manual (https://www.epa.gov/water-research/storm-water-management-model-swmm#downloads, last accessed on July 2nd, 2016). Compared to GI, landscape elements have surface and soil layers, with the latter represented by the unsaturated layer of the aquifer component of SWMM. The outflows from GI are specified in SWMM to drain onto landscapes or into storm drains.

The Penman-Monteith equation (Monteith, 1965) was used to estimate potential ET (PET) rates, following the standard practice (Kingston et al., 2009; Sherwood and Fu, 2014; Thompson et al., 2014). Parameters like albedo and surface resistances were set to represent the different GI and land surface covers (Feng and Burian, 2016). The water stress coefficient was set to convert PET rates to actual ET (ET_a) rates using the equation from the Food and Agriculture Organization of the United Nations, Paper 56 (FAO-56) (Allen et al., 1998; DiGiovanni et al., 2013). The moisture balance simulated by SWMM was used to calculate the water stress coefficient (Feng and Burian, 2016). Hourly PET and ET_a rates of six types of land covers including ponding water, bioretention, green roofs, turf landscapes, deciduous trees, and coniferous trees were estimated separately.

2.2. Study site

A small urban catchment (0.11 km^2) located on the campus of the University of Utah in northeast Salt Lake City (SLC), Utah, U.S.A. was chosen for this study (Fig. 1). SLC has a semi-arid climate (Bailey, 1979; Eubank and Brough, 1979; Bair, 1992; Russell and Cohn, 2012). From 1981 to 2010, the SLC average annual precipitation is 409 mm and the average annual air temperature is 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 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 approximately 0.899 cm/h; and its porosity is 0.459, while its wilting point and field capacity are 0.148 and 0.288, respectively (Merrell, 2013). The 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 aver-