ABSTRACT

Title of Thesis: THE STORMWATER RETENTION BENEFITS OF URBAN TREES AND FORESTS

Tuana Hilst Phillips, Master of Science, 2018

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The use of urban tree canopies as strategies to mitigate stormwater runoff is limited in part by a lack of empirically observed data. This thesis quantifies soil infiltration capacity in 21 forest patches in Baltimore, Maryland, and reports results from a meta-analysis on urban tree transpiration. Results show that the degree to which soil infiltration and tree transpiration functions reduce stormwater runoff depends on soil physical properties, tree characteristics, and management drivers. Yet, results conservatively estimate that Baltimore forest patch soils are capable of infiltrating ~68% of rainfall. In addition, urban trees transpire ~1.7 mm of water per day in the growing season or ~0.8 mm of water per day on an annual basis, an amount of water that equals approximately 26% of the annual rainfall in the Baltimore region. Thus, urban trees and forests impact urban hydrology and are an important component of stormwater green infrastructure in built environments.

THE STORMWATER RETENTION BENEFITS OF URBAN TREES AND FORESTS

by

Tuana Hilst Phillips

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Table of Contents

List of Tables

List of Figures

List of Abbreviations

SWGI	Stormwater Green Infrastructure
BMP	Best Management Practice
LID	Low Impact Development
SCMs	Stormwater Control Measures
WSUD	Water Sensitive Urban Design
CBP	Chesapeake Bay Program
K	Unsaturated Hydraulic Conductivity
BD	Bulk Density
SOM	Soil Organic Matter
CF	Coarse Fragments
VWC	Volumetric Water Content
VPD	Vapor Pressure Deficit

Chapter 1 Introduction

Urban Stormwater Problems

Excess stormwater runoff is caused by dramatic changes to landscapes that undergo urbanization. As urban areas grow, natural surfaces are replaced with impervious surfaces such as buildings and pavement. With an increase in impervious surface cover, the water balance of the urbanized landscape also changes. Rainfall infiltration into soil and vegetated surfaces is reduced, thereby increasing surface runoff from 10% to 30% (Hough, 1995; Pickett et al., 2011). Compared to pre-development conditions, evapotranspiration decreases from approximately 40% to 25% and groundwater levels diminish from 50% to 32%. Impervious surfaces direct about 43% of rainfall straight to storm drains and sewers (Hough, 1995; Pickett et al., 2011). Because rain is often channeled quickly and in large volumes to water bodies, overflowing at stream banks may cause localized flooding. If rainfall amounts surpass the urban drainage system's capacity to transport away water, flooding may also occur over developed surfaces (Apel et al., 2016; Rosenzweig et al., 2018). Further, excess stormwater alters the morphology of local streams and carries elevated concentrations of trash and pollution, therefore contributing to poor water quality and a decrease in biotic richness in nearby and connected waterways including streams, lakes, and estuaries (Booth & Jackson, 1997; Hood, Clausen, & Warner, 2007; Walsh et al., 2005). These urban stormwater problems will only become more pervasive as urbanization increases to accommodate 70% of the world's population living in developed areas by the year 2050 (UN 2008). In addition, the effects of climate change may lead to increases in stormwater runoff, as many locations are predicted to experience higher rainfall intensities in the future (Jefferson et al., 2017; Najjar et al., 2010).

Cities are also challenged by the way they were originally designed. Many cities were designed with a combined sewer system to collect and transport both sewage and rainwater for treatment. Although this kind of design is effective most of the time, large rainstorm events (e.g., 1-inch per hour storms) may cause the system to become overwhelmed and spill untreated sewage and stormwater into local water bodies (Abi Aad, Suidan, & Shuster, 2010). As most cities are continuing to grow and build impervious surfaces, combined sewer systems become more vulnerable to these kinds of situations. This is requiring cities to pursue updates to their existing grey infrastructure, such as their pipes, tunnels, pumping stations, and water treatment systems. For instance, the District of Columbia's Clean Rivers Project is building a series of tunnels to reduce the number of combined sewer overflows. As part of this project, the Anacostia River Tunnel was designed to reduce the number of combined sewer overflows into the Anacostia River by 98 percent (District of Columbia Water and Sewer Authority, 2012).

Stormwater Green Infrastructure Mitigation Strategies

Urban areas are also increasingly looking at green infrastructure solutions to reduce stormwater runoff and prevent combined sewer overflows (Askarizadeh et al., 2015; Golden & Hoghooghi, 2018; Lucas & Sample, 2015; Pennino, McDonald, & Jaffe, 2016). The growth of green infrastructure has coincided with the evolution of a diverse terminology that is used to describe green infrastructure strategies, including, but not limited to, low impact development (LID), best management practice (BMP), stormwater control measures (SCMs) and water sensitive urban design (WSUD) (Fletcher et al., 2014). In this thesis, I use the terms stormwater green infrastructure (SWGI) or BMPs to describe practices that reduce stormwater runoff and treat nutrients and pollution. Such practices mimic and restore the natural hydrology by augmenting infiltration, evapotranspiration, and harvesting of stormwater runoff (Askarizadeh et al., 2015; Davis, Hunt, Traver, & Clar, 2009; Hunt, Davis, & Traver, 2012). Examples of SWGI include centralized practices such as detention ponds and wetlands, which increase the residency time of rainfall before it is discharged to water bodies. Other decentralized practices, such as bioretention cells, rain barrels, green roofs, and pervious concrete infiltrate, collect, and reduce runoff at or near the source of rainfall. In addition to providing hydrological functions, many of these practices help to filter out pollutants by capturing sediment and nutrients such as nitrogen and phosphorus (Hathaway & Hunt, 2010; Jefferson et al., 2017).

Unlike grey infrastructure practices, SWGI are valued for other ecosystem services they provide in urban areas, such as uptake of carbon dioxide, cooling and cleaning of air, as well as cultural services such as improved well-being and opportunities for recreation (J. Ahern, Cilliers, & Niemelä, 2014; Gómez-baggethun & Barton, 2013). SWGI practices are increasingly being thought of as resilient strategies to adapt to climate change (Brink et al., 2016; Escobedo, Giannico, Jim, Sanesi, & Lafortezza, 2018; Giese et al. *in review*). Analyses have shown that green infrastructure practices are relatively inexpensive to install and maintain compared to conventional grey infrastructure systems (MacMullan & Reich 2007). For instance, Lancaster City in Pennsylvania found that employing a combination of LID and gray infrastructure practices would cost less than half the cost of using gray infrastructure alone (City of Lancaster, 2011).

Urban Tree Canopy and Stormwater

Urban trees also perform ecohydrological functions that reduce stormwater discharge, lessen the risk of flood, and improve water quality (Figure 1-1). Trees return water to the atmosphere through evaporation and transpiration. Further, tree canopies intercept rainfall and roots promote water infiltration and storage in the soil. However, the use of urban tree and forests as SWGI is limited, in part, by a lack of empirically derived performance data. Many studies in non-urban areas have contributed to a rich body of knowledge regarding the effects of trees and forests on the ecohydrology and water balance in those ecosystems (Bond et al., 2007; Ellison et al., 2017; Goldberg & Bernhofer, 2007; Osuch et al., 2009). Yet, this body of knowledge may not be directly transferable to urban areas due to the dissimilar conditions that trees are exposed to in the built environment, such as higher temperatures, compacted soils, and increased deposition of nitrogen (Arnfield, 2003; Asawa, Kiyono, & Hoyano, 2017; Law, Band, & Grove, 2004; Mccarthy, Pataki, & Jenerette, 2011). In addition, urban areas are spatially heterogenous landscapes that contribute to a range of environmental conditions that affect how urban trees and soils function (Cadenasso, Pickett, & Schwarz, 2007; Escobedo & Nowak, 2009; Pickett & Cadenasso, 2009). Unlike for non-urban areas, empirical studies of tree hydrology in developed areas have been limited to date and the amount of stormwater that urban trees remove through ecohydrological processes is not well established in the scientific literature (Berland et al., 2017; Kuehler et al., 2017). This therefore presents a challenge for the stormwater, urban planning, and watershed management sectors that require specific performance numbers to integrate new BMPs, such as increased urban tree canopies, into their watershed, flooding, and water quality designs and programs.

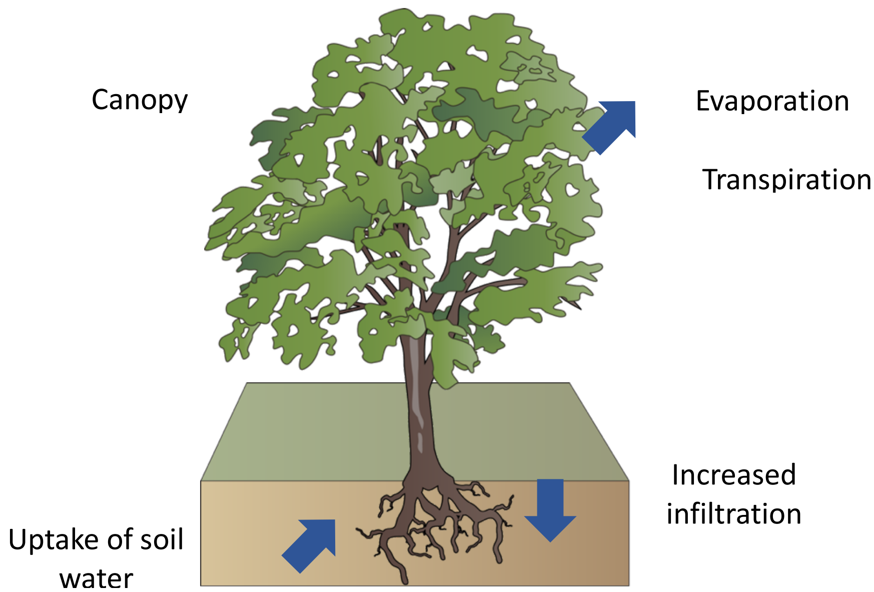


Figure 1-1 Ecohydrological tree functions that reduce stormwater runoff.
(Tree and soil images courtesy of the Integration and Application Network, UMCES)

Local Example

In the Chesapeake Bay region, urban stormwater is the fastest growing source of pollution to the Chesapeake Bay (Sparkman, Hogan, Hopkins, & Loperfido, 2017). Managers who are working to improve water quality in the Chesapeake Bay know that restoration will require a suite of BMPs that tackle stormwater and nonpoint source pollution from urban areas and other land uses. In 2016, the Chesapeake Bay Program (CPB), a regional partnership that works to improve the Bay’s health, approved a new Urban Tree Canopy Expansion BMP and Urban Forest Planting BMP to be simulated in a series of models that guide and credit Bay restoration efforts. The partnership also approved a new Urban Tree Canopy Land Use that will better capture the water quality benefits of existing urban tree canopy in their model. The loading rates and nutrient credits for the Urban Tree Canopy Land Use came from work by Justin Hynicka (Maryland Department of Natural Resources) and Dr. Marion Divers (University of Pittsburgh) in early 2016. Due to a limited number of studies in the literature, they had to rely on a water balance model to calculate the annual water and nutrient retention benefits from each of ecohydrological functions depicted in Figure 1-1. To do this, they had to make several assumptions and use data from naturally forested settings for some of their model inputs. An additional effort by an organized Expert Panel at the CBP was completed to determine the percentage of nitrogen, phosphorus, and sediment reduced by the urban tree expansion and urban forest planting BMPs (Law and Hanson, 2016). Their analysis, too, affirmed a lack of peer-reviewed studies that have specifically quantified the stormwater retention and water quality benefits of urban tree canopy. Therefore, the Expert Panel based the BMP nutrient reduction numbers off of the same modeling approach used by Hynicka and Divers. These two efforts quantified more general numbers of contributions by urban tree canopy and did not evaluate individual variations that may occur due to different soil, tree, and environmental drivers.

Context for this Thesis

This thesis is motivated by the knowledge gap identified by the CBP effort to incorporate an urban tree canopy land use in their Watershed Model and credit urban tree BMPs as strategies to clean up the Chesapeake Bay. Given the lack of empirically derived numbers on the stormwater and water quality benefits of urban trees and forests, the goal of this thesis is to generate new data that will contribute to this understanding. The focus is on soil infiltration capacity and tree transpiration as two ecohydrological functions that reduce stormwater runoff (Figure 1-1). Studies have shown that tree roots promote soil infiltration, percolation, and storage of stormwater runoff (Bartens, Day, Harris, Dove, & Wynn, 2008; Kuehler et al., 2017). However, as reported by Kuehler et al. (2017), no studies have yet quantified the amount of urban stormwater runoff that can be reduced by these urban tree functions that increase soil infiltration. In addition, water flux out of terrestrial ecosystems is dominated by transpiration from plants (Jasechko et al., 2013), yet urban areas have less vegetative cover compared to rural areas and little attention to date has been given to the role of tree transpiration in urban hydrology (Berland et al., 2017). Those studies that have assessed urban tree transpiration in situ have observed dynamic rates across temporal scales and different species (Berland et al., 2017; Cregg & Dix, 2001; Giraldo et al., 2015; Pataki et al., 2011; Peters et al., 2010; Riikonen et al., 2016; Wang et al., 2011). Therefore, in this thesis, I quantify soil infiltration capacity in urban forest patches and review the literature to quantify rates of urban tree transpiration. I also examine environmental drivers that affect soil infiltration and tree transpiration. The broader research questions are:

1. How much stormwater are urban forest patch soils capable of infiltrating?
2. What soil physical properties are the most important drivers of soil infiltration capacity?
3. How much water is transpired by urban trees?
4. What tree characteristics and management contexts are important drivers of transpiration?

Questions one and two will be addressed in Chapter Two, and questions three and four will be addressed in Chapter Three. In Chapter Two, I quantify the unsaturated hydraulic conductivity as an estimate of soil infiltration capacity in forest patches in Baltimore, MD (Question 1). Further, I explore soil physical properties as drivers of infiltration capacity (Question 2; Figure 1-2). I examine whether soil texture, percent of coarse fragments, soil organic matter, soil bulk density, and soil moisture affect infiltration capacity. Infiltration capacity is typically higher for sandy and coarser soil textures that have relatively large pore spaces and are therefore capable of conducting water at a faster rate compared to clay and other soil types with finer textures (Olorunfemi & Fasinmirin, 2011). In addition, organic matter enhances soil structure and increases porosity, thus promoting infiltration capacity (Boyle, Frankenberger, & Stolzy, 1989). Leaf litter and other debris in forested areas contribute to soil organic matter and the soil's ability to infiltrate stormwater runoff (Ossola et al., 2015). In contrast, soil compaction (i.e., high bulk density) physically reduces the amount of pore space in the soil and decreases the soil's ability to infiltrate water (Ossola et al., 2015; Yang & Zhang, 2011). Low bulk density and compaction values are often associated with high organic matter in the soil (Chaudhari et al., 2015).

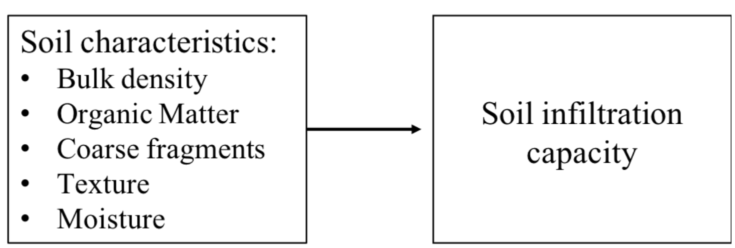


Figure 1-2 Soil physical characteristics that are evaluated in Chapter Two as drivers of infiltration capacity in urban forest patches.

In Chapter Three, I use a structured literature review and meta-analysis to gather average transpiration rates by trees in urban areas (Question 3). I also analyze whether tree characteristics, such as tree size, functional group, and wood structure, as well as the management context, drive transpiration rate (Question 4; Figure 1-3). Evergreen and deciduous species are two functional groups that may explain differences in transpiration rates. Evergreen trees retain their leaves all year round, whereas deciduous trees gain and lose their leaves once every year, leaving only the growing season to photosynthesize and fix carbon. These differences likely affect how these two types function ecohydrologically and how much water they use at different times of the year. In addition to assessing transpiration differences by functional group, I examine how urban tree transpiration may be affected by different wood structures or xylem anatomies. The xylem anatomy of conifer (i.e. softwood) trees is made up of tracheids, or relatively small, single-celled conduits of water. In contrast, the xylem anatomy of diffuse- and ring-porous trees (i.e. hardwoods) consists of both tracheids and vessel elements, the latter which are larger and multiple-celled conduits of water (Sperry et al., 2003; Peters et al., 2010). These types of wood structures have been shown to affect water transport in trees in non-urban areas (Catovsky, Holbrook, & Bazzaz, 2002; Sperry, 2003; Taneda & Sperry, 2008). The same is likely true for trees in urban areas. For example, Bush et al. (2008) and Litvak, Mccarthy, & Pataki (2012) report that urban trees with vessel elements use more water and exhibit higher maximum rates of tree transpiration.

Trees exist in a range of management contexts in urban areas. Trees in different management contexts are exposed to different environmental conditions and stressors that likely affect how much water they use. Trees along streets, for instance, experience high evaporative demands but often have compacted soils and less space for root growth and water access (Asawa et al., 2017; Riikonen et al., 2016; Whitlow, Bassuk, & Reichert, 1992). In contrast, the canopies of trees in parks have been shown to be relatively cooler than the canopies of street trees (Leuzinger, Vogt, & Körner, 2010) and intercept less long-wave radiation reflected from the surface (Kjelgren & Montague, 1998). Therefore, park trees may experience relatively lower evaporative demands. Soil moisture has also been shown to be higher in trees over turf environments compared to trees over paved surfaces (Cregg and Dix, 2001). These diverse micro-climates and soil conditions within the range of management contexts affect transpiration rates, but no study to date has specifically examined this. Thus, in Chapter Three, I use a meta-analysis to assess whether the management context can also explain differences in urban tree transpiration rates (Figure 1-3).

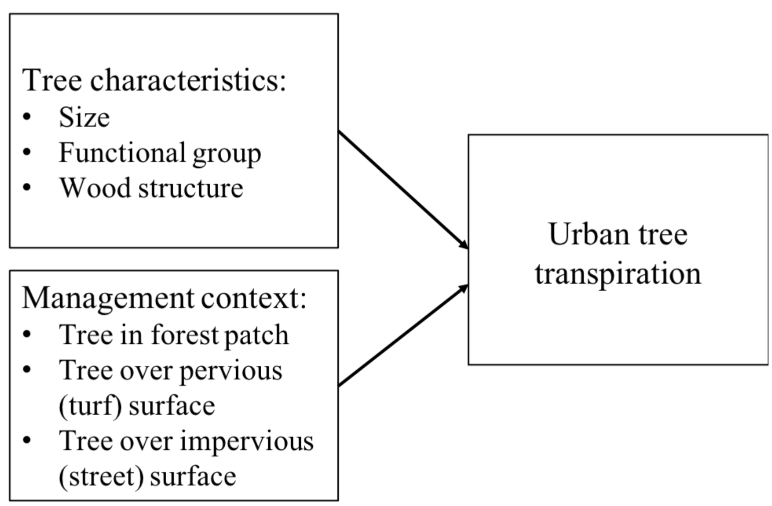


Figure 1-3: Tree characteristics and management contexts that are evaluated in Chapter Three as drivers of urban tree transpiration.

Research Approach

In Chapter Two of this thesis, I describe a study I conducted that empirically quantified infiltration capacity in 21 urban forest patches in Baltimore, Maryland, United States. I assessed infiltration capacity in urban forest patches by measuring the unsaturated hydraulic conductivity, a proxy for the ease at which water infiltrates the soil when it is not saturated. Based on unsaturated hydraulic conductivity measured at 62 locations, I estimated how much rainfall can be infiltrated by the surface soil in the urban forest patches. I also collected soil cores at each of the locations to assess the soil physical properties that affect soil infiltration capacity.

The study described in Chapter Three is focused on tree transpiration and broadens the scope to include urban trees in multiple management contexts, including trees over impervious surfaces (i.e., street trees), trees over pervious or turf surfaces, and trees in forest patches. I conducted a meta-analysis of urban tree transpiration rates from the peer-reviewed literature to better understand how much water urban trees transpire in the growing season and on an annual basis. Based on values and information reported in the same studies, I analyzed whether tree characteristics, such as tree size, functional group, and wood structure, as well as the management context, drive transpiration rate.

Chapter 2 The capacity of urban forest patches to infiltrate stormwater is influenced by soil physical properties and soil moisture

Abstract

Forest patches in developed landscapes perform ecohydrological functions that may reduce urban stormwater flows. However, urban forest patch contributions to runoff mitigation are not well understood due to a lack of performance data. In this study, we focus on the potential of urban forest patch soils to infiltrate rainfall by characterizing rates of unsaturated hydraulic conductivity (K) in 21 forest patches in Baltimore, Maryland. Soil bulk density, organic matter, soil moisture, percent of coarse fragments ($\geq$ 2 mm), and texture were evaluated at the same locations in order to assess drivers of K. K was significantly higher in soils with high sand content and related positively with the percent of coarse fragment material in the soil. We estimate that 68 percent of historic rainfall could be infiltrated by urban forest patch soils at the measured K rates. Continuous monitoring at one forest patch also showed that K is dynamic in time and influenced by antecedent soil moisture conditions. Although forest patches maybe less effective at infiltrating stormwater relative to designed green infrastructure practices, our results conservatively estimate that urban forest patch soils alone are capable of infiltrating the majority of rain storms of low to moderate intensities in the Baltimore region. Considering this ecohydrologic function, the protection and expansion of forest patches in Baltimore can make substantial contributions to stormwater mitigation in Baltimore.

1.0 Introduction

Urbanization alters the hydrologic cycle by creating impervious surfaces (e.g. roofs, parking lots, and roads) that reduce watershed infiltration capacity. Rainfall is channeled through storm drains and pipes and, as a result, urban watersheds experience high volume and "flashy" stormwater runoff following precipitation events (Askarizadeh et al., 2015; Shuster et al., 2005). These conditions cause receiving water bodies to exhibit high discharge peaks and degraded water quality, among other symptoms (Booth & Jackson, 1997; Hood, Clausen, & Warner, 2007; Walsh et al., 2005). In addition, urban areas are susceptible to flooding due to overflow at stream channels (fluvial flooding) or drainage failure and water accumulation over roads and surfaces (pluvial flooding) (Apel et al., 2016; Rosenzweig et al., 2018). Both types of flooding can lead to traffic disruption, property damage, and pose hazards and health risks to local residents (M. Ahern, Kovats, Wilkinson, Few, & Matthies, 2005; Qin, Li, & Fu, 2013).

Stormwater green infrastructure (SWGI) practices and other nature-based solutions (e.g., urban forests) mimic the natural hydrology by promoting infiltration, storage, and evapotranspiration of runoff (Askarizadeh et al., 2015; Golden & Hoghooghi, 2018; Kuehler et al., 2017). In addition to combating urban stormwater problems, these solutions are increasingly being thought of key strategies to building resilience and adapting to climate change in urban watersheds (Brink et al., 2016; Escobedo, Giannico, Jim, Sanesi, & Lafortezza, 2018; Giese et al. *in review*). Traditionally, SWGI practices have been designed in a centralized manner to increase the residency of stormwater before it is discharged to local water bodies. Since 2000, decentralized SWGI practices such as rain gardens, cisterns, and green roofs have become increasingly popular strategies to store, infiltrate, and treat a specific amount of rainfall at or near the source (Jefferson et al., 2017). The growth of such practices has coincided with a surge in research studies and knowledge about SWGI efficiency and effectiveness (Davis et al., 2009; Golden & Hoghooghi, 2018; Hunt et al., 2012; Jefferson et al., 2017; Lefevre et al., 2015).

Urban trees and forests are increasingly being looked to as a component of stormwater management portfolios in cities because urban forests contribute approximately 27% of urban land cover in the United States (Nowak, Noble, Sisinni, & Dwyer, 2001). However, in contrast with SWGI practices, urban trees and forests are not engineered to handle specific rainfall quantities. Moreover, the degree to which urban forests and their underlying soils decrease runoff is not well researched and quantified to date (Berland et al., 2017; Kuehler et al., 2017). As such, urban trees and forests are not always promoted as a best management practice by stormwater practitioners (Kuehler et al., 2017). One manner in which urban forests reduce stormwater runoff is through soil infiltration of rainfall. Forest patches reduce impervious cover and soils capture, filter, and slow the release of runoff by feeding groundwater supplies and stream baseflow. The rate at which water moves in soil is affected by soil physical properties such as bulk density (BD), soil organic matter (SOM), texture, and soil moisture (Gupta & Larson, 1979; Saxton & Rawls, 2006; Yang & Zhang, 2011). BD often serves as an indicator of soil compaction, a state that physically reduces the amount of pore space in the soil and impedes the soil's ability to infiltrate water (Kozlowski, 1999; Ossola, Hahs, & Livesley, 2015; Yang & Zhang, 2011). SOM from leaf litter and other debris in forests enhances soil structure and increases porosity, thus promoting infiltration (Boyle et al., 1989; Y. Chen, Day, Wick, & McGuire, 2014).

Unlike soils in undisturbed areas, however, urban soils are highly modified by humans and experience additional urbanization effects that influence their properties and therefore how they function ecohydrologically (Effland & Pouyat, 1997; Herrmann, Schifman, & Shuster, 2018; Herrmann, Shuster, & Garmestani, 2017; McDonnell et al., 1997; Pavao-Zuckerman, 2008). For instance, urban forests in the Northeastern U.S. have been shown to support high numbers of non-native earthworm species (McDonnell et al., 1997) that contribute to a decrease in the SOM in the O horizon (Burtelow, Bohlen, & Groffman, 1998). Additionally, compacted soils on urban construction sites in North Central Florida exhibited a 70-99 percent reduction in infiltration rates compared to non-compacted sites (Gregory, Dukes, Jones, & Miller, 2006). Herrmann et al. (2018) surveyed soil horizons in 11 cities and found evidence of widespread loss of B horizon soils that provide important functions for water drainage and soil water storage. Given the diverse soil physical conditions caused by direct and indirect effects from urbanization, study of the urban forest soil's properties is key to understanding the soil's ability to infiltrate stormwater runoff.

Urban forest soils likely contribute to hydrological processes in cities, but we know relatively little about their function (Kuehler et al, 2017). This is important because urban forest soils are being looked to as a key element of stormwater management in built environments, yet there is a need for better quantification and empirical studies (Law & Hanson, 2016; Berland et al., 2017; Kuehler et al, 2017). In this study, we evaluate the potential of urban forest patches in Baltimore, Maryland, USA to infiltrate stormwater by analyzing the soil unsaturated hydraulic conductivity

(K), a key process in regulating flow into unsaturated soils (Perkins, 2011). Further, we assess soil properties including BD, SOM, texture, percent of coarse fragment material, and soil volumetric water content as potential drivers of K. We further assess whether K differs between forest patches of different sizes that also relate to how they are managed.

2.0 Materials and Methods

2.1 Study Sites

Baltimore, Maryland, receives about 108 cm of precipitation each year, an amount that is evenly distributed throughout all months. The city has cold winters and hot, humid summers, averaging a low of 5.6º C in January and a high of 31.7º C in July. Two physiographic provinces make up the city. The Piedmont Plateau province has deep and well-drained upland soils with moderate slopes that overlay semi-basic, mixed basic, and acidic rocks. The Coastal Plain province consists of deep, well-drained upland soils that overlay sediments that are sandy, gravelly, or clayey in texture. The dominant soil types are Ultic Hapludalfs and Typic Hapludults in the Piedmont Plateau and Coastal Plain, respectively (USDA Forest Service https://www.nrs.fs.fed.us/ ef/ locations/md/baltimore/).

Forest patches are areas of tree canopy at least ~0.1 hectares in size with complex habitat structures that include understory shrubs, small trees, woody debris, and leaf litter (Avins 2013). In Baltimore, Maryland, USA, forest patches account for 34% of the city's tree canopy cover (Avins 2013). Twenty-one forest patches across Baltimore City were chosen as the study sites (Figure 2-1). These patches were chosen because they are part of an ongoing research and conservation effort (a partnership between Baltimore Green Space, a land trust organization, and researchers with the Baltimore Ecosystem LTER Study). They are predominately located in northern Baltimore and include 11 larger forest patches (15.62 ha ± 2.91) that are protected under city easements and 10 smaller forest patches (1.9 ha ± 0.52) that are nested within neighborhoods and overseen by local stewards and Baltimore Green Space. Although the term “forest patch” suggests one area of connected forest cover, in actuality some of the larger sites include two or more patches of forest, or are part of an even larger forested area that extends beyond the defined perimeter of the forest patch. The dominant tree species across all forest patches are native to eastern North America and include, *Liriodendron tulipifera*, *Quercus alba*, *Acer Rubrum*, *Quercus rubra*, *Fagus grandifolia*, and *Fraxinus Pennsylvanica* (Yesilonis and Baker, et al., *unpub. data*). On average, 7.7% of trees and 56.4% of groundcover species in the patches are non-native. The most common non-native groundcover species are vines: *Hedera helix*, *Lonicera japonica*, *Ampelopsis brevipedunculata,* and *Celastrus orbiculatus* (Yesilonis and Baker, et al., *unpub. data*).

2.2 Hydraulic conductivity (K) measurements as estimates of infiltration capacity

In the summer of 2017, we measured K at three locations per forest patch, with the exception of one smaller forest patch (Belvedere) where we took measurements at two locations. Locations for measurements in each forest patch were chosen based on locations of preliminary soil texture and BD data acquired by researchers who applied a systemic random design to sample across all forest patches (Yesilonis and Baker, et al., *unpub. data*). Digital elevation models were also used to guide the selection of locations to avoid areas of high slope. Sixty-two locations, in total, were sampled across all forest patches. At each location, three measurements of K were taken within a two m^2 area except for a few occasions in which an uneven slope led to taking measurements farther apart so that the infiltrometers sat on relatively flat areas.

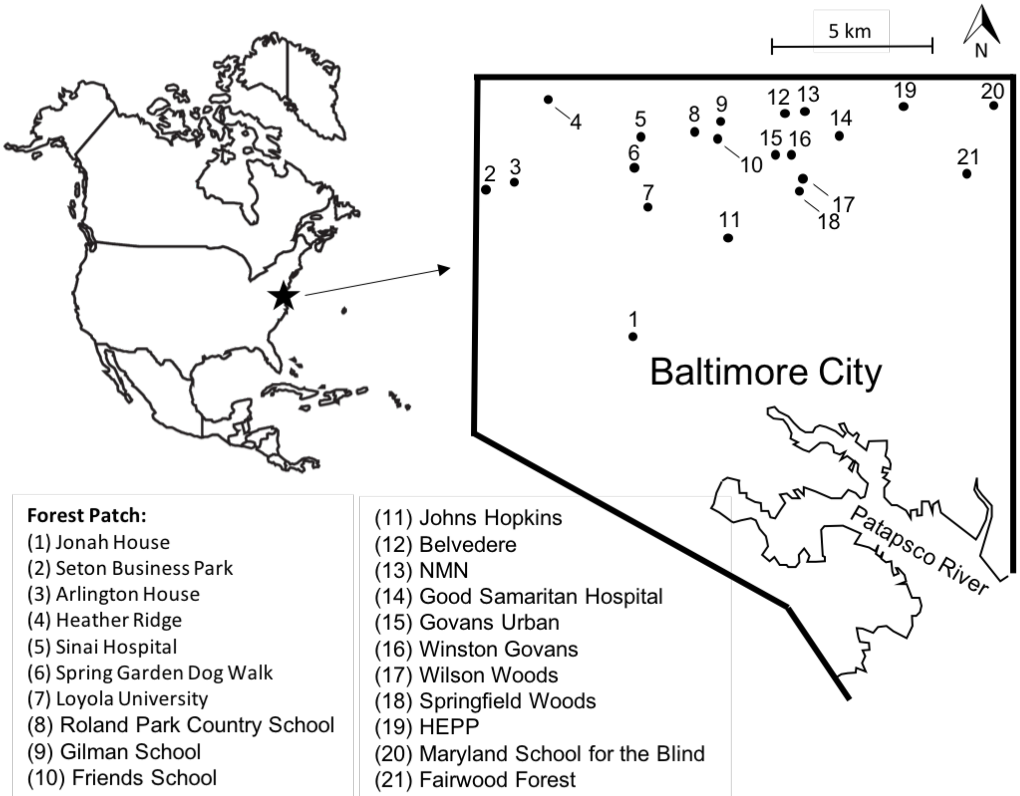


Figure 2-1: Locations of study sites, 21 forest patches within Baltimore City, Maryland, USA. Patch names are derived from the entity that owns them (e.g., Johns Hopkins University), or are a given name (e.g., Fairwood Forest).

We used tension infiltrometers (Mini-Disk Infiltrometer®, Decagon Devices, Pullman WA, USA) to take three measurements of K at each location. To ensure that measurements were performed on unsaturated soils, we avoided going out on days immediately following precipitation events. We applied a suction head of two cm to the infiltrometer to assess surface infiltration through meso- and micropores less than or equal to 1.45 mm in diameter. By preventing water from entering larger macropores, the effects of preferential flow paths are reduced and the infiltrometer captures infiltration capacity due to the matric potential and hydraulic forces present in the soil (Mini Disk Infiltrometer User Manual, Decagon Devices, Inc., Version: September 2, 2016). Because roots and soil fauna in forests create macropores and preferential flow paths, the measurement by the device served as an accurate but conservative estimate of the ability of soil in urban forest patches to infiltrate rainfall.

Loose litter and organic debris were brushed out of the way to ensure that the bottom of the infiltrometer made full contact with the surface soil. We then monitored and recorded the volume of water in the infiltrometer until at least 15 mL entered the soil. Manufacturer protocols were used to model the three measurements of K per location using the following equation:

$$k = \frac{C_1}{A}$$

Where C_1 is the slope of the cumulative infiltration curve versus the square root of time, and A is a van Genuchten parameter based on the suction rate of the infiltrometer and texture class of the soil (Mini Disk Infiltrometer User Manual, Decagon Devices, Inc., Version: September 2, 2016).

2.3 Soil samples and physical properties

In the winter and spring of 2018, we used BD samplers to take three 5 cm diameter x 5 cm deep surface cores at 62 locations corresponding to the same places where we assessed soil infiltration capacity. To assess BD immediately below each infiltrometer measurement, all cores were taken without removing the O soil horizon. Soil BD was calculated by dividing the weight of the oven dried soil (105 °C, 72 hours) by its volume. To capture the true BD of the soil mineral and decomposed organic material, large rocks and roots were not included in the analysis by removing them from the soil sample and accounting for their total weight and volume as measured through water displacement.

At the same locations, a soil auger was used to take three additional cores of the upper five cm of soil. Soils cores were homogenized and sieved to 2 mm to remove coarse fragments. Percent SOM was determined based upon loss on ignition of sieved soil (550 °C, 2 hours). The percent of coarse fragments in each location was calculated as the weight of the cleaned and dried coarse fragment material divided by the oven-dried weight of the entire soil sample. Soil texture of the sieved soil was determined by feel analysis (Thien 1979). In the field, prior to initiating each infiltration capacity test, we used a HydroSense Soil Water Measurement System (Campbell Scientific, Inc., Logan UT, USA) to measure soil moisture (volumetric soil water content) in the upper 20 cm of soil.

2.4 Soil capacity to infiltrate stormwater

We downloaded 38 years (1975–2013) of hourly precipitation amounts in the Baltimore region from the NOAA National Centers for Environmental Information website (https://www.ncdc. noaa.gov/cdo-web/datatools/findstation). Borrowing from methods in Herrmann et al. (2015), we calculated the percent of stormwater infiltrated by Baltimore forest patch soils based on the mean K measurements from each sampled location. In these calculations, we excluded small wet-up events with one mm or less rainfall accumulation (Herrmann et al. 2015). In addition, we compared the mean K values per soil type to rainfall rates that are generated by storms of different durations and recurrence intervals (1-, 2-, 10-, 50-, and 100-year storms) (NOAA National Weather Service Hydrometeorological Design Studies Center Precipitation Frequency

Data Server: https://hdsc.nws.noaa.gov/hdsc/pfds/pfds_map_cont.html?bkmrk=md). This gave us an approximation of the capacity of soil in urban forest patches to infiltrate stormwater generated by higher intensity storm events that occur less frequently.

2.5 Temporal changes in hydraulic conductivity

In July and August of 2018, we measured K rates about once every week in one location (39° 21' 59.09" N, 76° 32' 5.75" W) of the Maryland School for the Blind forest patch. This location was chosen for these measurements because it was also the location of another ongoing study that required weekly visits to monitor environmental sensors. This forest patch is characterized with having surface soils that are predominately sandy loam in texture, with an average BD of 0.95 g per cm^3 and an average SOM content of 11.89 percent. As part of a separate ecohydrologic monitoring study, we had installed 15 soil volumetric water content reflectometers (CS616, Campbell Scientific, Inc.) in this forest patch and deployed a weather station (HOBO U30 USB Weather Station, Onset Computer Corporation) outside of the forest patch to continuously monitor precipitation (0.2 mm Rainfall Smart Sensor, Onset Computer Corporation), temperature (Temperature and Relative Humidity Smart Sensor, Onset Computer Corporation), and additional weather parameters. This set-up allowed us to assess how changes to K rates over time may be influenced by weather parameters and soil moisture conditions.

2.6 Statistical Analyses

Due to the non-normality and heteroscedasticity of the data, we used Spearman's rank correlation to assess for monotonic relationships between explanatory and response environmental variables. We further explored linear associations using linear regression analyses of log-transformed data and accounted for a potential lack of independence among samples by checking for spatial autocorrelation using a semivariogram and Moran's I coefficient. Kruskal-Wallis rank sum tests were used to assess soil texture class effects on K values and Wilcoxon two-sample tests were used to identify which texture classes were significantly different from each other. We used the statistical package R (ver. 3.4.1, R Foundation for Statistical Computing, 2016) and RStudio (1.0.153 RStudio, Inc., 2009-2017) to perform our analyses.

3.0 Results

3.1 Soil physical properties

Mean soil physical property values for each forest patch are reported in Table 2-1. Soil BD averaged 0.88 $\pm$ 0.03 g per cm^3 ($\pm$ 1 SE) and ranged from 0.41 to 1.47 g per cm^3 in the upper 5 cm of soil. SOM was significantly related to BD (rho = -0.82, n= 62, $p < 0.0001$), with low BD values corresponding to high SOM content. SOM ranged from 3.83 to 29.44 percent and averaged 11.42 $\pm$ 0.57 percent. In addition, the mean percent of coarse fragments for all 62 sampled locations was 7.38 $\pm$ 1.3 percent, with a minimum of 0 and a maximum of 46.04 percent. Volumetric water content (VWC) values prior to each infiltration capacity test ranged from 4.08 to 43.53 percent and averaged a value of 18.26 $\pm$ 1.24 percent. Most locations were classified as clay loam (n = 39), followed by loam (n = 12), clay (n = 8), and sand (n = 3) in soil texture.

3.2 Soil infiltration and capacity to infiltrate stormwater

We assessed soil infiltration potential in urban forest patches by measuring the unsaturated hydraulic conductivity in 21 Baltimore forest patches. Across all 62 sampled locations, urban forest soils averaged a hydraulic conductivity rate of 0.61 ± 0.3 cm per hr with a minimum value of 0.06 cm per hr and a maximum value of 3.66 cm per hr. Infiltration capacity varied across forest patches (Table 2-1), and we detected no spatial autocorrelation of K values, suggesting that K rates are also dissimilar among locations within close proximity to each other (Moran's I. = 0.04, p = 0.2). Average K values for the 11 larger forest patches protected under easement did not significantly differ from the 10 smaller forest patches nested within neighborhoods (0.59 cm per hr and 0.62 cm per hr on average, respectively).

Considering the average K rate of 0.61 cm per hr, 89 percent of storms generated rainfall at this rate or lower based on hourly rainfall data from 1975 to 2013. When accounting for the total amount of rainfall that fell during this same time period, Baltimore forest patch soils could infiltrate, on average, ~68 percent of all rainfall that fell (Table 2-1).

3.3 Drivers of K

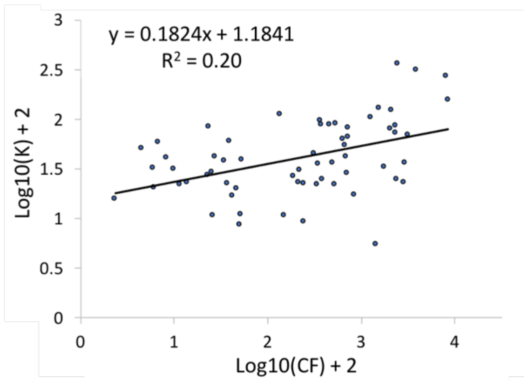
Spearman correlation analyses showed a positive, significant relationship between K and percent of coarse fragment in the soil (rho = 0.45, n= 62, $p < 0.001$). No significant correlations were found between K and soil BD (rho = -0.17, n = 62, p = 0.18), K and SOM (rho = 0.07, n = 62, p = 0.6), and K and soil VWC (rho = 0.19, n = 62, p = 0.13). Linear regression analyses of log-transformed data identified percent of coarse fragments as well as BD as significant ($p < 0.05$) predictors of K. However, the produced models had relatively weak R^2 values equal to 0.20 and 0.08 for the predictors coarse fragments and BD, respectively (Figures 2-2a and 2-2b). Spatial autocorrelation within the data was detected for soil BD and soil moisture (Moran's I. = 0.15, $p < 0.01$

and Moran's I = 0.26, p < 0.001, respectively), but not for percent of coarse fragments (Moran's I. = 0.05, p = 0.15) and SOM (Moran's I. = -0.02, p = 0.08).

Table 2-1: Soil physical properties, unsaturated hydraulic conductivity (K), and calculated stormwater infiltration capacity for 21 forest patches in Baltimore, Maryland, USA. Reported values are means ± the standard error. From cores of the upper 5 cm of soil (including surface fine organic matter), BD = bulk density, SOM = soil organic matter, and CF = coarse fragments. VWC = volumetric water content in upper 20 cm of soil. Forest patches identified by asterisks (*) are smaller forest patches nested within neighborhoods. All other forest patches are generally larger in size and protected under easement.

Forest Patch	Approx. size (ha)	BD (g/cm^3)	SOM (%)	CF (%)	VWC (%)	K (cm/hr)	Stormwater infiltrated (%)
Jonah House*	5	1.10 ± 0.08	8.52 ± 1.58	8.19 ± 3.59	18.4 ± 1.9	0.16 ± 0.04	40.5 ± 13.7
Seton Business Park	16	1.04 ± 0.09	9.64 ± 2.26	2.5 ± 2.21	26.9 ± 1.9	0.28 ± 0.06	58.6 ± 7.6
Arlington House	5	0.94 ± 0.02	10.91 ± 0.04	0.26 ± 0.10	30.4 ± 2.9	0.23 ± 0.05	48.7 ± 12.9
Heather Ridge	3	0.81 ± 0.15	11.99 ± 0.72	29.83 ± 27.77	29.3 ± 2.7	0.73 ± 0.27	75.4 ± 10.2
Sinai Hospital	15	1.06 ± 0.04	7.77 ± 0.78	17.10 ± 5.26	20.2 ± 4.4	0.54 ± 0.12	75.8 ± 8.0
Spring Garden Dog Walk*	2.5	0.82 ± 0.12	16.41 ± 6.57	0.07 ± 0.03	11.2 ± 0.6	0.24 ± 0.05	55.0 ± 7.3
Loyola University	22	0.81 ± 0.11	13.26 ± 1.68	24.10 ± 10.57	19.1 ± 2.4	1.51 ± 0.55	86.1 ± 8.5
Roland Park Country School	4	0.76 ± 0.08	12.51 ± 2.12	3.32 ± 2.00	8.5 ± 0.6	0.25 ± 0.03	58.6 ± 2.8
Gilman School	7	0.84 ± 0.14	13.24 ± 3.17	2.84 ± 1.84	15.4 ± 3.0	0.43 ± 0.11	72.3 ± 5.4
Friends School	6	0.88 ± 0.18	9.12 ± 1.31	35.42 ± 23.90	26.8 ± 2.3	1.53 ± 0.40	91.7 ± 3.7
Johns Hopkins University	28	1.20 ± 0.20	6.62 ± 2.78	20.91 ± 5.72	21.5 ± 0.9	0.69 ± 0.11	80.9 ± 6.9
Belvedere*	0.3	0.85 ± 0.03	11.96 ± 1.15	11.49 ± 4.13	28.3 ± 1.9	1.98 ± 0.35	88.4 ± 4.8
NMN*	0.5	0.96 ± 0.07	8.56 ± 1.39	9.09 ± 7.12	7.5 ± 1.3	0.73 ± 0.15	83.3 ± 5.0
Good Samaritan Hospital	6	1.06 ± 0.04	7.86 ± 1.39	1.38 ± 0.47	13.1 ± 1.7	0.14 ± 0.06	36.7 ± 10.4
Govans Urban*	0.4	0.84 ± 0.01	9.91 ± 1.31	1.70 ± 1.46	25.2 ± 1.6	0.67 ± 0.13	79.7 ± 8.4
Winston Govans*	1.5	0.75 ± 0.17	13.99 ± 3.30	9.94 ± 7.41	23.3 ± 2.5	1.63 ± 0.53	84.7 ± 9.7
Wilson Woods*	1	0.90 ± 0.09	9.55 ± 2.62	0.08 ± 0.02	21.5 ± 1.4	0.44 ± 0.10	71.4 ± 8.7
Springfield Woods*	2.5	0.80 ± 0.10	15.43 ± 1.71	3.69 ± 2.00	8.2 ± 0.7	0.55 ± 0.16	74.7 ± 10.0
HEPP*	3.5	0.62 ± 0.04	15.13 ± 1.56	17.76 ± 7.44	5.6 ± 0.8	0.64 ± 0.19	72.9 ± 10.5
Maryland School for the Blind	15	0.95 ± 0.05	11.89 ± 2.54	1.08 ± 0.55	11.8 ± 1.2	0.27 ± 0.07	55.7 ± 13.1
Fairwood Forest*	4	0.62 ± 0.07	15.69 ± 2.21	3.63 ± 0.85	14.9 ± 1.5	0.26 ± 0.06	51.9 ± 14.5

(a) (b)



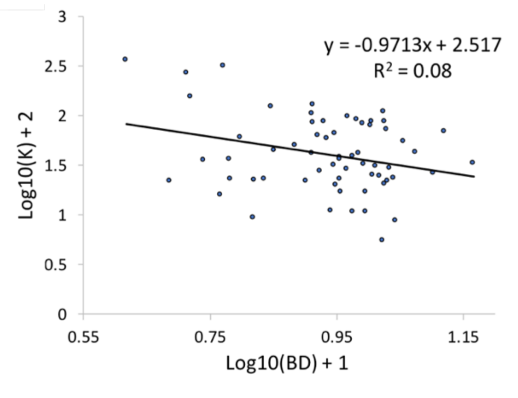


Figure 2-2: The relationship between soil infiltration capacity, measured as the unsaturated hydraulic conductivity (K) in cm per hour, and (a) the percent of coarse fragments (CF) and (b) soil bulk density (BD) in 21 forest patches in Baltimore, Maryland, USA.

Infiltration capacity was significantly higher in soils with high sand content compared to loam, clay loam, and clay soils (Kruskal-Wallis rank sum and Wilcox two-sample tests, $p < 0.05$). K in loam, clay loam, and clay soils did not significantly differ from each other. Soils classified as sand averaged a K rate of 3.19 cm per hr compared to 0.61 cm per hr in loam soils, 0.43 cm per hr in clay loam soils, and 0.47 cm per hr in clay soils (Figure 2-3). Mean infiltration capacity (K/unit time) of each soil type was lower than rainfall rates produced by larger, less frequent storm events of ≤ 1 hour durations and recurrence intervals of 1, 2, 10, 50, and 100 years, except for sandy soils in a 1 hour, 1 year storm (Table 2-2).

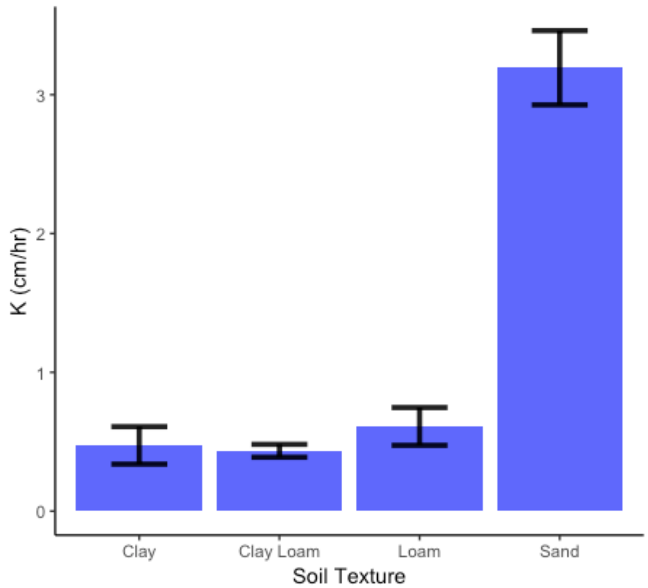


Figure 2-3: Average unsaturated hydraulic conductivity (K) rates for different soil texture classes in 21 urban forest patches in Baltimore, MD, USA. Error bars represent the standard error.

3.4 Temporal changes in K

Continuous monitoring of K in one forest patch revealed the dynamic nature of soil hydraulic conductivity (Table 2-3). Lower values of K rates were observed in mid- to late July of 2018, after a few weeks of minimal rainfall. In contrast, late July through mid-August saw high rates of K at the same location. This increase in K can be attributed to a continuous period of high rainfall starting in mid-July that led to a new rainfall record for the month of July in Baltimore. It can also be attributed to subsequent changes to the urban forest soil's water content, which more than doubled from July 5th to August 14th (Table 2-3).

Table 2-2: Precipitation (cm) in the Baltimore region generated by larger, infrequent storm events of 5-, 10-, 30-min and 1 hour durations and average recurrence intervals ranging from 1 to 100 years, and whether precipitation exceeds infiltration capacity (K, cm/unit time) for each urban forest patch soil texture type (C = clay, CL = clay loam, L = loam, S = sand). Y = yes; N = No. Table adapted from Ossola et al. (2015). Precipitation values from NOAA: https://hdsc.nws.noaa.gov/hdsc/pfds/pfds_map_cont.html?bkmrk=md.

	Duration	5 minute				10 minute				30 minute				1 hour			
	Soil Texture	C	CL	L	S	C	CL	L	S	C	CL	L	S	C	CL	L	S
Average recurrence interval																	
1 year	Precipitation	0.88	0.88	0.88	0.88	1.40	1.40	1.40	1.40	2.41	2.41	2.41	2.41	2.99	2.99	2.99	2.99
	Exceeds K?	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	**N**
2 years	Precipitation	1.05	1.05	1.05	1.05	1.68	1.68	1.68	1.68	2.92	2.92	2.92	2.92	3.66	3.66	3.66	3.66
	Exceeds K?	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y
10 years	Precipitation	1.40	1.40	1.40	1.40	2.23	2.23	2.23	2.23	4.09	4.09	4.09	4.09	5.33	5.33	5.33	5.33
	Exceeds K?	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y
50 years	Precipitation	1.72	1.72	1.72	1.72	2.72	2.72	2.72	2.72	5.21	5.21	5.21	5.21	7.04	7.04	7.04	7.04
	Exceeds K?	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y
100 years	Precipitation	1.84	1.84	1.84	1.84	2.92	2.92	2.92	2.92	5.66	5.66	5.66	5.66	7.80	7.80	7.80	7.80
	Exceeds K?	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y

Table 2-3: Temporal changes in unsaturated hydraulic conductivity (K) are associated with changes in soil volumetric water content and antecedent rainfall. Measurements of K and soil volumetric content were taken at one forest patch, while a rain gauge outside of the patch monitored rainfall.

Date	K (cm/hr)	Soil Volumetric Water Content (%)	Date of last rainfall event > 1mm	5-day total antecedent rainfall (mm)
07/05/2018	0.073	16.3*	6/24	0
07/13/2018	0.055	14.2*	6/24	0
07/19/2018	0.066	17.3	7/17	22.2
07/30/2018	1.226	33.4	7/29	20.4
08/07/2018	1.327	31.9	8/06	11.2
08/14/2018	2.162	33.7	8/13	44.4

*Due to sensor failure on July 5th and 13th, these data were taken from soil volumetric water content sensors at another location within the same forest patch.

4.0 Discussion

4.1 Urban forest patch soil's capacity to infiltrate stormwater

Based on the observed average K rates, unsaturated surface soils in urban forest patches are capable of fully infiltrating low-intensity (< 0.25 cm per hr) to moderate-intensity (0.25-0.61 cm per hr) rainfall events that have been historically common (i.e. approximately 89 percent of hourly rainfall amounts greater than 1 mm) in the Baltimore region. In contrast, high-intensity storm events that occur less frequently, with 1, 2, 10, 50, and 100 year recurrence intervals in the Baltimore region, generate rainfall rates that exceed the average K (Table 2-2). Therefore, urban forest patch soils may infiltrate some but not all of the stormwater during those storm events. When accounting for the total hourly precipitation amounts generated from 1975-2013, urban forest patch soils could infiltrate 68 percent of rainfall at the measured K rates.

The observed K rates in Baltimore forest patches are similar to K rates in urban forests and other green spaces as reported by studies that also used the Decagon Mini Disk infiltrometer to characterize infiltration. For example, the range of soil K rates in Baltimore forest patches is comparable to the range found in high-complexity remnant forests in Melbourne, Australia (Ossola et al., 2015) and in desert parks and residential parcels within the Phoenix, Arizona metropolitan area (Shuster, Dadio, Burkman, Earl, & Hall, 2014). Back-transformed means of K in vacant lots in two cities in the midwestern United States indicate that infiltration capacity in Baltimore forest patches are similar to infiltration capacity in Detroit, Michigan vacant lots (0.6 cm per hr) but lower than infiltration capacity in Cleveland, Ohio vacant lots (1.2-1.4 cm per hr) (Herrmann et al., 2017). Further, in Cincinnati, Ohio, the mean soil K of 0.6 ± 0.1 cm per hr in a turf area with clay loam soils (Shuster, Darner, Shifman, & Herrmann, 2017) is almost identical to the mean K from soils in Baltimore's forest patches. Although we expect soils in urban forests to have relatively high infiltration rates compared to non-forested areas (Bartens et al., 2008; Kuehler et al., 2017), our measurements of K are conducted under a slight tension that eliminates macropore flow. Macropores are abundant in forest soils and conduct the majority of water flux under saturated soil conditions (Watson & Luxmoore, 1986). The difference between unsaturated and saturated hydraulic conductivity is evident in Ossola et al. (2015), who saw higher soil K rates in saturated conditions as opposed to in unsaturated conditions for remnant forests and urban parks with high-complexity habitats relative to urban parks with low-complexity habitats.

Results from studies of surface K measured under slight tension in bioretention cells indicate that designed green infrastructure elements are capable of infiltrating rainfall at a K rate at least twice as fast as what was observed on average in Baltimore forest patches. Shuster et al. (2017) studied a two-tiered infiltrative rain garden system during the warm-season periods from 2012-2015 and report near-saturated hydraulic conductivity rates of 2.2 +/- 0.4 and 2.0 +/- 0.5 cm per h for the mulch surface soils of the two rain gardens. Below the mulch layer, infiltration capacity in the loamy sand soil of one garden varied by year and peaked at 12.9 +/- 1.7 cm per hr. The mean K rate in the sandy loam soil of the second rain garden was more consistent throughout the years and averaged approximately 2 cm per hr. In another rain garden in Cleveland, Ohio, hydraulic conductivity averaged a rate of 1.2 +/- 0.82 cm per hr for the A horizon and approximately 8 cm per hr for the engineered biosoil (Stewart, Lee, Shuster, & Darner, 2017). Three dual-purpose park-stormwater retention basins in Phoenix, Arizona had relatively low K rates (0.4-1.1 cm per hr). The low infiltration rates were attributed to year-long trampling of the basin surface which also served as a park (Shuster et al., 2014). These data suggest that designed SWGI such as bioretention cells on average perform better than urban forest soils. However, the maximum rate observed in Baltimore forest patches was 3.66 cm per hour, therefore there may be some overlap in infiltration capacity depending the urban forest patch soil properties and antecedent soil moisture of the location.

4.2 Soil physical properties as drivers of infiltration

Out of all of the soil physical properties assessed in this study, the percent of coarse fragments in the top 5 cm of soil was the most important in determining K rates. Soils with relatively high percentages of coarse fragments displayed relatively higher K rates, and this relationship was supported by significant results from Spearman correlation and linear regression analyses. Soil BD was also determined as a key driver in linear regression analyses. However, a weak R^2 value for the relationship between BD and K (Figure 2-2b) suggests that other soil properties are more important in determining K. Low BD soils did display high K rates but this was not consistent across all locations. For instance, the Fairwood Forest patch exhibited low BD values averaging 0.62 g per cm^3, but a relatively low mean K rate of 0.26 cm per hr. Although other studies have shown that the saturated hydraulic conductivity decreases with soil compaction due to a reduction in macropore space (Gregory et al., 2006; Yang & Zhang, 2011), this was not the case in Ossola et al. (2015), who found no significant relationship between BD and the saturated hydraulic conductivity in parks and forests in Australia. Therefore, although soil BD likely affects K, it may not be well-suited to serve as an ecosystem service indicator of infiltration capacity (J. Ahern et al., 2014; Herrmann et al., 2017) due to inconsistent observations of relationships. In our study, SOM, which was strongly related to BD, did not significantly correlate with K either and may also not work as an indicator of infiltration capacity.

Along with the percent of coarse fragments in a soil, texture seems key in determining K. Urban forest patch soils with high sand content demonstrated higher infiltration potential compared to other soil texture classes (Figure 2-3). The effects of texture on K are also noted in Herrmann et al. (2017). Their findings corroborated what is known theoretically about the relationship between texture and infiltration, i.e. higher clay content and lower sand content in soils lead to a decrease in hydraulic conductivity and infiltration. Thus, it can be said that coarser-textured urban forest soils that have high sand content and/or abundant coarse fragments ($\geq$ 2 mm) demonstrate a higher capacity to facilitate infiltration of rainfall and lessen the amount of runoff generated from the land. These two soil physical characteristics could be used as indicators to identify locations of high infiltration potential.

4.3 Temporal changes in K and importance of soil moisture

Studies have depicted a non-linear, positive relationship between soil moisture and K until K reaches steady-state infiltration as expressed by the saturated hydraulic conductivity (Chaudhari et al., 2015; Homolák, Capuliak, Pichler, & Lichner, 2009). Based on the measurements from all 21 forest patches, we did not find a significant relationship between the average soil water content and average K per location. However, our results still point to the importance of soil moisture after monitoring K at one forest patch location over a span of several weeks that saw substantial changes to the soil's water content (Table 2-3). As soil moisture content increased at this site, so did the hydraulic conductivity. A sharp increase in K with increasing soil moisture has also been reported in Gonzalez-Sosa et al. (2010) and Gadi et al. (2017). The former study attributed the increase to the macroporosity effect as soils approach saturation. Indeed, the hydraulic conductivity is a function of the soil matric potential, which relates to the moisture content in the soil. As soil moisture decreases, the matric suction increases and hydraulic conductivity declines due to the drying of the largest macropores that empty out quickly and become filled with air, consequently obstructing water flow (Gallage, Kodikara, & Uchimura, 2013; McCartney, Villar, &

Zornberg, 2007). The rate that water can move through the soil is then determined by flow through the smaller pores that are not filled with air or by slow movement of water along the walls of the larger pores. The path that water can take becomes more tortuous and difficult with a decline in soil saturation (Gallage et al., 2013; McCartney et al., 2007). The opposite effect is the case as soil moisture increases and larger soil pores become filled with water, increasing the conductivity of water.

Our monitoring of K over time in one forest patch therefore shows that soil hydrological characteristics such as K can vary over time and are a function of dynamic soil properties such as soil moisture. To assess infiltration capacity, we calculated the percent of stormwater infiltrated per location based on rainfall data from 1975-2013 and the measured K rates. By doing so, we assumed that the K measurements taken across the 21 forest patches were constant for all years. We already knew that the measurements of K in Baltimore forest patches serve as conservative estimates of infiltration capacity due to tension infiltrometer technique preventing flow in macropores greater than 1.45 mm in diameter. Still, the temporal dynamics observed indicate that the macroporosity effect can also be registered by the tension infiltrometer for meso- and macropores less than 1.45 mm in diameter. Thus, we speculate that some of the measurements taken in relatively dry soil conditions (approximately ≤ 18 % VWC) during the summer of 2017 may have registered flow only through the smallest pores in the soil and therefore represent an even more conservative estimate of infiltration. Although our calculations made some assumptions about constant K values, urban forest patches are likely capable of infiltrating more stormwater than our estimate of 68% of rainfall in Baltimore.

4.4 Implications for urban stormwater management

Many urban ecosystem service studies to date have relied on coarse-scale mapping or modeling techniques to evaluate ecohydrological processes in urban areas (Gonzales-Sosa et al., 2017; Ossola et al., 2015; Revelli & Porporato, 2018; Rova, Pranovi, & Müller, 2015; Tratalos, Fuller, Warren, Davies, & Gaston, 2007). In the Chesapeake Bay region, decision makers who are interested in crediting the stormwater retention benefits of urban tree canopy have also had to rely on modeling techniques due to a lack of performance data (Law & Hanson, 2016). Measuring steady-state infiltration rates in the field can be difficult and time-consuming to do. However, empirical measurements of surface soil K are relatively quick and simple and give an indication of infiltration capacity and how it varies across different locations. There are several assumptions that have to be made because the rate that water is transported into soil during rainfall is not governed by K alone. Instead, soil infiltration depends on many things—including the soil pressure potential and gravitational potential, as well as other site-specific conditions such as degree of slope or restrictive layers below the surface that control the rate of infiltration into deeper layers. Yet, quantifying K in urban forest patches offers novel data on infiltration capacity and contributes to the broader knowledge of the role of urban forests in reducing stormwater runoff.

Our results conservatively estimate that urban forest patch soils reduce stormwater runoff by infiltrating approximately 68% of rainfall based on historical precipitation data. Climate change models predict that the Chesapeake Bay region will see an increase in annual precipitation as well as number of dry days, with episodic, high-intensity rain storm events becoming more common (Najjar et al., 2010). Urban forest patch soils with high sand content demonstrate higher capacity to infiltrate stormwater given the relatively high K rates observed (Figure 2-3), and therefore may perform better in future climate conditions. For instance, based on the average K rate of 3.19 cm per hr, soils classified as sand are likely capable of infiltrating a one-hour, one year storm that produces approximately 3 cm of rainfall in the Baltimore region (Table 2-2). However, it is important to note that soils classified as sand are relatively rare across Baltimore forest patches. Thus, although our results suggest urban forest soils are capable of fully infiltrating 89% of historical rainfall rates, their potential to mitigate stormwater runoff in the future may be reduced due to larger rain storm events that will become more frequent with climate change.

With respect to stormwater infiltration, designed SWGI practices may offer a better solution for climate change adaptation in urban watersheds (Brink et al., 2016; Escobedo et al., 2018). We saw that mean K was relatively lower in urban forest patches compared to in rain gardens (i.e., bioretention cells) as reported by studies (Shuster et al, 2017; Steward et al., 2017). Other studies suggest that SWGI practices may serve as effective solutions for climate adaptation in urban watersheds (Pyke et al., 2011; Giese et al., *in review*). Rain gardens and SWGI practices that are infiltration-focused are typically designed to be able to infiltrate a certain amount of rainfall within a defined drainage area (Askarizadeh et al., 2015; Shuster et al., 2017). Design standards for SWGI practices vary by state or locality and may include criteria for infiltration, recommended maximum size of drainage area, and texture requirements of planting soil. In Maryland, for instance, SWGI for new development projects must be able to handle a 1-year, 24-hour storm event (MDE, 2009). A challenge of using urban forests as nature-based solutions for stormwater is that they are not designed or managed like SWGI to store and infiltrate a certain amount of stormwater. Instead, our research shows that urban forest patch soil infiltration capacity is determined by soil physical properties such as texture and abundance of coarse fragments. Moreover, our results indicate that infiltration capacity and some soil physical properties (e.g., percent of coarse fragments) are not spatially autocorrelated. This reinforces the importance of local knowledge and site specific analyses of soil conditions for urban management and ecosystem service provision despite recent studies that report common patterns and convergences in urban soils across the USA (Herrmann et al. 2018) and the globe (Pouyat et al., 2015).

Our study suggests that urban forest patches impact the hydrology of cities via soil infiltration, and therefore they are an important element of the city's green infrastructure portfolio. Urban forest patches constitute 34% of the tree canopy in Baltimore (Avins, 2013), which in turn constitutes approximately 27% of the landscape in the city (Chuang et al., 2017). Smaller forest patches may be just as important as larger forest patches; in our study, the smallest patches nested in neighborhoods displayed K rates that were within the same range of K in larger forest patches protected under easement (Table 2-1). In addition to soil infiltration, it is important to consider other urban forest ecohydrological functions that contribute to stormwater runoff mitigation. For instance, some rainfall is lost due to interception by trees, understory plants, and leaf litter (Helvey & Patric, 1965; Inkiläinen, McHale, Blank, James, & Nikinmaa, 2013; Nytch, Meléndez-Ackerman, Pérez, & Ortiz-Zayas, 2018; Ossola et al., 2015). Transpiration by trees is also an important ecohydrological function in urban forests and likely a key contributor to stormwater abatement (L. Chen et al., 2011; Jacobs et al., 2014; Kuehler et al., 2017; Hua Wang, Wang, et al., 2012). Transpiration is an undervalued function in current stormwater green infrastructure practices that are largely infiltration-focused (Berland et al., 2017; Bhaskar, Hogan, & Arch, 2016). The conservation and expansion of forest patches in cities can therefore make potentially large contributions to runoff mitigation by promoting ecohydrological functions that infiltrate, intercept, and transpire rainfall.

5.0 Conclusion

Urban forest infiltration capacity, as evaluated across 21 forest patches in Baltimore, is temporally dynamic on a weekly scale and dependent on soil moisture conditions in addition to other soil properties that are more stable over time. In particular, soil texture and the percent of coarse fragment material in soils drive K and can serve as ecosystem service indicators to identify locations in forest patches that exhibit higher infiltration capacity. Overall, our data show that urban forest soils have the potential to infiltrate most rain storm events, thus impacting urban hydrology. However, they may be less capable of infiltrating more intense storm events that will become more common in the future according to projected climate conditions (Najjar et al., 2010). While the use of designed SWGI practices (e.g. bioretention cells) may be more effective at infiltrating stormwater, the infiltration capacities we observed in forest patches suggest that urban forests are important stormwater control measures and can reduce runoff and flooding in urban areas.

Chapter 3 Tree size and classification schemes explain urban tree water use

Abstract

Urban tree transpiration provides valuable ecosystem services through the cooling of air and reduction of stormwater runoff in developed areas. However, findings from tree transpiration studies in non-urban areas may not be directly transferable to the built environment and urban studies to date emphasize that transpiration amounts are highly species-specific and dependent on environmental conditions such as temperature and relative humidity, therefore complicating our understanding of urban tree transpiration amounts. In this meta-analysis, we gathered urban tree transpiration rates from the peer-reviewed literature to better understand urban tree water use and explore whether tree characteristics, including tree size and tree type, can be used to explain species' differences in urban tree water use and make generalizations about urban tree transpiration. We found that DBH and canopy area are strong predictors of urban tree water use in the growing season. Results affirm that transpiration by deciduous urban trees is significantly higher than transpiration by evergreen urban trees during the growing season; the opposite is true on an annual basis. In addition, hardwood species in urban areas exhibit higher growing season transpiration rates relative to softwood species. This study further assessed how urbanization (i.e., increased impervious surfaces) and different management contexts affect tree transpiration. We found that, for trees that are not water-stressed, transpiration rates by street trees are higher than transpiration rates by trees over pervious surfaces and trees in urban forest patches. These findings help to broaden the understanding of urban tree water use and may be used to scale up and facilitate the

integration of transpiration-based ecosystem services into urban planning and management.

1.0 Introduction

Urban development drastically changes the environment by replacing natural vegetative cover with built impervious surfaces. This transformation presents a number of social-ecological challenges in urban areas, including excess stormwater runoff after rain events (Booth & Jackson, 1997; Lee & Heaney, 2003; Shuster et al., 2005) and elevated temperatures from the urban heat island effect (Arnfield, 2003; Brazel, Selover, Vose, & Heisler, 2000; Igun & Williams, 2018). Integrating trees and other green infrastructure elements into urban landscapes can offset the negative effects of the urbanized built environment. In particular, trees are of key interest to urban planning and management due to their ability to provide ecosystem services that lessen stormwater runoff and cool the ambient air and surface temperatures. Trees are capable of transpiring large amounts of water and, as a result, (1) reduce runoff from the land by returning water to the atmosphere and increasing infiltration capacity through the emptying of soil pore spaces (Gotsch, Dragulji, & Williams, 2018; Law & Hanson, 2016; Riikonen, Järvi, & Nikinmaa, 2016; Scharenbroch, Morgenroth, & Maule, 2015), and (2) cool the ambient air temperature by releasing water vapor and converting sensible heat to latent heat (Ballinas & Barradas, 2016a; Green, 1993; Rahman, Smith, Stringer, & Ennos, 2011).

To date, however, there is limited understanding of how much water urban trees transpire (Berland et al., 2017; Kuehler et al., 2017). Although there is a rich body of literature on tree ecophysiology and transpiration in non-urban areas, results from such studies may not be directly transferable to trees in urban and suburban areas due to the different and varying environmental conditions that urban trees and their underlying soils experience (Calfapietra, Peñuelas, & Niinemets, 2015; Pickett & Cadenasso, 2009; Whitlow & Bassuk, 1988). Relative to trees in naturally forested areas, trees in urbanized areas may experience altered micro-climates that affect urban tree transpiration, including higher temperatures, lower relative humidity, increased evaporative demand, and increased exposure to wind and photosynthetic active radiation due to isolation from other trees (Asawa et al., 2017; Yilmaz, Toy, Irmak, & Yilmaz, 2007; Zipper, Schatz, Kucharik, & Loheide, 2017). Restricted root space, reduced soil water availability, and soil compaction may lead to a reduction in transpiration in urban areas due to drought stress and strong stomatal regulation of water loss (Komatsu et al., 2007; Pataki et al., 2011; Riikonen et al., 2016; Whitlow, Bassuk, & Reichert, 1992). The integration of transpiration-based ecosystem services into urban planning and management is challenging because transpiration amounts vary substantially depending on the tree species (Giraldo, Jackson, & Van-horne, 2015; Pataki, McCarthy, Litvak, & Pincetl, 2011; Peters, McFadden, & Montgomery, 2010). Moreover, individuals of the same species may vary in their transpiration amounts depending on seasonality and environmental conditions such as soil moisture and vapor pressure deficit (VPD) in air (Ballinas & Barradas, 2016b; Cregg & Dix, 2001; Riikonen et al., 2016; H Wang et al., 2011). Land managers and urban residents further contribute to a spatially heterogeneous urban forest by planting cultivars and non-native species and through various management practices, such as application of fertilizer, irrigation, and mulching (McCarthy & Pataki, 2010). Considering all of these effects, the result is a unique set of urban forest contextual and management typologies as well as micro-climates that lead a range of ecohydrological behaviors (Chen et al., 2011).

The use of trait-based classifications is gaining significant traction in ecological research and has been successfully applied to link species' traits to environmental response (Lavorel & Garnier, 2002; Lavorel, McIntyre, Landsberg, & Forbes, 1997), adaptation strategies (Tapolczai, Bouchez, Stenger-Kovács, Padisák, & Rimet, 2016), performance (Pataki, McCarthy, Gillespie, Jenerette, & Pincetl, 2013; Poorter & Bongers, 2006; Pywell et al., 2003), and urban ecosystem services (Pataki et al., 2013). Trait-based classifications may also offer a method to make sense of urban tree transpiration patterns despite the variation created by different species, management situations, and environmental conditions. Peters et al. (2010) propose several tree classification schemes to explain differences in species' water use in urban areas, including classifications that separate species by functional group (evergreen and deciduous species), wood anatomy (diffuse-porous, ring-porous, and conifers), and other tree characteristics. In addition to categorizing transpiration differences by tree type, the effects of urbanization may be examined by partitioning transpiration differences by management context, such as by comparing transpiration by street trees to trees in a park setting and to trees in a forest patch. Results from such categorizations can be used by managers scale up urban tree water use to stand and city levels and inform management decisions regarding transpiration-based ecosystem services by urban tree canopies.

In summary, evapotranspiration plays an important role in water cycling in urban areas, yet there is limited understanding of how much water is lost from urban tree transpiration due to varying rates and patterns that are reported from relatively few studies (Kuehler et al., 2017; Berland et al., 2017). By classifying urban species into trait-based or management groups, it is possible that generalizations can be made about urban tree transpiration amounts to better understand the role of trees in urban hydrology and mitigation of stormwater runoff and urban heat islands. Other than Peters et al. (2010), no studies to our knowledge have specifically set out to explain differences in urban tree transpiration by tree classifications. Also, Peters et al. (2010) were only able to analyze the trees in their study and evaluate differences by evergreen and deciduous species. Thus, the goal in this study was to use a meta-analysis to gather urban tree transpiration values form the peer-reviewed literature and explore the effects of tree characteristics, including tree size and tree type, in addition to the effects of urbanization and management context, on urban tree transpiration rate. Our research objectives were to:

1. Better understand how much water urban trees transpire.
2. Assess if tree size and tree type (functional group and wood structure) can explain water use by urban trees.
3. Assess if different management contexts (tree in forest patch, tree over pervious surface, tree over impervious surface) affect transpiration rates.

2.0 Methods

2.1 Literature search

We searched the peer-reviewed literature through July 2018 to find studies presenting data on urban tree transpiration rates. We used Google Scholar (scholar.google.com) and Web of Science (thomasonreuters.com) databases with the following search terms: “urban tree” AND “transpiration”. We included all studies that empirically quantified urban tree transpiration, including studies that took direct measurements in an urban environment, as well studies in greenhouse, experimental, and orchard settings as long as the research questions related to tree water use in urban or non-natural, managed areas. We located 55 publications examining approximately 104 species total.

Five of the 55 publications used the same data from a previous publication, therefore we found 50 studies representing unique efforts to quantify transpiration. Out of these 50 unique efforts, ultimately 15 studies reported mean transpiration values or other mean parameter values that we could use to calculate mean transpiration (Table 3-1). Some studies reported ranges or minimum/maximum transpiration values that we could not use. The remaining studies reported equations of transpiration as a function of VPD and environmental parameters or graphically displayed temporal (e.g., seasonal) trends of sap flux, from which we could not retrieve accurate estimates of mean transpiration. Most of the 15 studies reported mean transpiration rates for each studied species, with the exception of Peters et al. (2010), who grouped mean transpiration rates by genus. Thus, the data we used and present in this study represent taxonomic (species- or genus-level) mean values of tree transpiration. From the 15 studies, we acquired mean daily transpiration by 42 taxa (species or genera) measured in the growing season or summer months. Two of the studies also evaluated transpiration over the course of at least one year. From these two studies, we acquired mean daily transpiration values by 12 taxa representing transpiration on an annual basis.

Each study's location, tree species, method for estimating tree water use, date and duration of measurements, and key findings were recorded. If reported, we also recorded tree water use and other tree parameters (e.g., height, leaf area, projected canopy area, diameter at breast height). We found that studies varied in the units they chose to report transpiration rate, using: mmol m^{-2} s^{-1}, Mg of water, g cm^{-2} d^{-1}, mm d^{-1} per unit leaf area, mm d^{-1} per canopy area, g d^{-1}, kg d^{-1}, or L d^{-1}. Where possible, we converted all daily transpiration values to mean liters per day (L d^{-1}) or mm per day per unit area (mm d^{-1} m^2) to describe the range of transpiration rates across the studies. In some cases, we were able to use some of the provided data to calculate values that were not directly reported. For example, when daily tree water use per canopy area was reported along with canopy area, we were able to use those two values to calculate tree water use in liters per day.

We report mean values of daily tree transpiration in units of L d^{-1} as well as standardized mean daily transpiration rates based on either DBH or canopy/leaf area (in units of L d^{-1} cm^{-1} and mm d^{-1} m^{-2}, respectively). Further, we report the mean daily sum of sap flux density in the outer 2 cm of sapwood from studies that reported that value in units of g cm^{-2} d^{-1}. While we recognize that deriving the effect size of means in meta-analyses requires the pooling of variances and number of individuals measured (Borenstein et al., 2009), we were not able to do that. Many of the mean daily transpiration values we report were calculated based on two other reported mean values (e.g., we calculated mean tree water use per DBH with the mean tree water use value and mean DBH value), which increases the complexity of calculating the variance of that calculated mean value. Calculating the variance of that calculated mean value would require the variance of each of the two mean values plus their covariance, which we would not be able to compute without the raw data used to calculate each of the two mean values.

Table 3-1: Fifteen Studies included in this meta-analysis of urban tree transpiration. N_{yr} = number of years; N_{spp} = number of species studied. *indicates the seven studies included in the tree classification analyses.

Climate	Country	City	Method	Duration	N_{yr}	N_{spp}	Reference(s)
Humid Subtropical	Japan	Miyoshi	Lysimeter	2+ months/yr	3	1	Asawa et al. (2017)
Subtropical Highland	Mexico	Mexico City	Sap flow	15 days total	1	4	Ballinas & Barradas (2016a, 2016b)
Marine West Coast	Czech Republic	Brno	Sap flow	1 month	1	1	Cermák et al. (2000)
Humid Continental	China	Dalian City	Sap flow	5 months	1	4	*Chen et al. (2012)
Humid Subtropical	United States	Atlanta	Sap flow	12 weeks	1	2	*Giraldo et al. (2015)
Marine West Coast	New Zealand	Palmerston North	Sap flow	9 days	1	1	Green (1993)
Humid Subtropical	Japan	Saitama	Gravimetrically	3 days	1	1	Hagishima et al. (2007)
Mediterranean	United States	Seattle	Stomatal conductance	1 day	1	1	Kjelgren & Clark (1992)
Humid Subtropical & Humid Continental**	United States (2 locations)	Carbondale & Logan	Gravimetrically	4 days total	1	3	Kjelgren & Montague (1998)
Mediterranean**	United States	Los Angeles	Sap flow	9 months	1	1	*Litvak et al. (2011)
Humid Continental**	United States	Logan	Lysimeter	20 days	3	5	Montague et al. (2004)
Mediterranean**	United States	Los Angeles	Sap flow	6 months	2	15	*Pataki et al. (2011)
Humid Continental	United States	Minneapolis–Saint Paul	Sap flow	Year-long	2	12	*Peters et al. (2010)
Humid Continental	Finland	Helsinki	Sap flow	2+ months	4	2	*Riikonen et al. (2016)
Humid Continental	China	Beijing	Sap flow	Year-long	1	6	*Wang et al. (2011, 2012a)

**Authors described the location of these studies as arid or semi-arid.

2.2 Urban tree water use as explained by tree size and tree type

Using the data acquired from the 15 studies, we plotted mean daily transpiration values against mean tree size. We assessed both DBH and canopy area, if reported, as metrics of tree size. For this analysis, we excluded trees that were not irrigated or showed signs of water-stress, as was done Pataki et al. (2011), to analyze the effects of tree size (i.e., DBH or canopy area) on tree water use.

To analyze whether urban tree water use can be explained by differences in tree type, we used a subset of the 15 studies. For this analysis, we decided to only include data from studies that continuously measured transpiration for at least three months within the summer or growing season. We also included data from studies that measured transpiration continuously for one or more years as part of a year-long transpiration analysis. We only used data from these longer studies because transpiration is highly influenced by environmental conditions such as VPD and temperature (Cregg & Dix, 2001; Riikonen et al., 2016; Wang et al., 2011). Although we recognize that studies measuring transpiration intermittently or continuously for a shorter duration are still informative, they may not capture the dynamics of transpiration over time as caused by seasonal changes (i.e. peak transpiration conditions around June in the Northern Hemisphere) and rather represent transpiration rates driven by conditions that were unique during the time of the measurement. In addition, this meta-analysis is not focused on evaluating effects of environmental conditions on urban tree transpiration; rather, it is focused on broadening the understanding of urban tree transpiration rates on seasonal and annual terms and examine whether tree and management drivers affect water use. Therefore, the criteria we used excluded studies that used the gas exchange method, for instance, as well as sap flow studies that measured transpiration for a month or less (e.g., in Ballinas and Barradas 2016a and Cermák et al., 2000). All of the studies that met our condition quantified transpiration using the sap flow method that allows for continuous measurements. We found one study (Asawa et al., 2017) that continuously monitored water loss during the growing season using the lysimeter method. However, in their calculations of mean transpiration, the authors excluded days with precipitation and therefore we did not include their data in our meta-analysis. In total, there were seven studies examining 30 taxa that met this criteria. All studies examined at least two individuals of each taxa.

Since urban tree transpiration rates will also vary with the climate of the study location (Kjelgren & Montague, 1998), the second criteria we used was to only include data from studies done in locations with humid climates (Table 3-1). However, because several studies have continuously monitored tree water use in the semi-arid climate of Los Angeles, we also included values from trees in those studies as long as they were irrigated, and therefore, likely not water-stressed. We checked that including this data would be valid by confirming that the mean and range of transpiration values standardized by tree size were similar for the species measured in Los Angeles compared to the mean and range of the same values for the species in measured in the more humid climates of the other studies. This condition excluded several individuals and one species representing non-irrigated trees in Los Angeles (Pataki et al., 2011). Considering this second criteria, in total we used seven studies examining 29 taxa to analyze whether urban tree type and wood structure can explain differences in transpiration.

To complete linear mixed effect model analyses, we used transpiration values standardized by DBH. We chose to standardize by DBH because fewer than half of the studies that met our conditions reported canopy area values. Further, not all studies reported sap flux density (g cm^{-2} d^{-1}), which could tell us a lot about how different tree classification types differ in rates of water use. All seven studies, however, reported mean DBH values. We analyzed whether transpiration rate in liters per day per DBH in cm differed between functional group—i.e. evergreen and deciduous species or genera—during the growing season as well as on an annual basis. In addition, we assessed whether urban tree water use in the growing season could be explained by differences in wood structure by comparing transpiration rates by conifers, diffuse-porous, ring-porous, and semi-ring species or genera. There were not enough data points representing the full range of wood structure types to analyze transpiration differences on an annual basis.

2.3 Urban tree water use as explained by management context

We further analyzed whether urban tree water use standardized by DBH can be explained by different management contexts. Specifically, we assessed whether transpirations rates vary for urban trees in more forest-like conditions (i.e., forest patches), compared to trees over managed pervious surfaces such as turf, as well as to trees over impervious surfaces. For this analysis, we used the same seven studies and 29 taxa that we included in the tree classification analysis, from long-term (three or more month-long) and continuous studies of tree transpiration in humid climates or irrigated conditions. We identified the management context or typology based on what was described in each study. For instance, if the tree was described as being in a mixed species stand with a well-developed understory, then we placed that tree in the "forest patch" category. If the tree was located in park-like setting, over mowed turf or another managed pervious surface then it was put in the "over pervious surface" category, and if it was located along a street then it was placed in the "over impervious surface" category. All 12 taxa from the two studies that reported mean daily transpiration rates based on an annual basis were in the "over pervious surface" category, therefore we could not perform this analysis on those annual-long transpiration values.

2.4 Statistical analyses

To address skewness, heteroscedasticity, and non-normality in the data, we used Spearman's rank correlation to assess for monotonic relationships between urban tree water use and tree size. We also performed linear regressions to create predictive models. Due to the relatively low number of n values (n = 22) in the plot of urban tree water use against canopy area, we log-transformed that data to perform the linear regression. The plot of urban tree water use against DBH had more values (n = 40), therefore we performed linear regression on the raw data but we also report the log-transformed linear regression for comparison. We checked that the models met the assumptions of linear regression analyses, including linearity, normality of predictors, homoscedastic residuals, and normality of residuals.

We used R (ver. 3.4.1, R Foundation for Statistical Computing, 2016), RStudio (1.0.153 RStudio, Inc., 2009-2017), and the package *lme4* to create linear mixed effects models and assess whether the effects of tree type, wood structure, and management context significantly affect transpiration standardized by DBH. In the models, these effects were treated as fixed factors and for random effects we borrowed from methods in Peters et al. (2010) and included intercepts for the study location and genus of the tree. For all models, we visually inspected the residuals for normality and homoscedasticity. Likelihood ratio tests determined whether the full models with the fixed effect in question were significantly ($p < 0.05$) different than reduced models without the effect.

3.0 Results

3.1 Description of urban tree transpiration studies

Out of the 55 publications that quantified urban tree transpiration, 28 concerned the effects of environmental factors (e.g., micro-climate), site type, or tree characteristics (e.g., species) on transpiration. Ten of the publications were broadly interested in understanding urban tree water use, such as using this information to develop watering criteria for newly-planted saplings. Four focused on new methodologies to test or model urban tree transpiration, and the remainder of the publications (13) were interested in quantifying ecosystem services provided by transpiration, including benefits that cool the air, reduce stormwater runoff, and improve air quality. The most popular studied species were *Fraxinus pennsylvanica*, *Acer platanoides*, and *Liquidambar styraciflua*.

Within the 50 studies that measured a unique set of data, 26 used the sap flow method, nine used lysimeters or weighed water loss gravimetrically, and 15 used gas exchange or stomatal conductance methods with or without additional modeling. Most studies took place in the United States (26), followed by Europe (10), East Asia (8), Mexico (3), New Zealand (2), and Thailand (1).

3.2 Urban tree water use

Table 3-2 summarizes the average urban tree water use rates from the 15 studies that measured transpiration during the growing season and reported mean values. Table 3-3 summarizes the same content for the two studies that measured transpiration for one or more years. Urban tree water use measured during the growing season ranged from 0.2 L per day for a small potted plant to 176.9 L per day for a tree with a DBH of 56.8 cm. Plots of urban tree water use versus tree size showed positive, linear relationships (Figure 3-1 and 3-2), and Spearman's rank correlation tests confirmed significance for urban tree water use versus DBH (rho = 0.58, n= 40, $p < 0.0001$), as well as urban tree water use versus canopy area (rho = 0.76, n= 22, $p < 0.0001$). Results from linear regression models (Table 3-4) suggest that DBH and canopy area are strong predictors of urban tree water use ($p < 0.001$).

Table 3-2 also reports the average water use values from the 7 studies that met our two conditions of monitoring urban tree transpiration continuously (3+ months) during the growing season, and either in a humid climate or in irrigated conditions if in a semi-arid climate. The data from these studies were used in our tree type, wood structure, and management context analyses and represent mature trees with a mean DBH ranging 11.1 to 67 cm and a water use average of 49.13 L per day (Table 3-2).

3.3 Urban tree water use as explained by tree type

Based on meta-analysis results, during the growing season, deciduous trees transpire significantly more water on a daily basis compared to evergreen trees (Table 3-5; Figure 3-3; $\chi^2(1) =8 .6$, p = 0.0034). On an annual basis, the opposite is found; average transpiration is significantly higher for evergreen trees, on a daily basis, compared to deciduous trees ($\chi^2(1) = 7.04$, p = 0.008; Table 3-5; Figure 3-4).

Tree wood structure can also explain mean tree water use in the growing season (Table 3-5). Based on linear mixed effect models, transpiration rate is not significantly different between diffuse-porous, ring-porous, and semi-ring porous species (i.e. hardwood trees). Therefore, we grouped these three groups into a "hardwood species" category to compare to conifers, or softwood species. We found that transpiration is significantly higher on a daily basis for hardwood species compared to conifer species in the growing season ($\chi^2(1) = 10.95$, p =0 .0009; Figure 3-5).

Table 3-2: Average urban tree transpiration rates from studies that measured water use during the growing season and reported mean values. Values in parentheses represent the standard error. DBH = diameter at breast height (cm). Here, n is the number of reported mean values used to calculate the overall average urban tree transpiration rate.

Studies included	**Tree water use ($L\ d^{-1}$)**	**Tree water use per DBH ($L\ d^{-1}\ cm^{-1}$)**	**Tree water use per unit leaf or canopy area ($mm\ d^{-1}\ m^{-2}$)**	**Daily sum of sap flux density ($g\ cm^{-2}\ d^{-1}$)**
All	37.02* (5.34)	1.27 (0.15)	1.65 (0.34)	140.22 (17.19)
	n = 50	n = 46	n = 32	n = 28
15 studies total	42 taxa	40 taxa	25 taxa	25 taxa
Those that reported mean values and monitored continuously for 3+ months in humid climates or irrigated conditions	49.13** (6.56)	1.49 (0.17)	1.85 (0.68)	150.48 (16.87)
	n = 33	n = 33	n = 13	n = 26
7 studies total	29 taxa	29 taxa	13 taxa	24 taxa

* Represents average from trees with DBH ranging from <1 to 67 cm.
** Represents average from trees with DBH ranging from 11.1 to 67 cm.

Table 3-3: Average urban tree transpiration rates from studies that measured water use for one or more years in trees with DBH ranging from 8.9 to 43.85 cm. Values in parentheses represent the standard error. DBH = diameter at breast height (cm). Here, n is the number of reported mean values used to calculate the overall average urban tree transpiration rate.

Studies included	**Tree water use ($L\ d^{-1}$)**	**Tree water use per DBH ($L\ d^{-1}\ cm^{-1}$)**	**Tree water use per unit leaf or canopy area ($mm\ d^{-1}\ m^{-2}$)**	**Daily sum of sap flux density ($g\ cm^{-2}\ d^{-1}$)**
All	22.89 (3.43)	0.83 (0.12)	0.79 (0.14)	118.95 (5.61)
	n = 12	n = 12	n = 12	n = 6
2 studies total	12 taxa	12 taxa	12 taxa	6 taxa

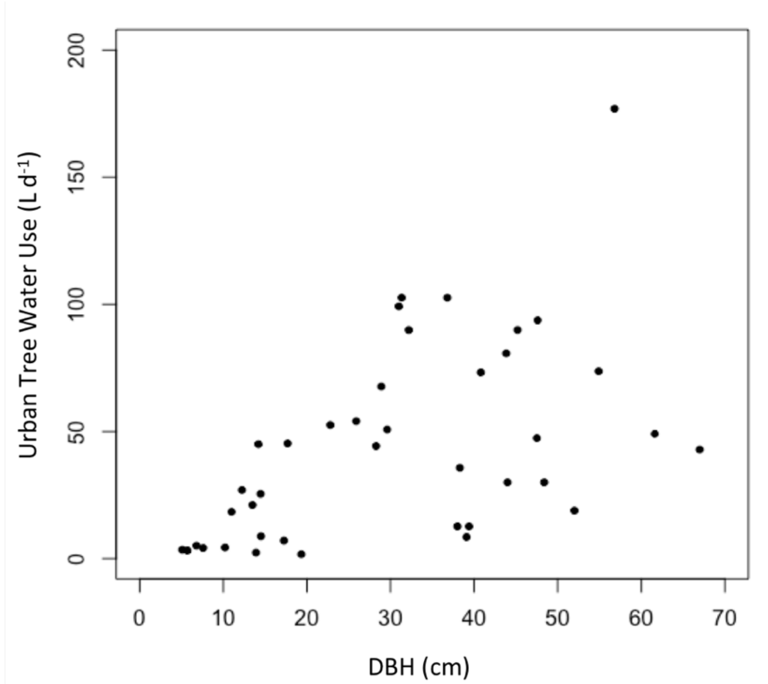


Figure 3-1: Mean water use by urban trees is positively related to mean DBH based on Spearman's rank correlation (rho = 0.58, n= 40, p < 0.0001). Values are from studies that reported both mean water use and DBH of trees that were not described as showing signs of water stress.

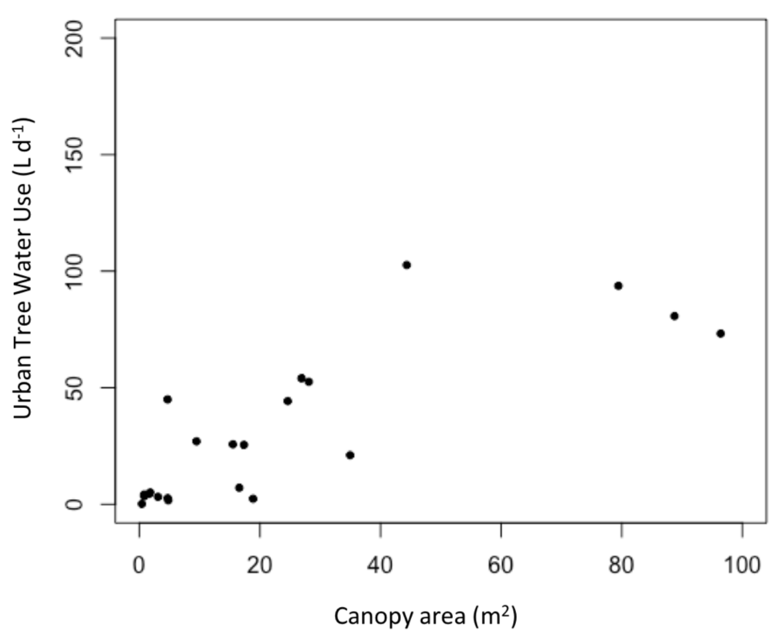


Figure 3-2: Mean water use by urban trees is positively related to mean canopy area based on Spearman's rank correlation (rho = 0.76, n= 22, p < 0.0001). Values are from studies that reported both mean water use and mean canopy area of trees that were not described as showing signs of water stress.

Table 3-4: Results from linear regression analyses evaluating tree size (DBH or canopy area) as a predictor of urban tree water use. CA = canopy area; DBH = diameter at breast height.

Linear Regression Model	Intercept	Slope	F	R^2	p	Model assumptions not met
Tree Water Use ~ DBH	7.63	1.20	14.7	0.28	< 0.001	Homoscedastic residuals
Log10(Tree Water Use) ~ log10(DBH)	-0.23	1.18	32.19	0.46	< 0.0001	Normality of predictor and residuals, homoscedastic residuals
Log10(Tree Water Use) ~ log10(CA)	0.28	0.84	38.55	0.66	< 0.0001	none

Table 3-5: Meta-analysis results of urban tree transpiration rates per tree type classification. Values in parentheses represent the standard error. n is the number of reported mean transpiration values used to calculate the overall mean tree water use per DBH. DBH = diameter at breast height.

Tree classification	Mean tree water use per DBH During growing season ($L\ d^{-1}\ cm^{-1}$)	Mean tree water use per DBH On annual basis ($L\ d^{-1}\ cm^{-1}$)
Deciduous	1.87 (0.22) n = 19	0.68 (0.20) n = 8
Evergreen	0.98 (0.23) n = 12	1.02 (0.10) n = 4
Conifer	0.81 (0.21) n = 10	NA
Diffuse-porous	1.93 (0.29) n = 12	NA
Ring-porous	1.66 (0.38) n = 5	NA
Semi-ring-porous	1.97 (0.28) n = 4	NA

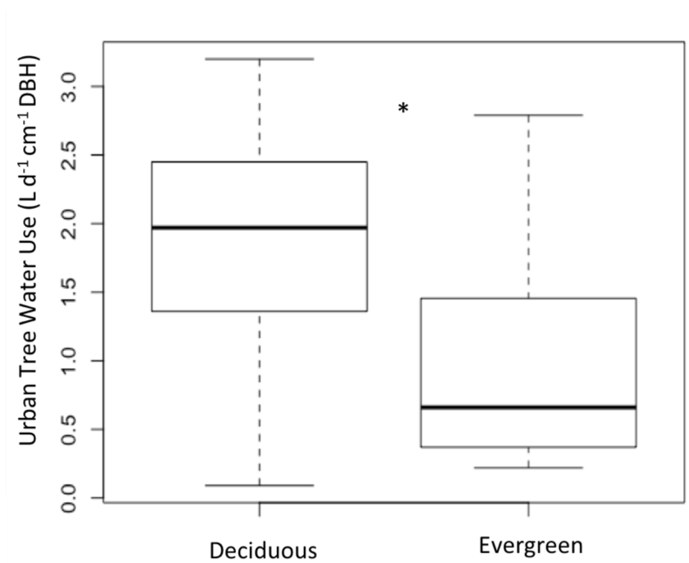


Figure 3-3: Growing season transpiration rates expressed as urban tree water use standardized by DBH for deciduous and evergreen taxa. DBH = diameter at breast height. * indicates significant differences based on linear mixed effects models ($\chi^2(1)$ =8 .6, p = 0.0034).

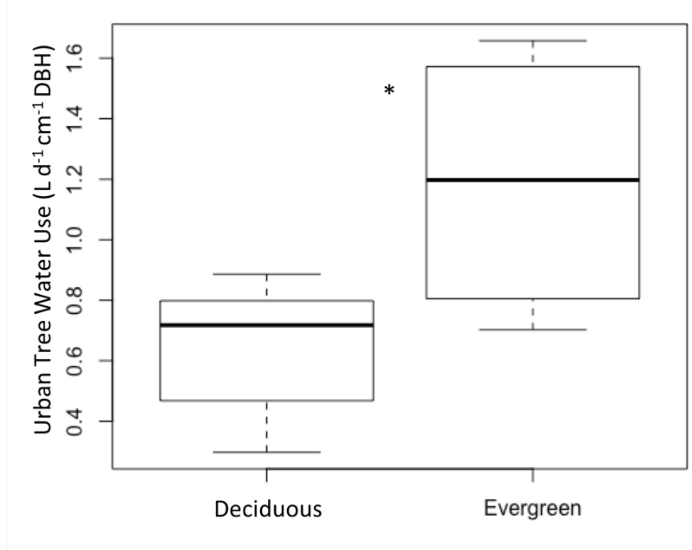


Figure 3-4: Year-long transpiration rates expressed as urban tree water use standardized by DBH for deciduous and evergreen taxa. DBH = diameter at breast height. *indicates significant differences based on linear mixed effects models ($\chi^2(1) = 7.04$, $p = 0.008$).

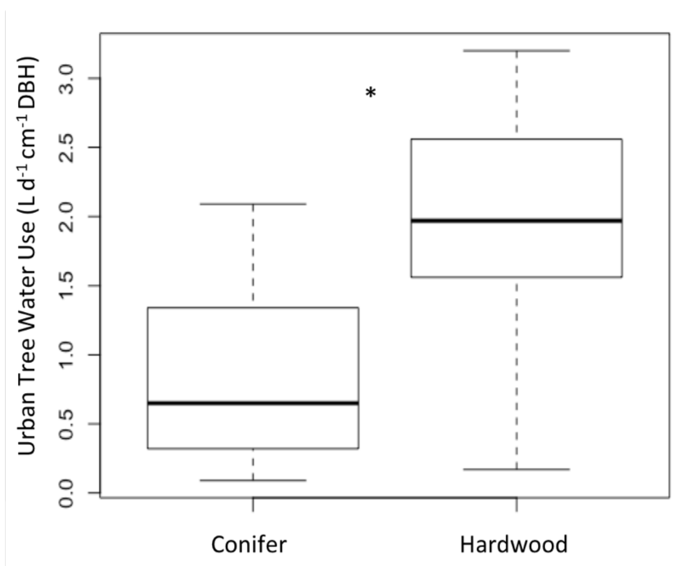


Figure 3-5: Growing season transpiration rates expressed as urban tree water use standardized by DBH for conifer and hardwood taxa. DBH = diameter at breast height. * indicates significant differences based on linear mixed effects models ($\chi^2(1) = 10.95$, p =0 .0009).

3.4 Urban tree water use as explained by the management context

On average, trees over impervious surfaces demonstrate the highest growing season transpiration rates, followed by trees over pervious surfaces and trees in forest patches (Table 3-6). The differences in transpiration for the three groups is significant according to linear mixed effects models ($\chi^2(1) = 11.1$, $p =0 .004$).

Table 3-6: Meta-analysis results of mean growing season transpiration rates standardized by DBH and per management context. Values in parentheses represent the standard error. n is the number of reported mean transpiration values used to calculate the overall mean tree water use per DBH. DBH = diameter at breast height.

Management Context	Transpiration rate (L d^{-1} cm^{-1} DBH)
Tree in forest patch	0.88 (0.43) n = 6
Tree over pervious surface	1.42 (0.19) n = 23
Tree over impervious surface	2.82 (0.22) n = 4

4.0 Discussion

4.1 Urban tree water use as explained by tree size

We assessed the relationship between urban tree size and daily transpiration in the growing season. Our results show that transpiration is positively correlated with DBH and canopy area (Figures 3-1 and 3-2). This contrasts what was found for urban trees in Los Angeles, USA (Pataki et al., 2011) and in Dalian City, China (Chen et al., 2012). However, studies in non-urban areas have also derived a positive relationship between tree size and water flux (Meinzer, Bond, Warren, & Woodruff, 2005; Meinzer, Goldstein, & Andrade, 2001; O'Grady, Eamus, & Hutley, 1999). O'Grady et al. (1999)

and Meinzer et al. (2011) report that DBH best explained the variation in water use or total daily sap flux density by trees in Australian and Panamanian forests, respectively. The results from linear regressions suggest that either DBH or canopy area data could be used to estimate water use by urban trees.

However, because the model of log-transformed mean tree water use vs. mean canopy area had a higher R^2 value and met all linear regression model assumptions, it may be a better model for estimating urban tree water use. The plot of urban tree water use versus DBH (Figure 3-1) does show some variation in mean transpiration values with mean tree size, attributing to the lower R^2 values observed in the linear regressions. This variation could be due to an inconsistent relationship between the diameter of water-conducting sapwood and the diameter of the entire stem, which has been observed to be different across tree species and sizes (Meinzer et al., 2005). In addition, different species within the same DBH class have been shown to vary in how they respond depending on environmental conditions and water availability (Chen et al., 2012). Further, studies that modeled or evaluated the effects of tree size (e.g., sapwood area or tree biomass) on transpiration have established different predictive functions for angiosperm and gymnosperm groups (Litvak, Mccarthy, & Pataki, 2017; Meinzer et al., 2005). Therefore, the observed variation may also be due to differences in tree type.

4.2 Urban tree water use as explained by tree type

Indeed, additional results from this meta-analysis affirm that tree type can explain urban tree water use standardized by DBH. We found that functional group (i.e., deciduous vs. evergreen) can explain urban tree water use in the growing season. Only two studies from our literature review previously examined this question in urban areas. Giraldo et al. (2015) found that water use by the deciduous species *Liquidambar styraciflua* occurred at a ratio of 2:1 compared to the evergreen pine species *Pinus taeda* in a suburban forest near Atlanta, USA, during late spring through summer. This result agrees with our finding that, during the growing season, temperate deciduous trees transpire more water on a daily basis compared to temperate evergreen trees. As explained in Peters et al. (2010), this may be due to lower leaf-level transpiration rates that are typical for evergreen needleleaf trees. In their study, however, they found that evergreen trees exhibited higher growing season transpiration rates per unit canopy area compared to deciduous broadleaf trees from late spring through November in Minneapolis-Saint Paul, USA. They explained that the difference was due to the relatively small canopy area, high leaf area index, and longer active growing season of the evergreen needleleaf trees (Peters et al. 2010). When we calculated the water use per cm DBH for the same trees, we found it to be higher for the deciduous trees, on average (2.04 ± 0.14 L d^{-1} cm^{-1}), compared to the evergreen trees (1.83 ± 0.26 L d^{-1} cm^{-1}). This highlights a potential difference when standardizing tree water use to unit canopy area versus DBH. Due to a lack of data points representing evergreen urban tree water use in the growing season per unit canopy area, we were unable to investigate this further in this meta-analysis.

Because evergreen trees keep their foliage throughout the entire year, they play an important role in providing canopy-dependent benefits in urban settings when deciduous trees are leafless (Clapp, Ryan, Harper, & Bloniarz, 2014). Evergreen trees have been shown to exhibit more conservative water-use strategies compared to deciduous trees (Mediavilla & Escudero, 2004; Peters et al., 2010; Tomlinson et al., 2013), but their longer leaf phenology could lead to greater transpiration amounts, in total, over the course of a year. Peters et al. (2010) and Wang et al. (2011, 2012a, 2012b) are the only studies, to our knowledge, that have explored annual transpiration rates of urban trees, including both deciduous and evergreen species. Transpiration values from both studies were included in this meta-analysis and represent eight deciduous and four evergreen taxa. Collectively, the results from both studies show that daily transpiration rates on an annual basis are higher for evergreen trees than for deciduous trees. However, some studies in non-urban areas suggest contrasting results. For instance, Catovsky, Holbrook, & Bazzaz (2002) found that the annual water flux was greater for broadleaf deciduous species *Acer Rubrum* and *Quercus rubra* compared to the evergreen species *Tsuga canadensis*. *Tsuga canadensis* showed higher transpiration rates during the dormant season, but the species also transpired less on an annual basis compared to deciduous *Betula lenta* in Daley et al. (2007). It is important to note that evergreen trees may not use water all year round as many evergreen species enter dormancy in part of the winter (Chan & Bowling, 2017), thus affecting total annual transpiration amounts as well as reducing the magnitude of transpiration-related benefits during the winter months.

The wood structure of a tree can also explain differences in urban tree water use. Most deciduous trees are hardwoods, with either diffuse- , semi-ring-, or ring-porous wood structures, while most evergreen species are conifers. This was the case for the taxa included in this meta-analysis, except for *Metasequoia glyptostroboides* (deciduous conifer), *Eucalyptus grandis* (evergreen hardwood), *Ficus microcarpa* (evergreen hardwood), and *Brachychiton spp.* (categorized as a “separate group” in Pataki et al., 2011). We found that daily transpiration rates for hardwood species was higher than daily transpiration rates for conifers during the growing season. This pattern is supported by what is known biologically in regards to water transport in the two types of wood structures. Water transport in conifers occurs only through tracheids, relatively small single-celled conduits in the xylem. Hardwood species, on the other hand, have both tracheids and vessel elements. Vessel elements are larger, multiple-celled conduits that are more effective at conducting water (Peters et al., 2010; Sperry, 2003). It is likely that urban transpiration rates are higher on average for hardwoods due to the presence of vessel elements in their xylem anatomy.

Among the hardwood types, some studies suggest that water flux in ring-porous and diffuse-porous species may be different depending on the micro-climate conditions. Ring-porous trees, for instance, exhibited strong stomatal regulation to high VPD in semi-arid urban environments (Bush et al., 2008; Litvak et al., 2012). Diffuse porous species are less sensitive to high VPD, and show higher maximum transpiration rates in response to peak summer conditions, whereas ring-porous trees show lower but more consistent transpiration rates throughout the entire growing season (Litvak et al., 2012; Peters et al., 2010). In Litvak et al., (2012), urban tree transpiration maxed at ~100 kg per day in conifers, ~150 kg per day in ring-porous species, ~175 kg per day in semi-ring porous species, and ~260 kg per day in diffuse-porous species. The maximum slope of sap flow during the growing season was also higher in large (≥ 30-m-tall), diffuse-porous *Acer platanoides*, *Acer saccarum*, and *Platanus occidentalis* individuals compared to large (~25-m-tall), ring-porous *Fraxinus Americana* individuals in an urban park setting in Lancaster, Pennsylvania, USA (Gotsch et al., 2018). Therefore, given the right urban micro-climate conditions (i.e., increased VPD), diffuse-porous trees are likely capable of using more water than ring-porous trees (Bush et al., 2008). The results from this meta-analysis do show that water use by semi-ring-porous and diffuse-porous taxa is, on average, higher than water use by ring-porous trees during the growing season. However, these differences were not found to be significant, suggesting that water use may be more similar across all hardwoods types despite differences in how they respond to environmental conditions. Studies in natural forests also report varying results depending on how water use is standardized. On a stem-area basis, water use by ring-porous oak trees was less than water use by diffuse-porous maple trees (Taneda & Sperry, 2008) but, on a ground-area or leaf-area basis, oaks used similar or greater amounts of water compared to maples (Catovsky et al., 2002; Taneda & Sperry, 2008).

4.3 Urbanization and management context effects on urban tree water use

Studies report conflicting findings when comparing water use by urban trees to water use by trees in non-urban areas. For instance, McCarthy et al. (2010), Litvak et al. (2011), Pataki et al. (2011), Wang et al. (2011), and Riikonen et al. (2016) report that their measured values of urban tree transpiration were similar or higher than transpiration values reported by other transpiration studies from natural areas. Halverson and Potts (1981) used a lysimeter to weigh the water loss of an urban honeylocust tree and found that it required approximately 155 percent of the water needed by the same species in a non-urban area as estimated by the Penman-Monteith model (Halverson & Potts, 1981). In contrast, in a controlled experimental setting, Rahman, Armson, & Ennos (2014) found reduced sap flux density and evapotranspirative cooling by *Pyrus calleryana* trees in an urban simulation compared to the same species in a non-urban simulation. In this meta-analysis, we assessed the effects of urbanization on transpiration by comparing transpiration differences among urban trees in three urban management contexts—forest patches, trees over pervious areas, and trees over impervious areas. Although trees from all three urban management contexts experience indirect effects from urbanization (e.g., from the UHI effect), we discuss them as a gradient from a least urbanized (i.e., forest patch) to an extremely urbanized (i.e., street) management context based upon the degree of built environment and impervious cover that the trees experience.

Our meta-analysis results suggest that tree water use increases with urbanization, as trees become more isolated and removed from other vegetation. Transpiration was highest by street trees, followed by trees over pervious or turf surfaces, and then trees in forest patch settings. It is important to note that there were relatively few mean values (n= 6) representing five species in forest patches and even fewer mean values (n = 4) representing four tree species over impervious surfaces. Three out of the five species in the forest patch settings were conifers, which may have contributed to the observed lower mean transpiration values we calculated for that management context (Table 3-6). The street values came from the studies Pataki et al. (2011) in Los Angeles, USA, and Riikonen et al. (2016) in Helsinki, Finland. In Pataki et al. (2011), the authors attribute the high water use by *Platanus racemosa* and *Platanus hybrida* street trees to high sap flux rates and deep sapwood which are typical characteristics of *Platanus* species. Interestingly, the street site where these trees were measured experienced relatively mild temperatures and low VPD compared to more inland sites that were farther from the coast (Pataki et al. 2011). In Riikonen et al. (2016), the authors measured water use by *Tilia* x *vulgaris* and *Alnus glutinosa f. pyramidalis* street trees and assumed that sap flux was uniform throughout the entire tree trunk, therefore slightly overestimating tree water use. *Alnus*, like *Platanus*, is also a water-loving genus. Thus, although results from this meta-analysis suggest that street trees in humid climates or irrigated conditions may exhibit higher water use compared to urban trees in pervious and forested areas, it is important to consider that transpiration values from the two studies measuring street trees represent high water-use species or more generous estimates of urban tree water use. Understanding the role of tree water-use traits may be a priority for future studies as it impacts total transpiration amounts and has implications for management of stormwater runoff and urban heat islands.

Yet, within the range of tree management contexts that occur in urban areas, trees over impervious surfaces experience higher levels of temperature and VPD that drive transpiration (Asawa et al., 2017; Montague, Kjelgren, & Rupp, 2000; Salmond et al., 2016; Zipper et al., 2017). In addition, canopies that are more isolated and located over impervious surfaces intercept more sunlight and long-wave radiation (Kjelgren & Montague, 1998). The sunlit leaves of seven tree species in urban areas transpired at a rate 2-6 times higher than shaded leaves in the same individuals (Konarska et al., 2015). Trees along the edge of forested areas tend to show higher transpiration rates than trees in the forest interior, likely due to some of the same reasons (Cienciala et al., 2002; Jan, Hsieh, Ishikawa, & Sun, 2013; Kunert, Aparecido, Higuchi, Santos, & Trumbore, 2015) (Cienciala et al., 2002; Jan et al., 2013; Kunert et al., 2015). Hagishima, Narita, & Tanimoto (2007) showed that transpiration by potted trees in a “low” plant density group was higher than transpiration by potted plants in “medium” and “high” plant density groups. The plants in the center of the “high” plant density group showed lower water usage compared to off-center and edge plants in the same group (Hagishima et al., 2007). Given these findings and patterns, it can be speculated that urbanization increases transpiration demands and perhaps transpiration. Edge-effects from roads and other development are common in urban areas (Alberti, 2005; Villaseñor, Driscoll, Escobar, Gibbons, & Lindenmayer, 2014), and trees experience elevated transpiration demands with increasing impervious cover and isolation from other vegetation (Asawa et al., 2017; Yilmaz et al., 2007; Zipper et al., 2017). The results from this analysis are in agreement with this expected trend by showing transpiration is highest for trees over impervious areas and lowest for trees in forest patch settings.

Many urban tree physiological studies show that water stress caused by soil compaction, lack of soil moisture, and increased VPD is relatively common in trees over impervious surfaces and can lead to decreases in gas exchange and transpiration (Cregg, 1995; Rahman et al., 2014, 2011). The street trees included in this meta-analysis were likely not water stressed during the duration of the studies. The street trees in Pataki et al. (2011) were occasionally irrigated, and as discussed in McCarthy and Pataki (2010) and Litvak et al. (2012), the site likely received runoff and fertilizer inputs from residential yards. Further, the street trees in Riikonen et al. (2016) were irrigated weekly during the first two years of the three-year study. Cermák et al. (2000) measured sap flow of street trees in Brno, Czech Republic during August 1997 with almost cloudless warm weather and relatively high bulk soil water content that was attributed to high rainfall amounts in July. They also found relatively high transpiration rates (calculated to be 3.27 L d^{-1} cm^{-1}) compared to other values from this meta-analysis.

Other studies show that water stress triggered by harsh urban conditions can decrease transpiration by trees. Kjelgren and Montague (1998) found that water loss was greater for potted trees over asphalt than for potted trees over turf in a humid climate, but they found the opposite trend in an arid environment. In the arid location, transpiration by the potted trees over asphalt was limited due to high leaf temperatures and stomatal closure (Kjelgren and Montague, 1998). Trees in a plaza in Seattle, USA, experienced water stress and lower transpiration rates compared to trees in nearby parks (Kjelgren & Clark, 1992). The same was observed for trees with higher proportions of impermeable surfaces in Nebraska, USA, Arizona, USA, and Gothenburg, Sweden (Celestian & Martin, 2005; Cregg, 1995; Cregg & Dix, 2001; Konarska et al., 2015). Ballinas and Barradas (2016a, 2016b) measured transpiration by four species for nine and six days in April and March, respectively, in Mexico City, Mexico, and in comparison to results from other studies found in our literature review, they observed the lowest transpiration rates acquired for mature trees, ranging from 3.59 to 4.35 liters per day. They point out that March and April are two of warmest and driest months of the year and that trees are typically not irrigated in Mexico City. Stomatal conductance by street trees in their study decreased linearly as VPD increased (Barradas et al., 2016a). Thus, although our study suggests that transpiration by street trees may be higher than transpiration by trees in other management contexts, street trees may exhibit reduced transpiration rates if water-stressed or not irrigated.

4.4 Implications for management

Our results suggest that tree size, tree type, and management context can be used as classification schemes to explain patterns in urban tree transpiration rates. Managers can use these classification schemes to estimate urban tree water use at a variety of landscape scales and make informed management decisions about transpiration-based ecosystem services. For instance, managers who have access to DBH or canopy area can use functional relationships, such as those derived in this study, to scale up urban tree transpiration to stand and even city-levels. However, it is important to note the tree size and water use relationships derived in this study (Table 3-4) may not accurately predict water use by trees in drought-stressed situations, since they do not include trees that were described in their corresponding studies as being water-stressed or unirrigated if in a semi-arid location.

The positive relationship found between tree water use and size points to the importance of prioritizing the conservation of larger trees that use more water and are therefore more capable of providing transpiration-dependent ecosystem services. As for planting trees, managers who are interested in promoting ecosystem services to reduce stormwater runoff and cool the ambient air temperature may select species based on tree types that promote higher rates of transpiration. If their goal is to mitigate hot temperatures in the summertime, for instance, they may choose a deciduous or hardwood species. If their goal is mitigate stormwater flows continuously on an annual basis, they may choose to plant an evergreen species. However, if applying this kind of information, managers should also think about biodiversity and planting a variety of species to promote sustainability and resiliency of the built environment (Ahern, 2011). The effects of including a mix of species on the cumulative transpiration-based ecosystem services is something future studies can explore.

The relationship between urban forest structure—defined as the "way vegetation is arrayed in relation to other objects such as buildings" (McPherson et al. 1997)—and ecosystem function is key to understanding how urban trees and forests improve environmental quality in urban areas (Livesley, McPherson, & Calfapietra, 2016; Nowak, Stevens, Sisnni, & Luley, 2002).The findings from this meta-analysis suggest that, as tree settings progress from a more forest-like to a more open-grown, urban-like condition, transpiration may increase due to the micro-climate conditions created by the surrounding environment. This study did not evaluate transpiration on a per unit area basis, however, which may be higher in forest patch conditions due to a higher concentration of trees. Additionally, the observed pattern may not hold true for water-stressed trees over impervious surfaces. Improved soil conditions and watering at street sites may help to prevent stomatal closure and encourage transpiration by trees over impervious surfaces (Bartens, Day, Harris, Wynn, & Dove, 2009; Cregg & Dix, 2001; Stabler, 2008). Compared to using the tree classification schemes, however, the scaling up of the meta-analysis results based on management context would be more difficult to do because managers may not know which trees are water-stressed and which are not. The scaling up of the data would assume that all street trees are not water-stressed.

5.0 Conclusion

The results from this meta-analysis suggest that tree size can be used as a metric to estimate urban tree water use. Further, results affirm that urban tree transpiration is affected by biological and physiological traits that place species into tree-type classifications defined by functional group or wood structure. These classifications can be used to explain tree water use in non-urban environments and here, in this meta-analysis, we show that they can also be used to explain transpiration differences by urban tree species. In contrast to trees in natural environments, however, urban trees are additionally affected by urbanization through altered micro-climates and management contexts. We found that transpiration by an open-grown tree over an impervious surface may be higher than for a tree of the same size located over a pervious surface or in a forest patch, as long as the tree is not water-stressed. These data and patterns suggest that it is possible to generalize how much urban trees transpire based upon tree classification schemes and management contexts. The results can be used by managers to aid the selection of tree traits for plantings, scale up transpiration amounts, and get a better understanding of the transpiration-based ecosystem services provided by urban tree canopy.

Chapter 4 Conclusion

Urban forests and their underlying soils provide multiple ecosystem services that contribute to human well-being by moderating the microclimate, conserving energy, improving air quality, sequestering carbon, (Livesley et al., 2016), and promoting other aesthetic, recreational, and health benefits (Nowak & Dwyer, 2007; Tzoulas et al., 2007). Municipalities can maintain and expand their tree canopies to promote these ecosystem services, increase quality of life, and create more resilient, sustainable cities (Gómez-Baggethun & Barton, 2013). Among the ecosystem services provided by urban trees, there is growing interest in using trees and forests for stormwater management in many urban and suburban parts of the United States. To further the promotion of such strategies, in this thesis I used ecohydrology approaches and a meta-analysis to characterize urban forest ecohydrological functions that reduce stormwater runoff. My research questions were:

1. How much stormwater are urban forest patch soils capable of infiltrating?
2. What soil physical properties are the most important drivers of soil infiltration capacity?
3. How much water is transpired by urban trees?
4. What tree characteristics or management contexts are important drivers of transpiration?

To address the first question, I conducted a study focused on soil infiltration capacity in 21 forest patches in Baltimore, Maryland (Chapter Two). I measured the unsaturated hydraulic conductivity of the surface soil and conservatively estimated that forest patch soils are capable of infiltrating rainfall at a rate of 0.61 cm per hour, on average. This infiltration rate equates to infiltrating approximately 68 percent of Baltimore rainfall amounts based on continuous hourly precipitation data from 1975-2013. Chapter Three of this thesis addressed questions number three and four. The peer-reviewed literature was searched for studies that quantified urban tree transpiration. Based on what is reported in the literature, I found that trees in urban areas average 1.7 mm of water per day per unit canopy or leaf area in the growing season, an amount that equals approximately 47% percent of the May-September rainfall in the Baltimore region (based on monthly precipitation values from https://www.usclimatedata.com/climate/baltimore/maryland/united-states/usmd0591). It is important to note that the calculation of the average value of 1.7 included 4 trees that were studied in semi-arid or dry conditions and were not irrigated, with values ranging from 0.02 to 0.03 mm of water per day. From studies that continuously monitored and reported mean water use for at least three months, including the growing season, and in a location with a humid climate or irrigated conditions, this average number increases to ~1.9 mm of water per day per unit canopy area in the growing season. Both of these average numbers are higher than what was used by the Chesapeake Bay Program's Expert Panel for Urban Tree Expansion and Urban Forest Planting BMPs. To derive the water quality benefits of urban tree and forest BMPs, in their model they used a transpiration value of 1.27 mm per day during the growing season (Law and Hanson, 2016), thus underestimating the transpiration-based stormwater benefits of urban tree canopy.

In Chapter 3, I report urban tree water use values in other units. For instance, I found that a mature tree that is likely not significantly water stressed transpires approximately 50 liters of water per day during the growing season. Further, scaling up results from year-long studies of eight deciduous and four evergreen species, urban trees may transpire approximately 0.8 mm of water per day per unit canopy area, on average. In the Baltimore region, this equates to approximately 26% of the annual precipitation (based on an average annual precipitation value of 1088 from https://en.climate-data.org/north-america/united-states-of-america/maryland/baltimore-10/).

In regards to questions two and four, I found that soil infiltration capacity and tree transpiration rates vary for soils and trees, respectively, depending on certain characteristics. In urban forest patches, soils with high sand content, albeit uncommon in the Baltimore area, showed greater capacity to infiltrate larger rain storm events such as a one-year, 24-hour storm event. In addition to texture, the percent of coarse fragments in the soil, soil bulk density, and soil moisture were also found to be important drivers of infiltration. Soils with lower bulk densities had higher rates of unsaturated hydraulic conductivity, but this relationship was not as strong as the significant and positive relationship between infiltration capacity and percent of coarse fragments (> 2 mm) in the soil. Soil moisture was also an important driver of the unsaturated hydraulic conductivity. In one location, we observed that as a soil gets wetter, the infiltration capacity increases, likely due to the macroporosity effect that promotes water infiltration through macropores as they become saturated. Based on tree transpiration values from the literature, we found that tree size is a strong predictor of tree water use. Deciduous trees transpire more water than evergreen trees during the growing season (on average, 1.87 and 0.98 liters per day per cm DBH, respectively), but evergreen trees transpire more water than deciduous trees on an annual basis (on average, 1.02 and 0.68 liters per day per cm DBH, respectively). Further, diffuse- and ring-porous (hardwood) species use more water than conifers (on average, 1.92 and 0.85 liters per day per cm DBH, respectively). The management context is important, too. Trees over impervious surfaces, if not water stressed, may transpire more water than trees over pervious surfaces and trees in forest patches. I speculate that this is likely due to the higher evaporative demands and solar radiation that they experience (Asawa et al., 2017; Yilmaz et al., 2007; Zipper et al., 2017). Based on values reported in the literature, trees used 0.88, 1.42, and 2.82 L of water per day per cm DBH when located in forest patches, over pervious surfaces, and impervious surfaces, respectively.

Implications

Forest patches are areas of tree canopy ~0.1 hectares or larger, and in Baltimore they account for 34% of City's canopy cover (Avins 2013). The results from this study show that infiltration capacity and transpiration amounts in urban forest patches can be substantial, therefore the conservation and expansion of urban forest patches can help to mitigate stormwater flows in Baltimore. Although soil infiltration capacity was not as great in Baltimore forest patches compared to in designed bioretention cells, results show that micro- and mesopores in urban forest patch soils alone can infiltrate storm events of low to moderate intensities. Moreover, soils with sandier properties were capable of infiltrating as much rainfall as engineered SWGI. The results from the tree transpiration meta-analysis suggests that transpiration also plays an important role in urban hydrology. Urban tree canopy will therefore play an even larger role in the future as cities aim to increase their tree canopy to cover 30 or more percent of the developed landscape. However, the amount of stormwater runoff reduced by urban forest patch soils and transpiration functions depends on characteristics that drive each of those functions. Managers who are looking to mitigate stormwater issues should take this into consideration as they continue to grow their urban tree canopy to promote these ecohydrological functions. Further, the stormwater retention benefits of urban trees and forests are not as great for storm events of larger flows and intensities, as seen in the soil infiltration capacity study in Chapter 2. Yet, overall, the findings from this thesis affirm that urban trees and forests impact urban hydrology via soil infiltration and transpiration. Stormwater mitigation strategies should continue to use engineered SWGI practices, but couple those strategies with urban trees and forests, to manage rainfall events of varying sizes throughout the entire year.

Future Research Needs

Urban trees and forests reduce stormwater runoff and flooding and are an important component of SWGI portfolios in the built environment. Altogether, the results from this thesis suggest that 68% of rainfall can be infiltrated by urban forest patch soils and on an annual basis approximately 26% of that soil water may be returned to the atmosphere via transpiration. There are several questions remaining that can be explored in future investigations. For instance, because urban forest patches contain multiple canopies that overlap together, the total transpiration amount per unit area is higher for that type of urban tree canopy compared to a pervious lawn area with more isolated trees. In addition, using data from this thesis, a future modeling study can be completed to estimate the change in transpiration-based ecosystem services that will come for a city aiming to increasing its canopy cover to a certain percentage. An additional study may analyze soil infiltration capacity in other urban tree canopy types, as infiltration may not be as high for soil under street trees due to soil compaction that hinders infiltration (Peters et al., 2010; Rahman et al., 2014). In addition, it is important to note that our soil infiltration capacity analyses assume that no additional losses of rainfall occurred from canopy interception. Additional losses from this ecohydrological function would also contribute to the mitigation of stormwater runoff. Some studies have looked at this function alone (e.g. Inkiläinen et al. 2013 and Nytch et al. 2018), but no study, to my knowledge, has empirically quantified and assessed the cumulative effects of all urban tree ecohydrological functions. Studies that pursue this question should analyze spatial differences among different types of urban tree types, as well as temporal dynamics that account for the reduced canopy benefits during the leaf-off period.

In addition to contributing to the broader understanding of the benefits provided by urban tree land uses, there is a need to better understand the role that trees play in designed SWGI practices, such as in bioretention systems. A study by Scharenbroch et al. (2015), for example, has shown promising results by finding that transpiration by trees in bioswales accounted for 46 to 72% of the total water outputs from the system. Finally, the importance of landscape topography and flow paths into soils that underlie trees and forests should be more carefully studied to inform stormwater practitioners and managers toward the goal of maximizing the stormwater retention benefits of urban tree canopies.

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