ADVANCING EARTH AND SPACE SCIENCES

Water Resources Research

RESEARCH ARTICLE
10.1029/2025WR041340

Special Collection:
Key Hydrological Processes and Its Controlling Factors in Terrestrial Ecosystems

Large Streamflow Differences Between Forested and Urbanized Watersheds in the Energy-Limited Eastern United States: The Role of Evapotranspiration and Impervious Surfaces

Ge Sun[1], Zihao Bian[2], Kul Khand[3], Peter V. Caldwell[4], Johnny Boggs[1], Chen Wang[5], Yujuan Chen[5], Ning Liu[6], Yulong Zhang[7], Xi Chen[8], Gabriel B. Senay[9], and Steven G. McNulty[1]

[1]Eastern Forest Environmental Threat Assessment Center, Southern Research Station, USDA Forest Service, Durham, NC, USA, [2]School of Geography, Nanjing Normal University, Nanjing, China, [3]ASRC Federal Data Solutions, contractor to the U.S. Geological Survey Earth Resources Observation and Science (EROS) Center, Sioux Falls, SD, USA, [4]Center for Integrated Forest Science, Southern Research Station, USDA Forest Service, Otto, NC, USA, [5]Department Environmental Sciences, Tennessee State University, Nashville, TN, USA, [6]CSIRO Environment, Canberra, ACT, Australia, [7]Nicholas School of the Environment, Duke University, Durham, NC, USA, [8]Department of Geography, University of Cincinnati, Cincinnati, OH, USA, [9]U.S. Geological Survey Earth Resources Observation & Science (EROS) Center, North Central Climate Adaptation Science Center, Colorado State University, Fort Collins, CO, USA

Key Points:

- Eleven pairs of forest-urban watersheds were examined to quantify benefits of forests in mitigating stormflow
- Compared to urban watersheds, forest watersheds had much lower streamflow during the rainy summer and fall seasons, but not in the winter dormant season
- Urban planning for mitigating stormflow should factor in the evapotranspiration functions of ecosystems in addition to impervious surfaces

Supporting Information:

Supporting Information may be found in the online version of this article.

Correspondence to:

G. Sun,
Ge.Sun@usda.gov

Citation:

Sun, G., Bian, Z., Khand, K., Caldwell, P. V., Boggs, J., Wang, C., et al. (2026). Large streamflow differences between forested and urbanized watersheds in the energy-limited eastern United States: The role of evapotranspiration and impervious surfaces. *Water Resources Research*, *62*, e2025WR041340. https://doi.org/10.1029/2025WR041340

Received 14 JUN 2025
Accepted 15 JAN 2026
Corrected 12 FEB 2026

This article was corrected on 12 FEB 2026. See the end of the full text for details.

Author Contributions:

Conceptualization: Ge Sun, Steven G. McNulty

Abstract

Urban forests and other green infrastructures have been viewed as part of the "Nature-based Solutions" (NbS) to mitigate emerging urban environmental change. This study focuses on the role of evapotranspiration (ET) in regulating water balances of small watersheds in the eastern United States. We compared streamflow and ET patterns at daily, monthly and annual scales and linked these hydrological variables to the physical properties of 11 paired watersheds dominated by forests (FW) or urban (UW) land covers. The annual precipitation ranged from 1028 mm to 1683 mm and potential ET (PET) from 815 mm to 1450 mm. The mean annual flow/precipitation (Q/P) ratios were 0.26 ± 0.13 and 0.41 ± 0.1 for FW and UW, respectively. Overall, UW had lower annual ET (772 mm in UW vs. 947 mm in FW), but higher mean annual and (~58% higher), monthly water yield (17%–186% higher), and peakflow rates (up to 100 times higher) than FW. The streamflow differences between FW and UW were most pronounced during the growing season and early winter (June-November). The mean Q/P ratios for 30 large hurricane events (2016–2021) were 0.12 ± 0.11 and 0.38 ± 0.23 for FW and UW, respectively. The flow rates in the dormant season (around December-May) in UW were similar or lower than FW. We developed conceptual models to explain the seasonal and storm event streamflow differences using background climate (PET), ET, and land surface characteristics. Urban NbS designs should factor in strategies that maximize ET while minimizing impervious surfaces enhancing watershed "sponge" and "pump" functions.

Plain Language Summary

"Urban Syndrome" including "Urban Heat Islands," "Urban Dry/Wet Islands," and "Urban floods" demonstrate the dramatic hydrological impacts of urbanization on natural environments. Urban forests and other green infrastructures have been regarded as a "sponge" to absorb storm runoff and serve as "air conditioners" to cool cities due to their shading and evaporation ecohydrological functions. Therefore, developing green space serves as a "Nature-based Solutions" (NbS) strategy to confront many urban problems that are only increasing with climate change and population rise. However, our knowledge about the forest hydrologic functions (e.g., evapotranspiration, streamflow generation) and relationship to desired ecosystem services (e.g., climatic cooling, flood mitigation) in an urban environment is lacking. This study compared the hydrologic patterns for 11 pairs of small watersheds that are dominated by forests or urban land covers during 2000–2022. We show that forest watersheds had much lower peakflows during 30 large hurricanes in the summer months likely due to the high forest evapotranspiration resulting in large available water storage in soil (i.e., sponge). Our study suggests that NbS strategies should take advantage of nature's evapotranspiration "biological drainage" (i.e., pump) power for reducing urban flooding while minimizing impervious surfaces and soil compaction that reduce soil water storage and infiltration capacity (i.e., maximizing the sponge size).

Data curation: Zihao Bian, Kul Khand, Peter V. Caldwell, Johnny Boggs, Chen Wang, Yujuan Chen, Gabriel B. Senay
Formal analysis: Ge Sun, Zihao Bian, Chen Wang, Ning Liu
Funding acquisition: Steven G. McNulty
Investigation: Ge Sun, Zihao Bian, Peter V. Caldwell
Methodology: Ge Sun, Johnny Boggs, Xi Chen, Steven G. McNulty
Project administration: Ge Sun
Resources: Ge Sun, Steven G. McNulty
Software: Ge Sun, Zihao Bian
Validation: Zihao Bian, Kul Khand
Visualization: Kul Khand
Writing – original draft: Ge Sun
Writing – review & editing: Kul Khand, Peter V. Caldwell, Johnny Boggs, Chen Wang, Yujuan Chen, Ning Liu, Yulong Zhang, Xi Chen, Gabriel B. Senay, Steven G. McNulty

1. Introduction

The global urban population is expected to increase by 2–3 billion from 2015 to 2050 and urban areas are projected to expand by 0.6–1.3 million km^2, representing a 78%–171% increase in urban footprint (Huang et al., 2019). Such permanent land cover changes have shown to cause dramatic disturbances in watershed hydrology (Boggs & Sun, 2011; Diem et al., 2023; Fletcher et al., 2024; Hao et al., 2015; Martin et al., 2017), ecosystem productivity (Chen et al., 2024) and carbon sequestration (Diem et al., 2006), watershed ecosystem function (Sun & Caldwell, 2015; Sun & Lockaby, 2012), local air temperature and humidity (Fang et al., 2020; Hao et al., 2023; Huang, Hao, et al., 2022; Huang, Jin, et al., 2022) and rainfall patterns (Li et al., 2025; Sui et al., 2024), global climate feedbacks (Georgescu et al., 2021; Kalnay & Cai, 2003), and human health (Maxwell et al., 2018). Future climate change is expected to elevate rainfall intensity by ∼12% in the 10-year and 50-year recurrence interval intensity under a 1.5°C warming scenario (Fletcher et al., 2024). An increasing number of studies have indicated that the observed "Urban syndrome" such as "Urban Heat Island (UHI)" (Zhou et al., 2016, 2022), "Urban Dry Island (UDI)" (Hao et al., 2023), and elevated storm runoff (Voter & Loheide, 2018) can be fundamentally explained by ecohydrological processes such as evapotranspiration (ET; Hao et al., 2018) in addition to soil infiltration processes. Understanding the nexus between evapotranspiration (ET) and energy, water, and carbon cycling is important for developing urban "Nature-based Solutions" (NbS) to maximize the benefits (e.g., cooling, flood mitigation) of specific NbS strategies (e.g., urban forests) in different regions (wet vs. arid; Cuthbert et al., 2022) and integrated watershed management (Niemczynowicz, 1999).

In the U.S., urban areas, including city centers and their surrounding suburbs, take up only 5.3% of total land and are home for about 80% of the total population (Alig et al., 2004). Urban population has been increasing rapidly since the 1850s with the growth rate peaking in the 1950–1970s (U.S. Census Bureau, 2023) and urban land area increased about 1 million acre (4,047 km^2) per year during the 2010s across the U.S. (Homer et al., 2020; Nelson et al., 2020). Urbanization was projected to result in a loss of 118,318 km^2 of forestlands between 2005 and 2050 in the U.S. (Nowak & Walton, 2005). A recent U.S. Resource Planning Act (RPA) assessment projected a net loss of forests of 30,756–60,703 km^2 between 2020 and 2070 (USDA Forest Service, 2023). Rigorous studies on the hydrologic effects of land use change and urbanization started as early as the 1940s in the eastern U.S. where forests were a common landcover for urban development (Lull & Sopper, 1969). Urbanization has been perceived as contributing to catastrophic flooding (Abegaz et al., 2024; Hopkins et al., 2015) such as the Johnstown Flood of May 1889 amid tropical storms in Pennsylvania. The large urban population boom in the eastern U.S. has caused concern of water supply shortage (Douglass, 1983) and the expansion of urban-rural interface and forest fragmentation has threatened watershed ecosystem integrity for several decades (DeWalle et al., 2000; Diem et al., 2023; Dow & DeWalle, 2000; Hopper et al., 2025; Nagy et al., 2012; Paul & Meyer, 2001).

However, previous empirical urban hydrological studies mostly focus on the impacts of impervious surface and soils (Fletcher et al., 2024; Oudin et al., 2018; Price, 2011) with little attention to ecohydrological processes such as the effects of vegetation on ET (Oswald et al., 2023). Most existing literature agrees that urbanization increases impervious surfaces and reduces soil infiltration capacity (Price, 2011), resulting in elevated stormflow and peakflow (Khand & Senay, 2022; Rose & Peters, 2001), and total flow (Kumar et al., 2018; Kundu et al., 2017a, 2017b). However, the urbanization effects on baseflow remain ambiguous, with some studies reporting an increase (Hao et al., 2015) or others a decrease (Kauffman et al., 2009; Oudin et al., 2018) in baseflow. Some retrospective studies did not detect any significant trend of annual water yield in urbanized watersheds, likely due to the masking effect of climate change (Kumar et al., 2018) and water management (Rougé & Cai, 2014). The observed variability of hydrologic response to urbanization has been attributed to the differences in the magnitude of urbanization (e.g., imperviousness; Weng, 2001), local climate (e.g., rainfall and temperature; Li, Sun, Caldwell, et al., 2020), land use and land cover (LULC) characteristics (Kundu et al., 2017a), and the temporal scale examined (Weng, 2001).

Modeling studies have been widely used to examine the effects of future land use change and climate change, or to separate their individual contributions to observed changes in streamflow (Braud et al., 2013; Praskievicz & Chang, 2009). However, these models often emphasize on modeling change in water pathways of drainage network and its impacts on flashness of hydrographs (Braud et al., 2013; Praskievicz & Chang, 2009) with simplistic assumptions about the ET or soil infiltration processes (Caldwell et al., 2012; Li, Sun, Caldwell, et al., 2020; Martin et al., 2017). For example, state and federal regulatory agencies have long been using a "generic approximation" to conceptually describe how urban imperviousness affects stormflow, ET, and infiltration (Livingston & McCarron, 1992), significantly underestimating water loss (i.e., ET) from lands (Diem et al., 2023). General hydrological models are often misused to predict urbanization effects on stormflows and groundwater recharge when the ET and subsurface processes are not fully considered (Sun et al., 2023). A lack of quantitative water balance data hiders developing accurate indicators for managing impervious surfaces and flood management (Arnold & Gibbons, 1996).

Current understanding of the hydrologic impacts of urbanization is mostly based on small scale theoretical principles of engineering hydrology (Cronshey, 1986) that often ignore the role of vegetation (Wang et al., 2008) and ET processes, the so called “biological drainage.” The eddyflux research community maintains thousands of tower sites for measuring carbon and ET fluxes, but only a fraction of these studies focuses on the urban environment (Lipson et al., 2022). For example, among the 745 registered AmeriFlux sites, only 18 are considered urban and built-up lands. The traditional “Paired Watershed Experiment” approach for detecting the hydrologic effects of deforestation is difficult to implement in long-term urban watershed research (e.g., Baltimore Urban Long Term Ecological Research; Bhaskar & Welty, 2012). Budyko-based empirical (Teuling et al., 2019; Teuling et al., 2019; Wang & Hejazi, 2011) and process-based mathematical models (C. L. Li et al., 2016; J. K. Li et al., 2016; Pumo et al., 2017; Zipper et al., 2018) have been used to project the hydrologic effects of natural and anthropogenic disturbances including urbanization, climate change and variability, and hurricanes (Worley et al., 2022) on individual watersheds. Our previous continental modeling studies (Caldwell et al., 2012; Li, Sun, Caldwell, et al., 2020; Li, Sun, Cohen, et al., 2020) in the U.S. show “urbanization impacts are not created equal” because background climate and vegetation ET plays an important role in affecting watershed water balances.

The motivation of this study was to quantify the benefit of forest cover in reducing urban runoff and stabilizing streamflow through ET, an ecohydrological process that is currently overlooked by existing literature. To achieve this goal, we contrasted watershed hydrology (i.e., annual water yield, monthly, and daily flow) for 11 forest-urban watershed pairs across the eastern U.S. We hypothesized that the hydrology of headwater watersheds is affected by urbanization at all temporal scales by an increase in impervious surface area (ISA) and reduction in ET. Specifically, our hypotheses were: (a) Forest watersheds have lower runoff than urban watersheds due to lower ISA and higher ET (Hypothesis H1), (b) H1 is mostly true during the growing season when ET overwhelms the effects of ISA on soil infiltration and groundwater recharge; and little differences in streamflow patterns in the dormant season (Hypothesis H2), and (c) the magnitudes of hydrologic influences of urbanization vary due to background climate and urbanization severity (i.e., ISA%; Hypothesis H3). These hypotheses guided our empirical comparison study on the hydrologic differences between urban- and forest-dominated watersheds across a large physiographic gradient in the eastern U.S.

2. Data and Methods

2.1. Paired Watersheds

Based on the U.S. Geological Survey's historical streamflow (U.S. Geological Survey, 2026) and the National Land Cover Database (NLCD; Wickham et al., 2023), we identified 11 pairs of gauged small watersheds across the eastern U.S. Each pair consists of one forest-dominated and one urban-dominated watershed (Figure 1 and Table 1). These watersheds met the following rigorous criteria. First, these paired watersheds are gaged with long (>7 year) streamflow records for the same period around calendar year 2000 (Table 1). Second, watersheds in each pair are located in close geographic proximity ensuring similar meteorological characteristics (i.e., daily precipitation distribution). Daily rainfall frequency analysis was performed to make sure there are not significant differences in overall precipitation patterns at daily, monthly, and annual scales. Third, these small watersheds are in headwater areas drained by first and second-order streams, so they are not heavily influenced by large human activities (e.g., inter-basin transfers or impoundment).

The size of 11 pairs of forest-urban watersheds varies between 1.8 and 347.1 km^2 (Table 1), with an average 68.0% forest cover (38% Deciduous Forest, 15.6% Conifer Forest, and 14.2% Mixed Forest; 9% Urban, 2% Impervious Surface Area) for 11 forest-dominated watersheds and 78.9% urban cover and 49.6% impervious surface area for 11 urban-dominated watersheds (NLCD data for the year 2019). For forest watersheds, the average annual precipitation (P) and PET were 1,366 and 1,080 mm, respectively, for the study periods (2000–2022). For urban watersheds, P and PET were 1,367 mm and 1,119 mm, respectively. These watersheds span a large latitude gradient from 29.79°N in Florida (Pair 9) to 42.25°N in Massachusetts (Pair 1). Details on data used in this study are provided in Table S1 of Supporting Information S1.

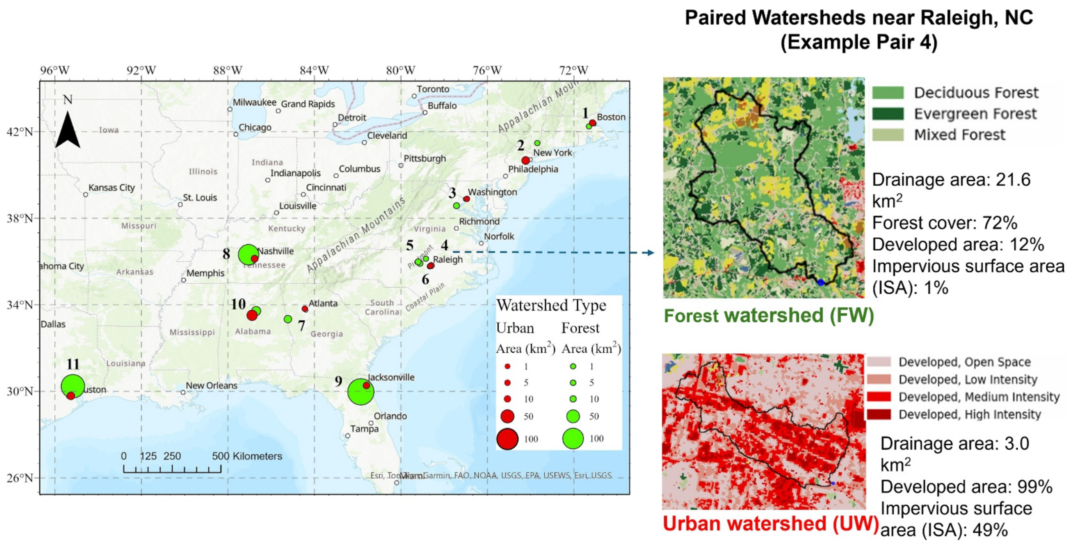

Figure 1. Locations of 11 pairs of forest-urban watersheds identified by pair ID in the eastern U.S. Inset shows land use differences for Pair 4 near Raleigh, North Carolina representing the piedmont region of the southeastern U.S. Watershed size and type are indicated by size and color of the solid dots.

2.2. Hypothesis Testing

To test the proposed first two hypotheses (H1, H2) that forest-dominated watersheds (denoted as FW) have lower runoff than urban watersheds (denoted as UW), but the differences are temporally variable, we compared mean annual streamflow (Q) and runoff coefficients (Q/P), and daily and monthly flow patterns. Daily streamflow records at 22 USGS stream gages were assembled along with precipitation, PET, and watershed characteristics (Table 1).

For testing the third hypothesis (H3) that magnitudes of hydrologic (Q and ET) differences vary due to background climate and urbanization severity, the stepwise regression analysis was used to determine major biophysical controls on streamflow and ET differences between FW and UW. These factors included land cover types and areas, impervious surface areas (ISA), background climate (Dryness Index, PET/P), vegetation phenology and leaf area index (LAI) as determined by MODIS data (Table S1 in Supporting Information S1). In lieu of monthly ET measurements at the watersheds level, the USGS Operational Simplified Surface Energy Balance (SSEBop) model Version 6 (Senay et al., 2023) was adopted to estimate monthly ET. The SSEBop ET model is a thermal-based simplified surface energy model based on the principles of satellite psychrometry. The global SSEBop ET products were created using a series of data streams from the Visible Infrared Imaging Radiometer Suite (VIIRS), NOAA-20 satellites, Daymet (Thornton et al., 2022), CHELSA (Karger et al., 2021), and ERA-5. The global SSEBop ET data stream represents a product of ET fractions generated from the VIIRS thermal imagery and FAO Grass Reference ET (ETo) calculated from weather data fields every ∼10 days using the Penman-Monteith Equation (TerraClimate and CSIRO-based data sets Senay et al., 2023; https://earlywarning.usgs.gov/fews). In this study, the accuracy of the SSEBop ET products was assessed at the annual scale against ET estimated using the long-term watershed water balance method: mean ET = precipitation (P) − streamflow (Q).

3. Results

We characterized differences in flow patterns at daily, monthly, seasonal, and annual scales in the context of local climate, seasonal ET, vegetation dynamics, and urbanization magnitude (ISA%). Attribution analysis was performed to explain the observed hydrologic differences in ET and water yield in relation to ISA, background climate, storm size, and antecedent watershed water storage deficit as indicated by baseflow prior to the peakflow events.

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Table 1
A Summary of Mean Hydrometeorological (Annual Precipitation, P and Potential Evapotranspiration, PET) and Land Use and Land Cover (NLCD 2019) (Wickham et al., 2023) Characteristics for 11 Pairs of Forest and Urban Watersheds (U.S. Geological Survey, 2026)

Watershed pair ID	Approximate location	Watershed type	USGS stream gage	Area (km^2)	Forest cover (%)	Deciduous forest (%)	Conifer forest (%)	Mixed forest (%)	Urban cover (%)	Impervious surface area (ISA) %	P (mm)	PET (mm)	Dryness index PET/P	Data period
1	Boston, MA	Urban	01103025	21.7	4.3	2.7	0.7	0.9	90	61	1,215	831	0.68	2016–2022
	Boston, MA	Forest	01103455	9.6	60.5	24.2	9	27.4	19	5	1,256	824	0.66	
2	Elizabeth, NJ	Urban	01393450	43.8	1.3	1.2	0	0.1	97	57.1	1,347	941	0.7	2016–2022
	Lake Carmel, NY	Forest	01374598	10.2	80	77.7	0.1	2.2	11	2.2	1,303	815	0.63	
3	Washington DC	Urban	01651800	8.7	11.2	10.6	0	0.6	87	37.8	1,281	1,016	0.79	2016–2022
	Independent Hill, VA	Forest	01658500	19.7	81.9	50.3	5.4	26.1	7	0.9	1,262	1,012	0.8	
4	Raleigh, NC	Urban	0208735012	3.0	0.3	0.3	0	0	99	49.2	1,482	1,105	0.75	2016–2022
	Chapel Hill, NC	Forest	02097464	21.6	68.2	35.9	14.5	17.8	12	1	1,386	1,063	0.77	
5	Raleigh, NC	Urban	0208732885	17.7	3.5	0.9	1.3	1.3	29.6	29.6	1,182	1,101	0.93	1990–2009
	Chapel Hill, NC	Forest	02096846	19.5	72.5	35.8	14.2	22.5	0.3	0.3	1,197	1,065	0.89	
6	Raleigh, NC	Urban	0208732534	1.8	0	0	0	0	56	56	1,069	1,101	1.03	2000–2007
	Durham, NC	Forest	0208650112	7.6	88.8	61.7	7.7	19.5	1	1	1,158	1,052	0.91	
7	Atlanta, GA	Urban	02336313	30.6	8.8	5.6	2.6	0.6	90	57.5	1,425	1,137	0.8	2016–2022
	Atlanta, GA	Forest	02338523	251.7	72.9	36.4	27.6	8.9	4	0.6	1,617	1,117	0.69	
8	Nashville, TN	Urban	03431300	7.0	9.5	4.6	0.1	4.7	61.7	61.7	1,307	1,069	0.82	2000–2020
	Nashville, TN	Forest	03431800	43.5	55	49.9	0.4	4.6	0	0	1,271	1,044	0.82	
9	Jacksonville, FL	Urban	02246515	25.6	4.4	0.1	4.3	0	75	37.7	1,491	1,395	0.94	2016–2022
	Jacksonville, FL	Forest	02245500	347.1	51.6	0.5	51.0	0	5	1	1,454	1,379	0.95	
10	Birmingham, AL	Urban	02458450	86.8	7.6	2.9	1.0	3.7	89	42.5	1,666	1,178	0.71	2016–2022
	Birmingham, AL	Forest	02455980	71.0	58.5	48	4.5	6	35	8.4	1,658	1,141	0.69	
11	Houston, TX	Urban	08075770	41.7	2.4	0.3	1.2	1	94	55.9	1,568	1,439	0.92	2016–2022
	Houston, TX	Forest	08071000	303	58	0.1	37.2	20.7	10.0	1.7	1,461	1,369	0.94	

Note. *PET is calculated by the Hamon method with a correction factor of 1.2 (Lu et al., 2005).

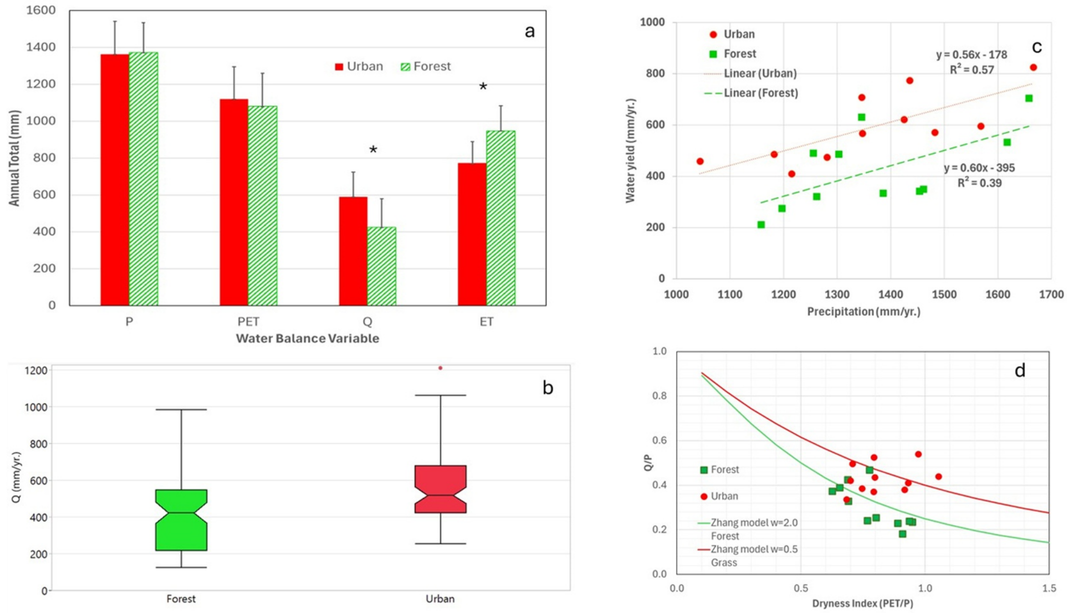

Figure 2. A comparison of mean annual water balances for 11 paired watersheds in eastern U.S. (a) key water fluxes. The symbol * indicates significant differences at a $p < 0.05$ level, (b) Box and Whisker diagrams show variability of streamflow (Q) differences between urban and forest watersheds, (c) Annual Precipitation (P)-Q relationship showing strong control of P on Q in the study region, and (d) Relations between mean runoff coefficient (Q/P) versus dryness index (PET/P) in the context of the Budyko framework models for typical forest and grass land covers (Zhang et al., 2001).

3.1. Streamflow Comparison at Multiple Temporal Scales

We compared watershed hydrologic characteristics of each urban-forest pair and the 11 pairs together by pooling all data at different temporal scales. The annual scale comparisons demonstrate the overall large range of changes in water balances (streamflow, ET) due to urbanization under different climatic regimes across the latitude gradient (Figure S1 in Supporting Information S1). The monthly and seasonal comparisons show intra-annual differences in streamflow as influenced by seasonal (winter vs. summer) climatic patterns and vegetation phenology. Daily flow analysis focuses on contrasts of low flow during dry periods in the peak growing season and large stormflow events under extreme rainfall episodes such as hurricanes occurring commonly from June to November in the U.S. eastern seaboard.

3.1.1. Annual Water Yield

As a whole, mean annual streamflow (Q) of the 11 urban watersheds (590 ± 134 mm) were significantly ($p < 0.01$) higher than 11 forest watersheds (425 ± 155 mm), representing a 39% difference (Figures 2a and 2b). For individual watershed pairs, only Pair 4, 5, 8, 9, 11 showed significant (t test $p < 0.05$) differences. When normalized by precipitation, mean Runoff Coefficient (Q/P) of urban watersheds (0.43 ± 0.07) was 39% higher than the forest watersheds (Q/P = 0.31 ± 0.09). Conversely, the mean annual ET of urban watersheds (772 ± 117 mm), calculated as P-Q, was 18.5% lower than forest watersheds (947 ± 137 mm). Annual ET estimated by the SSEBop model compared well to the water balance-derived ET for the 11 paired watersheds with a small overall overestimation of 6.9%, 0.3% for UW and 12% for FW (Figure S2 in Supporting Information S1).

Mean annual water yield (Q) responded linearly to precipitation (Figure 2c). Among the 11 pairs, the largest difference in mean annual Q (598 mm; 941 mm for UW vs. 342 mm for FW) was found in Pair #9 (Jacksonville, FL), representing a 1.6-fold higher Q/P in the UW (with 38% of ISA, Q/P = 0.63) than the FW dominated forest lands (1% ISA, Q/P = 0.24; Table 1 and Figure 1c). In contrast, mean annual water yield for UW in Pair 1 (Boston, MA; Q = 410 mm, Q/P = 0.34) with the lowest PET was lower than the FW (Q = 490 mm, Q/P = 0.39).

The Budyko framework suggests that all watersheds, except Pair 6 (Raleigh, NC), are "energy limited" hydrologic systems (PET/P < 1.0) characterized by a humid region with annual precipitation exceeding 1000 mm

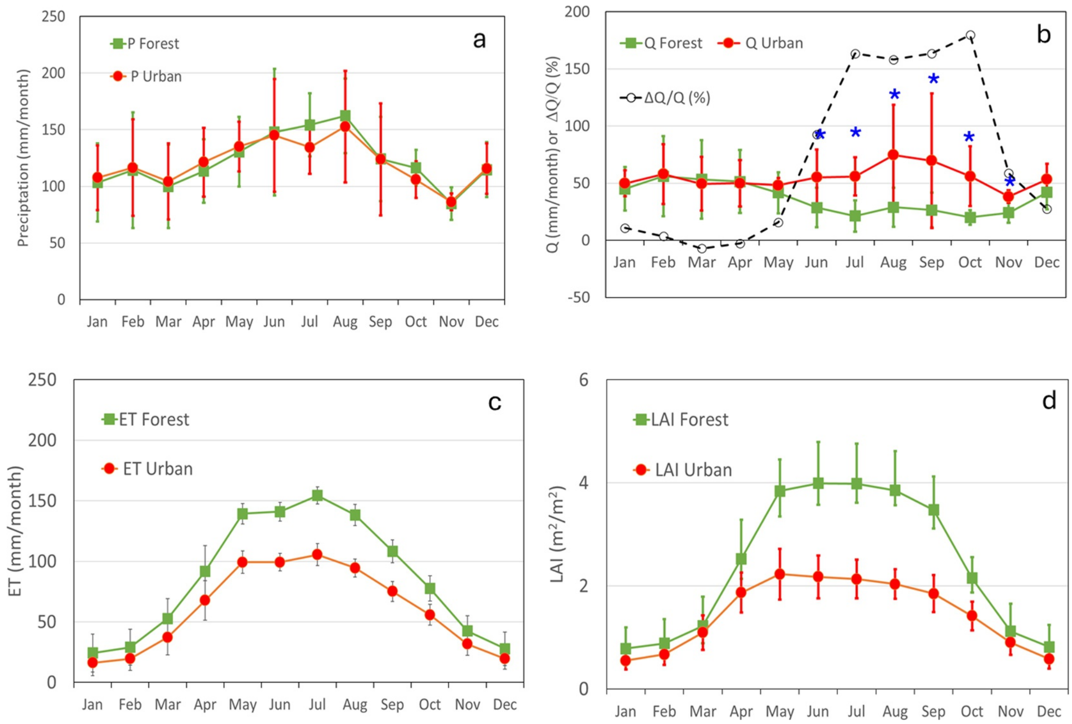

Figure 3. A comparison of monthly water balances and leaf area index (LAI) between forest (FW) and urban (UW) watersheds. (a) Precipitation, (b) Streamflow and percent differences between FW and UW. The symbol * indicates significant differences at a $p < 0.05$ level, and, (c) Estimated ET by SSEBop, and (d) Leaf Area Index (LAI) estimated using MODIS remote sensing data.

(Figure 2d). Q/P values for FW follow the Budyko curve (Zhang et al., 2001 model representing forests or grass land cover) more closely than UW which has larger variations in Q and Q/P.

3.1.2. Monthly Streamflow, Evapotranspiration (ET), and Leaf Area Index (LAI)

According to the Köppen-Geiger climate classification system, the southern region in the eastern U.S. represents a temperate climate (humid subtropical, Cfa) and northern region is dominated by a continental climate (cold winter, Dfb; Kottek et al., 2006). Both regions have hotter summers with higher temperatures (PET) and precipitation (P) than in winter months (Figure 3a). Convective rainfall and Atlantic hurricane dominate the precipitation in the growing season (roughly May–October). Snowpack is not uncommon in areas north of New York, controlling on spring high flows.

Overall, statistical analysis (*t* test) suggested that there were no significant differences ($p > 0.05$) in P between UW and FW (Figure 3a) indicting the paired watershed approach was validated in terms of similar climate. However, the *t*-test on monthly streamflow (Q) showed that there were significant differences in Q between UW and FW for June–November. Despite the higher precipitation in the peak growing season (roughly May–September), the higher PET and leaf area index (LAI; Figure 3d) resulted in a 54% lower total streamflow ($Q = 167 \pm 4$ mm) in this season than in the rest of the months ($Q = 272 \pm 11$ mm) in FW (Figure 3b). In contrast, the mean and standard deviation of UW streamflow were highest in August-September following a similar pattern to precipitation (Figure 3b). Consequently, the largest difference in streamflow (Q/P; 0.2 for FW vs. 0.5 for UW, or −60% lower in FW) were found in the peak growing season (July-September). It was noted that the differences in Q remained large (−55% lower in FW than UW) and were statistically significant during late fall and early dormant season (October and November; Figure 3b). The differences in Q and Q/P between FW and UW were much smaller, −6% and −10% respectively, during the later part of the dormant season (December–April) and late spring in May (Figure 3b). In other words, overall FW and UW behaved similarly in the late dormant season (December–May) but differed greatly in the peak growing season and early fall.

SSEBop ET estimates show large differences (306 mm or 42%) in annual total ET between FW (1,028 ± 119 mm) and UW (722 ± 99 mm). FW had a higher ET than UW year-round, especially during the peak growing seasons from May to August (Figure 3c; Mean monthly ET difference = 44 mm) with the largest monthly difference of 57 mm in Houston, TX (Pair 11), and Atlanta, GA (Pair 7). The monthly LAI dynamics for FW and UW follow a similar pattern as ET with the largest differences (0.6–2.7 m^2/m^2) in May-September and smaller differences in winter months from −0.1 to 0.9 m^2/m^2.

A closer examination of seasonal streamflow dynamics for individual sites revealed that streamflow of FW was consistently lower than UW during the growing season, but not in the dormant season. The contrasts between FW and UW varied across the latitude gradient from the northern cold climate in Boson, MA (Pair 1) to the warm subtropical Florida Panhandle (Pair 9). For example, the streamflow of FW for Pair 1 (Boston, MA; Figure S3 in Supporting Information S1) and Pair 2 (Lake Carmel, NY; Figure S4 in Supporting Information S1) was significantly higher than UW in the dormant season at these sites with cold winters and cool summers. In comparison, the streamflow rates for FW and UW were similar in the late dormant season in the two pairs with moderate air temperature in winters (e.g., Washington D.C., Figures S5 and S6 in Supporting Information S1; Raleigh, central NC, Atlanta, Figure S7 in Supporting Information S1). The largest streamflow differences occurred during the low flow periods (Figures S7a and S7b in Supporting Information S1) in the summer months. Groundwater recharge in FW began around October and soil water storage was filled up around end of January in both FW and UW. At the monthly and annual scales, more fluctuations in streamflow in UW indicated the higher sensitivity to precipitation. Unlike other watersheds in the northern states, the southern states (Pair 9, near Jacksonville, FL) had a long growing season with a warm winter and hot summer, and streamflow in FW that was dominated by evergreen pine forests remained lower than in UW in the winter months (Figure S8 in Supporting Information S1).

As a whole, the correlations analysis indicated that the differences of monthly streamflow (ΔQ) were significantly ($R^2 = 0.13$) correlated with the differences in monthly mean LAI (ΔLAI) ($p < 0.001$) and monthly P ($p < 0.05$; Figure S9 in Supporting Information S1). By individual pairs, the correlations between ΔQ and ΔLAI were significant ($p < 0.05$) for Pair 3, 4, 5, 9, and 10 ($R^2 = 0.33$–0.59). In contrast, monthly P explained most the variance of ΔQ with R^2 ranging from 0.35 to 0.93 for Pair 2, 4, 6, 7, 9,10, and 11 ($p < 0.05$; Figure S9 in Supporting Information S1).

3.1.3. Daily Flow

Daily streamflow Flow Duration Curves (FDC; Figure 4a) (Searcy, 1959) show that UW had higher flows than the FW for all flow frequencies, especially in the growing season (Figures 4b and 4c). The relative differences during the growing season were greater than 50% and occasionally over 250%. However, the contrast in daily flow between FW and UW was more complex in the dormant season (Figures 4b and 4d). UW had higher flow in the 1%–40% percentile but had lower flow in the 40%–100% percentile (Figure 4d), suggesting FW had higher flow in the dormant season. It is evident that FW had lower baseflow and total flow in the growing season (Figures 4b and 4c).

3.1.4. Hurricane Storm Events

We identified a total of 30 large hydrologic events due to Hurricanes or Tropical Storms during 2016–2021 (Table S2 in Supporting Information S1). These storms occurred from late May through early November each year and lasted 1–5 days and with total rainfall ranging from 20 to 887 mm (mean = 95 mm; Figure 5a). Stream flows in both UW and FW increased nonlinearly as a power function with the increase in the size of rainfall events. The hydrologic differences (ΔQ) between UW and FW linearly correlated with precipitation ($R^2 = 0.81$), becoming larger under bigger storms in general. However, outliers existed, as shown in the following examples. Under a relatively dry antecedent soil water condition, Hurricane Irma struck Florida on 9 September 2017. Both FW and UW (Pair 9, Jacksonville, FL) had a very small response: P = 287 mm, Q = 7.5 mm, Q/P = 0.03 for FW, and P = 272 mm, Q = 47, Q/P = 0.17 for UW, respectively (Figure 5a). In contrast, under extreme wet conditions, a moderate storm event, Tropical Storm Fred, impacted Pair 7 (Atlanta, GA) on 17 August 2021, causing widespread flooding (Pasch, 2022). The total precipitation was moderate (68 mm for UW, 86 mm for FW), but Q/P was found to be rather large, 1.05 for the UW and 0.28 for FW (Figure 5b). It seems that Q/P for FW was more variable than UW, especially for smaller storm events (P < 80 mm/day, Q < 5 mm/day).

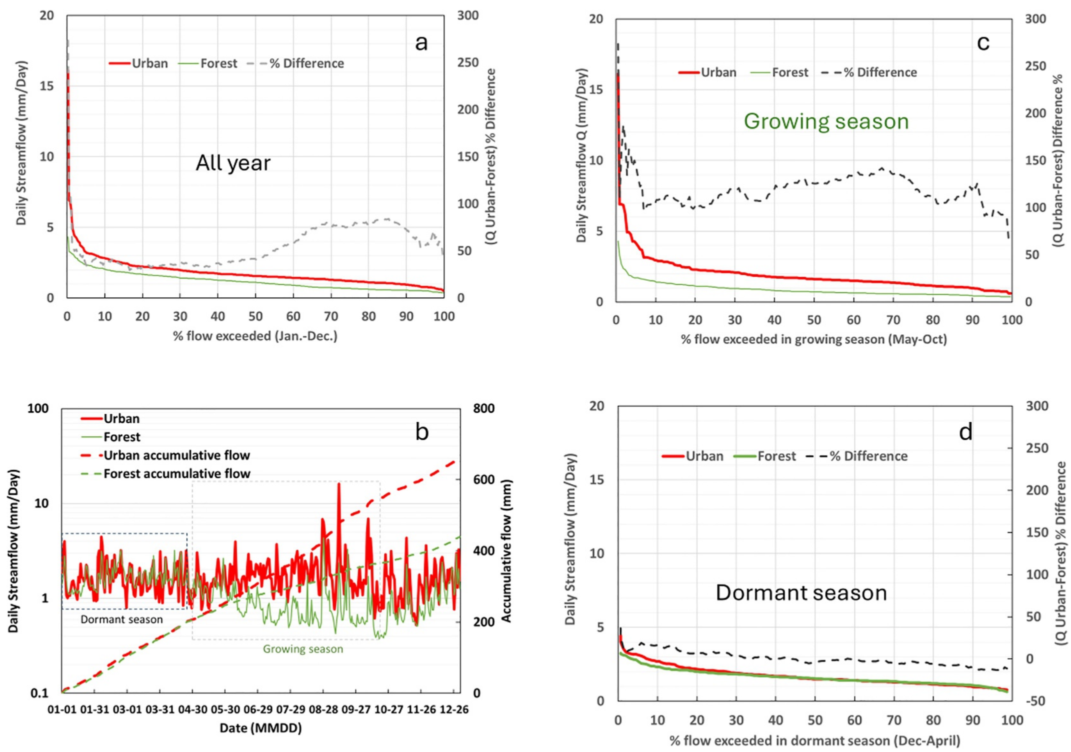

Figure 4. A comparison of average daily stream flow across 7-paired watersheds (2016–2022; Pair 1, 5, 6, 8 not included for consistency in data time periods). (a) Flow Duration Curves (FDC) to show flow frequency distribution, (b) composite mean daily flow patterns, (c) FDC for the growing season (May–September), and (d) FDC for the dormant season (October–April).

Overall, the UW had a significantly higher mean Q/P ratio (0.40 ± 0.2) than FW (0.10 ± 0.1; $p < 0.01$; Figure 5b). The individual storm peak flow ratios (Q/P) for UW were several hundreds to thousands of times higher than those of FW. For example, Hurricane Issaias with an 80 miles (129 km) per hour wind passed the region where Pair 4 (Raleigh, NC) is located, dumping a total rainfall of 87 mm for UW and 60 mm for FW during the peak growing season (August 4–5, 2020). This event resulted in a Q/P ratio of 0.36 in UW which was 15-fold higher than FW (Q/P = 0.02).

FW could absorb much more rainfall than UW during the growing season when the ET and available soil storage was generally higher in FW than UW. This can be best illustrated by the hydrologic response to Hurricane Danny in 1997 and Tropical Storm Alberto in 2006 (Figure 6). For the 1997 event, the UW of Pair 5 (Chapel Hill, NC)

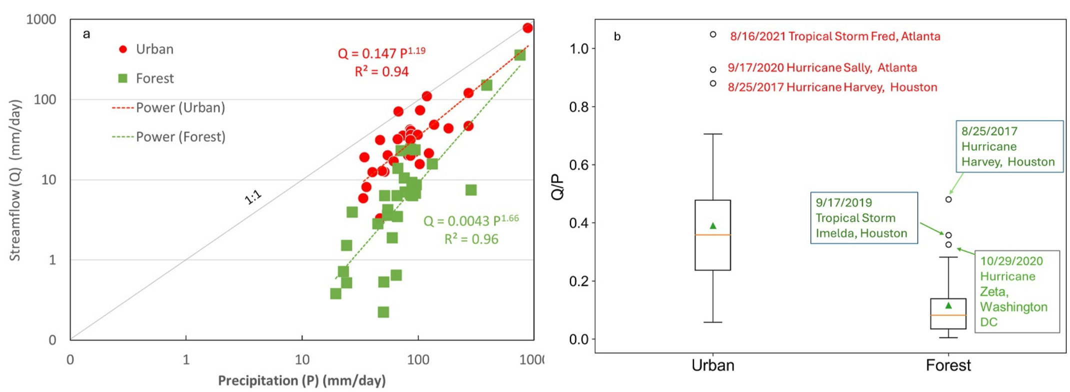

Figure 5. A comparison of streamflow response to 30 large storm events (2016–2021) in eight selected small watershed pairs (Urban vs. Forest) in the eastern U.S. (a) Correlations between event rainfall (P) and stream flow (Q), and (b) Box and Whisker diagrams of Streamflow/Precipitation (Q/P).

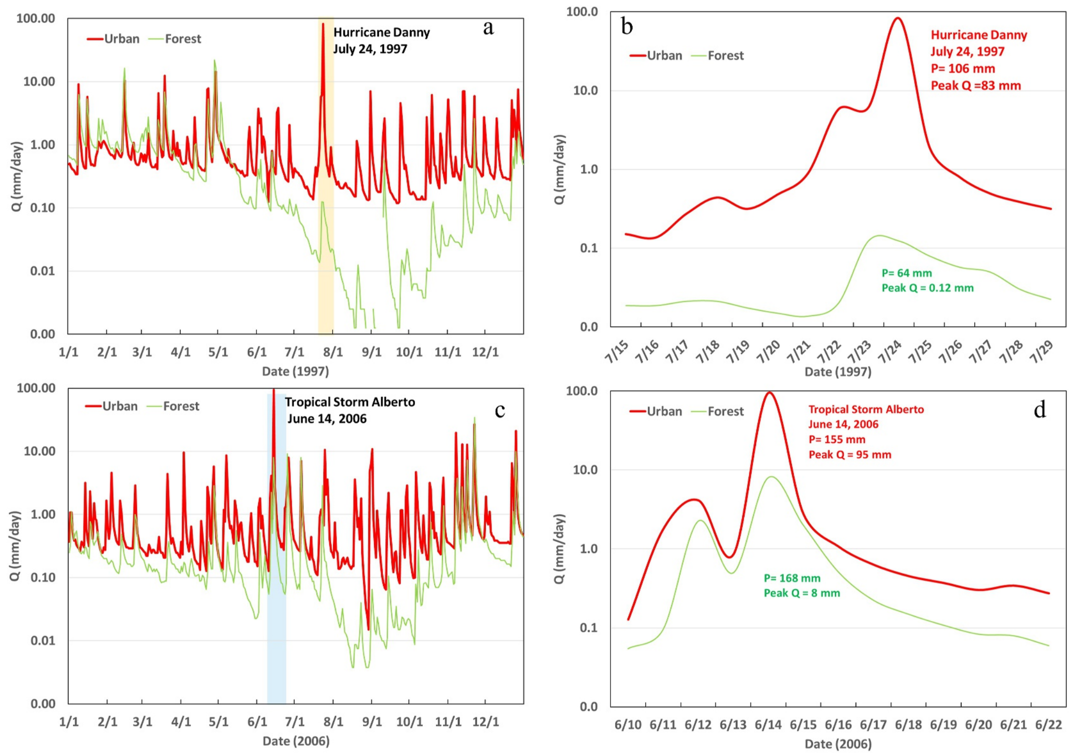

Figure 6. Two examples to compare hydrologic responses to large rainfall events between a forest (FW) and urban watershed (UW; Pair 5) in the Raleigh-Chapel Hill, NC piedmont region. (a) daily hydrograph in 1997, (b) a relatively low initial soil water condition, Hurricane Danny (24 July 1997), (c) daily hydrography in summer of year 2006, and (d) a relatively wet initial soil condition, Tropical storm Alberto (14 June 2006). The shaded areas represent hydrographs for large storm events with information detailed in panels (b, d), respectively.

received more than 106 mm precipitation and peakflow increased from 0.9 mm/day to approximately 83 mm/day (a 93% rise). While reported precipitation was lower ($P = 63$ mm) at FW, the response was rather small in terms of absolute change (peakflow Q increased from 0.02 mm/day to 0.12 mm, or 500% in a relative term). In this case, the watersheds were dry as indicated by the low streamflow rates prior to the hurricane event. In contrast, prior to the arrival of Tropical Storm Alberto hitting the Raleigh areas (Pair 5) with precipitation (168 mm for FW and 155 mm for UW), the watershed was likely wet (Q = 0.5 mm/day for FW and 0.85 mm/day mm for UW) due to previous rainfall. Consequently, FW had a large response reaching 8 mm/day while UW had an extremely high flow of 95 mm/day. Streamflow in both watersheds declined quickly when the hurricane passed and returned to pre-hurricane baseflow conditions withing 4 days (Figures 6b and 6c) in the warm summer with high ET.

3.2. Attribution Analysis

3.2.1. Mean Annual ET Differences Between Urban and Forest Watersheds

The streamflow comparison results above showed that ET played a critical role in explaining hydrological responses at multiple scales. Therefore, we explored the factors that controlled the annual mean ET differences between UW and FW (ΔET = UW ET – FW ET) across the 11 pairs that had variable climate background and magnitude of urbanization as indicated by ISA.

The mean annual ΔET significantly correlated with the ratio of urban area fraction (in %) in UW and FW (denoted as UAr; $r = -0.76$, $p < 0.01$), the ratio of impervious area between UW and FW (ISAr; $r = -0.64$, $p = 0.035$; Figure 7a), and the Dryness Index (DI=PET/P) of the UW ($r = -0.79$, $p < 0.01$) or FW ($r = -0.67$, $p < 0.03$; Figure 7b). The Pair 9 (Jacksonville, FL) appeared to be an outlier. The sites are characteristic of the flat lower coastal plains dominated by wetlands and sandy soils exhibited a rather large ET difference (451 mm/year) between FW and UW. When excluding this Florida site, the correlation between ΔET with DI would be much improved ($r = -0.78$) while the correlation with UAr and urban ISAr increased to -0.85, and $r = -0.91$,

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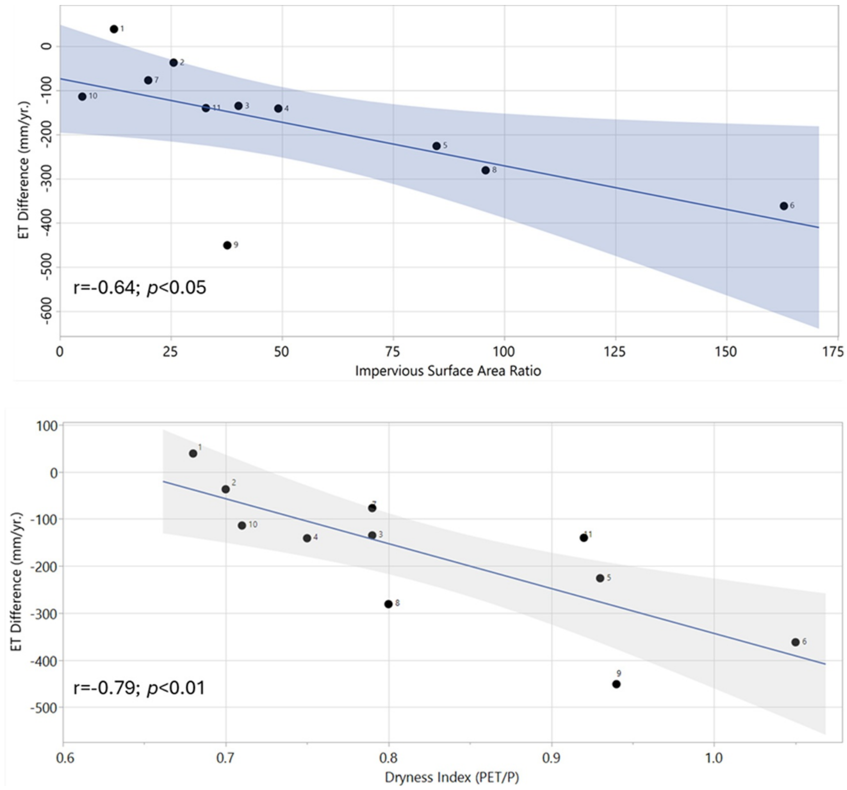

Figure 7. Differences in mean annual evapotranspiration (ΔET) between Urban Watersheds (UW) and Forest Watersheds (FW) significantly correlate with: (a) Impervious Surface Area Ratio, ISAr (ISA of UW/ISA of FW), and (b) background climate as represented by atmospheric Dryness Index (DI=Potential ET/Precipitation) calculated by the urban watershed climate data. The data points are labeled by watershed pair ID. Shade areas represent 95% confidence intervals of the regression line.

respectively. Upon developing a multivariate predictive model (Equation 1; $R^2 = 0.76$, RMSE = 81 mm, $n = 11$, $p < 0.01$) combining both UAr and DI (for FW), 76% of the variance of ΔET was explained by surface conditions (UAr) and background climate represented by atmospheric Dryness Index (DI) for the UW:

$$\Delta \text{ET} = 447 - 609.37 * \text{DI} - 13.37 * \text{UAr} \quad (1)$$

3.2.2. Peak Flow Differences Between Urban and Forest Watersheds in Response to Large Storms

There were significant ($p < 0.03$) positive correlations between the differences in peakflow runoff coefficient (ΔQ/P) and the differences of watershed impervious surface areas (ΔISA%) expressed as the ratio of ISA in UW and that in FW ($r = 0.43$; Figure 8a). In addition, ΔQ/P has a significant positive correlation ($r = 0.37$, $p < 0.05$) with Antecedent Baseflow (i.e., flow rate at the start of the rising limb) in FW (ABF; Figure 8b), but a similar correlation was not observed in UW. These results suggest both watershed imperviousness, dryness of the natural watershed as indicated by ABF, and storm size matter to the hydrologic responses in the eastern U.S.

4. Discussion

The direct comparisons between daily, monthly, and annual streamflow for headwater watersheds with different disturbances (i.e., urbanized vs. natural conditions) offered new insights into how vegetation and soils together regulate precipitation partitioning in the humid eastern U.S. We emphasize the role of ET and ISA in explaining the differences in flow patterns observed in a regional study that covers a large physiographic gradient in humid "energy-limited" region.

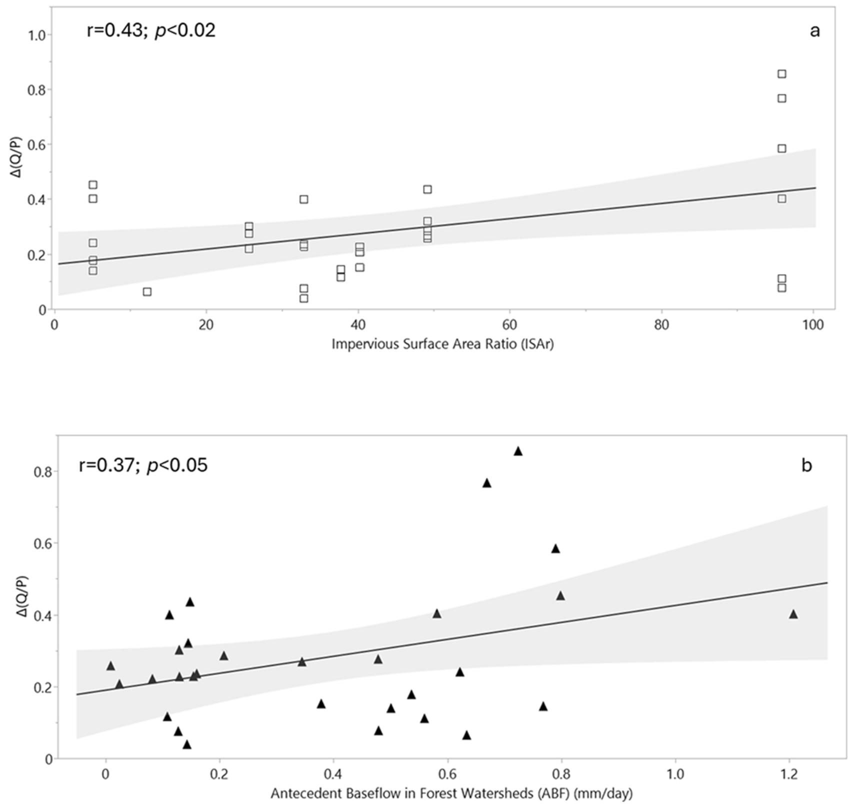

Figure 8. Correlations between differences in watershed characteristics and peakflow responses to 30 large storm events in seven watershed pairs in the eastern U.S. (a) Differences in rainfall-runoff coefficients (Q/P) versus Impervious Surface Area Ratio, ISAr (ISA of UW/ISA of FW), and (b) ΔQ/P versus antecedent soil water storage conditions as represented by baseflow in the forest watershed (FW) immediately prior to the peakflow Q. Shade areas represent 95% confidence intervals of the regression line.

4.1. Background Climate and Impervious Surface Area (ISA) Characterize Urban Watershed Hydrology

Our study showed that watersheds dominated by built-up landcovers (UW; ISA = 40–60%, growing season LAI = 1.4–2.6) had much higher annual water yield (Mean = 52%, ranging from −13% to 141%) and lower total ET (17% less) than forest dominated watersheds (FW; ISA = 1–8%, growing season LAI = 3.0–4.8; Figures 2a and 2d). The magnitude of urbanization affecting streamflow was also influenced by background climate: highest difference was found in relatively drier sites (Equation 1). Our findings are consistent with a remote sensing-based study on 210 cities in China (Wu et al., 2025). They suggest that green spaces have higher water-retention capacity in relatively arid regions but have more pronounced cooling effects in humid regions, with both effects being largely season dependent.

Unlike FW, UW runoff coefficients (Q/P) do not follow the general Budyko curve designed for natural ecosystems (Figure 2d), suggesting that factors other than climate or/and vegetation may have more important roles in regulating streamflow in UW. Soil compaction, impervious surfaces (Oswald et al., 2023), lawn irrigation, artificial impoundment (e.g., stormwater ponds), rain gardens, and drainage (e.g., leaking pipes) are all likely to contribute to the high peakflow and baseflow in urban watersheds (Darrel Jenerette & Alstad, 2010).

The relative differences in annual Q between UW and FW (ΔQ/Q = 51% ± 46%) found in this study were generally higher than reported for the first-year hydrologic response to complete deforestation in experimental studies in the eastern U.S. Reported values for the mountainous watersheds at the Coweeta Hydrological Laboratory range from 20% to 40% in western NC (Douglass, 1983), from 10% to 28% for Fernow Experimental Forest in West Virginia, and about 45% for Hubbard Brook Experimental Forest in the White Mountains of New Hampshire. The three pairs (4, 5, and 6) in the Piedmont region in NC with the highest dryness index (0.75–1.0; Figure 2 and Table 1) showed the highest response (ΔQ/Q = 60%–140%, ΔQ = 210–250 mm). This is consistent with the notion that the piedmont region is particularly sensitive to forest removal in comparison to the mountains or the coastal plain geophysical regions due to the relatively high PET but low precipitation in the "dry" hilly Piedmont (Sun et al., 2002). Clearcutting two small watersheds covered by mixed forests in Durham, NC in a "paired watershed" experimental study resulted in an increase in annual water yield of 160 mm or 130% (Boggs et al., 2016). The higher response in urban watersheds than experimental deforestation is understandable because forest removal experiments generally have little damage to soil physical properties and infiltration capacity. The UW examined in this study had rather high ISA (>40%), well above the 10%–20% threshold that serve as indicator of watershed ecosystem degradation (Arnold & Gibbons, 1996; Oudin et al., 2018). Therefore, our study results represent the likely extremes of hydrologic impacts of urbanization due to the combinations of soil compaction and reduction of ET.

The UW ET rates, 585–972 mm per year or 57% of P in this study were comparable to reported values of 360–800 mm in the southeastern U.S. (Diem et al., 2023). However, our estimates were much higher than previous conceptual illustration by Livingston and McCarron (1992). Unfortunately, the ET values (40% of precitation for natural systems) in Livingston and McCarron (1992) have been widely cited in literature (Arnold & Gibbons, 1996; Paul & Meyer, 2001) and used by state and federal regulation agencies (EPA, 2003) as a guide for stormwater management. A detailed ET modeling study in the Atlanta metropolitan areas suggested that there was a large uncertainty in urban ET estimation: open-space developed land (e.g., grass, forests; ISA < 8%) had an ET total of 920 mm per year that was over four times that of high-intensity developed land (ET = 241 mm per year) with high ISA (>90%; Diem et al., 2023). Our attribution analysis also indicated that both local climate (Dryness Index, PET/P) and land cover and development status (ISA) were the two major factors controlling watershed ET at the regional scale (Equation 1).

4.2. Seasonal Shift of the Dominant Role of Forests Serving as a "Sponge" and "Pump"

Forests are known to serve as large "sponge" (Bruijnzeel, 2004) to absorb rainfall due to porous soils and massive root systems underground, resulting in little overland flow and in many cases promoting groundwater recharge (Bruijnzeel et al., 2025). The ideas of building a "Sponge City" in modern urban planning emphasize the desire of green infrastructure to enhance infiltration by the ground surface and to enhance its ability to store stormwater (Zha et al., 2021). Forests are also known to have higher latent heat flux (i.e., ET) than other landcovers due to their low albedo (thus high net radiation), high leaf biomass and aerodynamic roughness, and deep rooting systems (Aguilos et al., 2024). Thus, forests are often regarded as biological "pumps" and ET is considered as "biological drainage" (Heuperman, 1999) that returns recharged soil water back to the atmosphere to complete the water cycle.

Our comparison study included multiple watersheds across a climatic gradient with various land cover types. The comprehensive analysis at the seasonal and daily scales demonstrated that forest watersheds function as a more powerful "pump" and "sponge" than urban systems. However, the dominance of these two functions could vary seasonally. For example, the differences in streamflow (Q; Figure 3b) and ET (Figure 3c) between FW and UW were most pronounced during the growing season. In contrast, the Q differences diminished during dormant seasons when the ET "pump" was shut down (Figure 4b) and the soil "sponge" was wet and fully filled with water likely to the field capacity.

Because of lower vegetation covers, higher soil compaction and ISA coverage that resulted in lower ET and limited available water storage (small "sponge"), Q in UW was much higher than FW in the growing season coincident with high P characteristic of the eastern U.S. region. However, Q in FW was the lowest during the growing season when the forest soils were dry with a large available water storage (large, dry "sponge") presumably due to porous soils and vegetation covers with higher LAI that favored higher ET than UW.

Although monthly Q of FW was always lower than in UW in the growing season, the contrasts in Q varied across the study gradient (Figures S3–S8 in Supporting Information S1). For example, the monthly Q was higher in FW than UW in the dormant season (Figures S3–S4 in Supporting Information S1) in the cold region (Pair 1, 2 in MA and NY) where ET differences between FW and UW were rather small (7 mm per month). This was not the case in Pair 3 and 6 in Washington DC and Raleigh, NC (Figures S5–S7 in Supporting Information S1) where Q was similar between FW and UW. These watersheds had moderate winter temperatures (PET) and the ET differences were higher (14 mm per month) than Pair 1 and Pair 2. There were reasons to believe that soil water storage reservoirs for these watersheds (Pair 1–6) were recharged to their full capacity over the fall and during the dormant season. This is a common assumption in the southern Appalachians to facilitate estimating annual ET using the water balance approach when soil is recharged to its maximum often by April or May and thus change in storage is negligible (Hwang et al., 2018). Because Q was more controlled by soil moisture in the winter when precipitation was less frequent than the growing season, FW had higher Q due to its higher soil water storage capacity (i.e., bigger "sponge") than UW. In this situation, the forest "pump" function was either diminished (Pair 1 and 2) or balanced by the forest "sponge" ability to control Q (Pair 3 and 6). Under a hotter climate (PET $>$ 1380 mm) with mild winters and a much longer growing season, such as the case of Pair 9 (Jacksonville, FL), Q from UW remained higher than FW year around and the ET differences in the dormant season (October–April) defined by this study was 14 mm (Figure S8 in Supporting Information S1). In this case, like the behaviors of other forest watersheds in the growing season, the forest "pump" function dominated the "sponge" role year-round.

4.3. Streamflow Response to Extreme Rainfall Events

There were large differences between FW and UW in hydrological responses to large rainfall events from hurricanes and tropical storms (Figures 5 and 6). Attribution analysis showed that ISA and antecedent watershed hydrological conditions were partially responsible for these differences (Figures 7 and 8). We observed that forest watersheds absorbed a large amount of rainwater (refer to example in Figures 6b and 7) in the growing season when the soils were generally dry due to high ET despite the higher precipitation than in the dormant season. In the growing season, FW served as a large dry "sponge." In contrast, the urban watersheds were relatively wet in the growing season (Figure 4b) due to frequent rainfall despite the higher ET than in the dormant season. In this case, UW represents a wet "sponge" with limited soil water storage. Therefore, streamflow in UW responded quickly to hurricanes, most likely due to large ISA and small soil water storage. In some cases, Q/P could reach unity (Figure 5b).

Despite the existing debate on the role of forests in abating large floods (Alila et al., 2009; Hewlett & Doss, 1984), the effects of reduction of ET from forest removals on peak flows of small watersheds have been well documented in forest hydrology literature (Bathurst, Fahey, Iroum, et al., 2020; Hewlett et al., 1977). Traditional small "Paired Watershed Experiments" in the southeastern Appalachian Mountains in western North Carlina have shown that tree removals with minimum soil disturbances could elevate peakflow by over 7% and quick flow (stormflow) volume by 11% on average. The maximum increase in peakflow reported was 48 mm, or 22% for a 7-day record flood (Hewlett & Helvey, 1970). The relative impacts of forest removal on peakflows were more pronounced in the drier and hotter Piedmont watersheds in North Carolina. Clearcutting a mixed hardwood-pine forest resulted in a much bigger effect under extreme storms (Boggs et al., 2016). In this case, high flow volumes (>95% flow exceedance) were elevated by 45% on average. The three observed largest peaks (9–11 mm/day) represented an increase in peakflow of 4–8 mm/day or 58%–600% during the post-harvest periods. These studies indicate that evapotranspiration from forests and the consequent soil water deficits under forests can substantially affect the production of floodwaters (Hewlett & Helvey, 1970). Global studies on forest-peakflow relations concluded that the effects of forests on flood peaks depend on the size of the storm (return period), antecedent soil water conditions, and land disturbances such as road network (Bruijnzeel, 2004) and drainage ditch density (Bathurst, Fahey, Iroum, et al., 2020). Urbanized watersheds examined in this study had ISA over 40% and tree covers less than 44% (Table 1), representing high-intensity developed areas. The soil disturbances and flow path alterations in these urban watersheds far exceeded those represented in the "paired watershed" experiments in traditional forest hydrology research where soils are less affected. However, urban watersheds are not fully covered by ISA and a large percentage area such as productive lawns, wetlands, storm ponds that can have high ET and function as effective "pump" in the eastern "water rich" U.S. Urban areas generally have higher PET than surrounding rural areas (i.e., FW) due to "Urban Heat Island" (Zhou et al., 2016) and "Urban Dry Islands" Effects (Hao et al., 2018, 2023), which somewhat mitigates the loss of "pump" functions in cities because actual ET depends on energy availability, especially in energy limited regions (Aguilos et al., 2024; Sun et al., 2011).

4.4. Implications to Urban Watershed Management

The present study documented large differences in streamflow between forest-dominated watersheds and urbanized watersheds in the humid eastern U.S. Vegetation cover characteristics (i.e., amount and types) and the magnitude (i.e., percentage of a watershed) of impervious surface explained most of the hydrologic contrast at multiple temporal scales. Using these data sets and knowledge gained from this study, a conceptual model was developed to illustrate the likely differential hydrologic impacts of urbanization across a climatic gradient

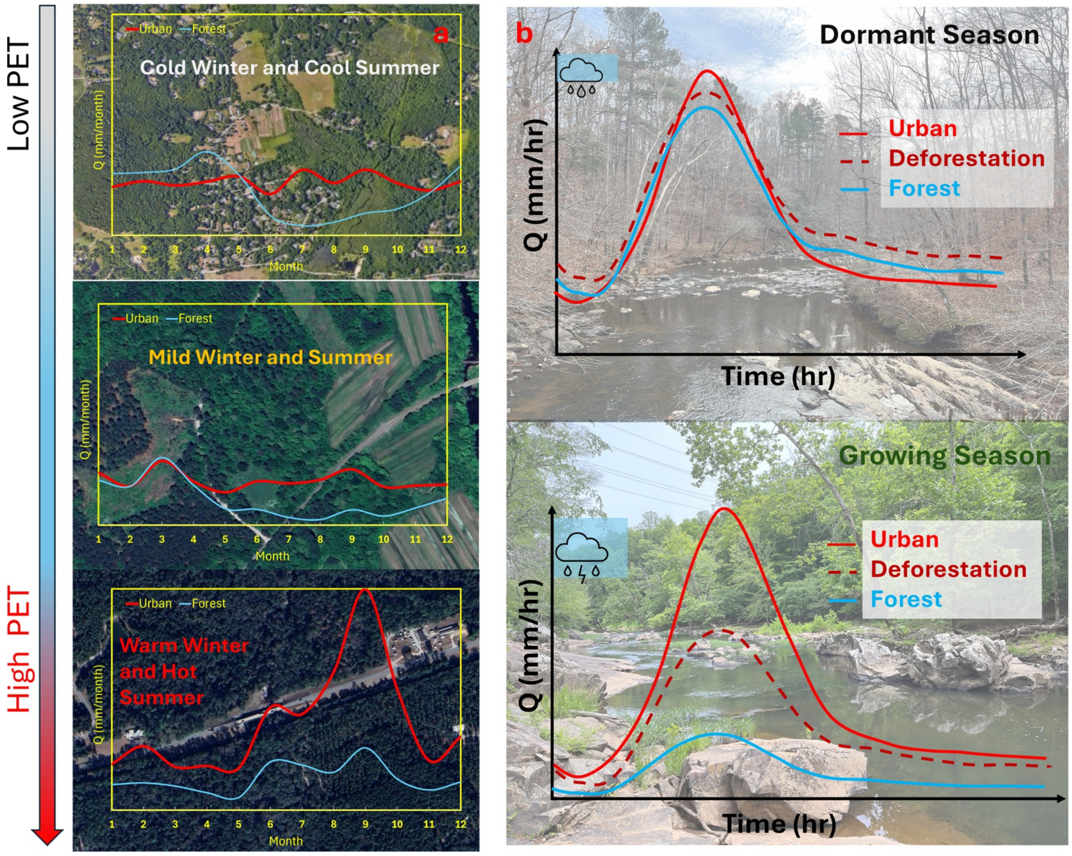

Figure 9. Conceptual models illustrating differences in watershed hydrographs at two temporal scales. Panel (a) seasonal scale across an energy availability (potential evapotranspiration, PET) and vegetation phenology gradient. Background satellite land cover images (Google, 2025) represent forested watersheds dominated by deciduous forests in Pair 1 in Massachusetts, mixed forests in Pair 4 in central North Carolina, and evergreen pine forests in Pair 9 in northern Florida. Panel (b) individual storm events under growing and dormant seasons. The deforestation scenario represents forest removal with limited soil compaction and degradation.

(Figure 9a) and the effects of deforestation alone (tree removals with little damage to soils) and urbanization dominated by ISA on stream hydrographs in different seasons (Figure 9b). We show that both ET dynamics and ISA play a critical role in influencing the large differences in hydrologic patterns among the three land cover types at monthly (Figure 9a) and sub-daily scale (Figure 9b). These findings have important implications for urban planning, flood mitigation, and land management at a watershed level.

Overall, our findings indicated the widely used urban water balance conceptual model (Livingston & McCarron, 1992) and adopted in practice for stormwater mitigation may need to be updated because its reported ET rate (<40% of P) for natural ecosystems is considered to be too low under most situations in the U.S. We hypothesize that when the model is applied for designing urban green infrastructure the role of vegetation in mitigating storm runoff might be greatly underestimated. We also found that forest-dominated watersheds had much lower hydrologic response to large runoff events brought by tropical storms or hurricanes. Thus, the common perception that forests can only mitigate small or moderate floods (Bathurst, Fahey, Iroumé, et al., 2020) might not be accurate, at least in the eastern U.S. The tremendous ET "pump" function cannot be overlooked in urban planning which generally has focused on reducing ISA in the past. Maintaining vegetation ET, the "biological drainage," that is, responsible for the "dry sponge," is important in storing urban stormflow during the growing season (Hao et al., 2015). Urban watershed management that promotes green infrastructure and low impact development practices should minimize ISA (Fletcher et al., 2013) and maximize ET (Ekness & Randhir, 2015). Maintaining forest canopy covers and wetlands, natural or managed, increases available storm water storage and ET rates, thus reducing frequent flooding risk (Cao et al., 2022) and mitigating "Urban Heat Island" and "Urban Dry Islands" through cooling and humidifying (Hao et al., 2018), and non-point source water pollution (Sun & Lockaby, 2012).

4.5. Study Limitations and Future Research

We have assembled a set of long-term hydrometeorological data for 22 watersheds that provided a direct comparison of hydrologic behaviors at daily, monthly, and long-term scales. The analysis of 30 extreme events offered new insights into the role of soil imperviousness and ET in controlling how watersheds store and release water during hurricanes. Despite the added capacities of this straight-forward comparison approach, our paired watershed approach had inherent uncertainties compared to a standard "Paired Watershed Experiment" that follows a Before-After Control-Impact (BACI) study design (Ssegane et al., 2013); for example, bias may exist in detecting hydrologic differences without watershed calibrations (Nigro et al., 2024). Although we have made every effort to ensure each paired watershed had similar climate and rainfall frequency distribution at the daily and monthly scales in particular, rainfall intensity at sub-daily could be different within each paired watershed. The spatial heterogeneity of rainfall could be large during hurricanes or tropical storms. In addition, the streamflow data records were limited to certain time periods across the study region (Table 1). To remedy these issues, we have used the mean normalized streamflow with precipitation (i.e., Q/P) in our analysis. In addition, we examined ET differences at monthly and annual scales to remove the likely impacts of bias of precipitation on Q. The ET differences between adjacent watersheds represent land cover change impacts on streamflow in a humid region (Aguilos et al., 2024) when energy and water availability (PET) is similar.

In this study, we assumed that the background soils and geology are similar between each selected watershed pair. This assumption might not be accurate in some cases when the paired watersheds are not in the same geological regions. Soil texture and structure are known to be an important factor in controlling streamflow response to watershed disturbances (Bergeson et al., 2022; Boggs et al., 2013). However, the impacts of urban ISA, underground stormflow drainage network, and overland flow connectivity on peak flow timing and magnitude likely overwhelm the differences of soil hydraulic properties especially for small watersheds ($<10\ km^2$) with sandy soils (Cronshey, 1986).

Limited by the nature of stream gages, the sizes of FW are much larger than UW in our study in most of the 11 pairs selected (Table 1). Watershed size has the potential to affect streamflow patterns. Larger watersheds are known to have more baseflow than smaller ones ($<10\ km^2$), but smaller watersheds can respond to stormflow quickly with short travel time (time of concentration; Cronshey, 1986), resulting in higher peakflow and higher streamflow rates (Seethapathi et al., 2008). However, watershed size concerns are minor. For example, against the general notion that larger watersheds have higher baseflow, we found that the larger FW had much lower baseflow than the smaller UW in the growing season. Similarly, we found that peakflow rates for the FW were consistently lower than the UW regardless of watershed size differences, especially in the summer season (Figures 3 and 4).

The hydrology of an urban watershed is deemed more complex than its natural counterpart (Cronshey, 1986; Seethapathi et al., 2008) when the urban environment is dominated by ISA, mixed vegetation covers, and human activities (i.e., lawn irrigation, inter-basin transfer) that fundamentally altered the land energy and water balances (Jin et al., 2023) and flow pathways (i.e., stormwater drainage network for quick discharge; Voter & Loheide, 2018). Future studies are needed to better quantify how urbanization-induced microclimate and macroclimate change has affected water loss as ET in a buildup setting. The global eddyflux measurement network provides a platform for quantifying energy, carbon, and water balances. In collaboration with urban experimental watershed monitoring (e.g., Urban Longterm Ecological Research), such flux measurements will provide insight into hydrological responses to land cover change and urban development (Kellner & Hubbart, 2017). Equally important are that studies on how soil hydraulic properties have been impacted by urbanization and how altered flow pathways, both surface and subsurface routes (Voter & Loheide, 2018), affected water balances at multiple spatial and temporal scales (Bergeson et al., 2022).

5. Conclusions

Using a paired watershed research approach, we compared streamflow patterns and water balances in 11 natural watersheds dominated by forests (FW) and 11 adjacent urban watersheds (UW) dominated by urban land use and impervious surfaces areas (ISA) across the eastern U.S. We found that urban watersheds (UW) had a much higher annual total flow. UW had higher peak flow and higher low flow (dry season flow) mostly during the summer growing season than FW, but not in the dormant season. Forest evapotranspiration and soil water storage capacity dominated monthly flow patterns in the growing season and the dormant season, respectively. The 30-large storm events data demonstrated that forest watersheds could substantially mitigate extreme storms such as hurricanes in comparison to urban watersheds. We concluded that both vegetation water use (ET), ISA, and background climate (PET) were key factors that explained the hydrologic differences between the two types of watersheds.

Our study supported the hypothesis that, in addition to ISA, vegetation cover is an important driver for the evolution of urban hydrology through changes in ET. Forest watersheds are large dry "sponges" because forests have high soil water storage and high ET rates that create large storage available to store rainwater during storm events. Therefore, maintaining watershed ET capacity or "biological drainage" through conserving existing trees, forests and wetlands or developing other "green infrastructure" is critical for minimizing flooding vulnerability in urban watersheds. This study supports the idea of "Keeping forestlands as forests" (Caldwell et al., 2023) that help maintain watershed retention and ET functions that benefit urban inhibitors by reducing water pollution, flood risk, "Urban Heat Islands," and "Urban Dry Islands."

Conflict of Interest

The authors declare no conflicts of interest relevant to this study.

Data Availability Statement

Key data sets used for this study were obtained from several published sources:

Daily precipitation data are available at: Thornton, M. M., Shrestha, R., Wei, Y., Thornton, P. E., & Kao, S.-C. (2022). *Daymet: Daily Surface Weather Data on a* 1-km *Grid for North America, Version 4 R1* (Version 4.1). [Dataset] ORNL Distributed Active Archive Center. https://doi.org/10.3334/ORNLDAAC/2129 Date Accessed: [2024-10-17] https://daymet.ornl.gov/.

Streamflow data are available free of charge at: U.S. Geological Survey. (2024). National Water Information System data. [Dataset] USGS Water Data for the Nation, https://waterdata.usgs.gov, accessed [10 June 2024] http://dx.doi.org/10.5066/F7P55KJN.

Land use and land cover, and impervious surface area data are available at: Dewitz, J., and U.S. Geological Survey, (2021). National Land Cover Database (NLCD) 2019 Products (ver. 3.0 February 2024): [Dataset] U.S. Geological Survey data release, https://doi.org/10.5066/P9KZCM54.

Potential ET data are accesses through: Senay, G. B., Parrish, G. E., Schauer, M., Friedrichs, M., Khand, K., Boiko, O., & Ji, L. (2023). Improving the operational simplified surface energy balance evapotranspiration model using the forcing and normalizing operation. Remote Sensing, 15(1), 260 [Dataset]. SSEBop V6 freely available at https://earlywarning.usgs.gov/.

Acknowledgments

The authors acknowledge financial support from the USDA Forest Service Southern Research Station Eastern Forest Environmental Threat Assessment Center. The authors also thank anonymous reviewers for their insightful comments during revision of this manuscript. Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. government.

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Erratum

The originally published version of this article contained an error in panel d of Figure 4. The error did not affect the conclusions of the study because the correct figure was used for the analysis. The error has been corrected, and this may be considered the authoritative version of record.