AGU PUBLICATIONS

Geophysical Research Letters

RESEARCH LETTER
10.1002/2016GL072190

Urban heat island-induced increases in evapotranspirative demand

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Key Points:

- Urban heat island (UHI) increases reference evapotranspiration (RET), with consistent response to impervious cover across years
- RET and vapor pressure deficit changes are explained by increased air temperature, with minimal impact of decreased air moisture content.
- Changes in within-season RET are more important than UHI-induced lengthening of growing season

Supporting Information:

- Supporting Information S1

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Citation:
Zipper, S. C., J. Schatz, C. J. Kucharik, and S. P. Loheide II (2017), Urban heat island-induced increases in evapotranspirative demand, *Geophys. Res. Lett.*, *44*, doi:10.1002/2016GL072190.

Received 3 DEC 2016
Accepted 18 JAN 2017
Accepted article online 20 JAN 2017

Abstract Although the importance of vegetation in mitigating the urban heat island (UHI) is known, the impacts of UHI-induced changes in micrometeorological conditions on vegetation are not well understood. Here we show that plant water requirements are significantly higher in urban areas compared to rural areas surrounding Madison, WI, driven by increased air temperature with minimal effects of decreased air moisture content. Local increases in impervious cover are strongly associated with increased evapotranspirative demand in a consistent manner across years, with most increases caused by elevated temperatures during the growing season rather than changes in changes in growing season length. Potential evapotranspiration is up to 10% higher due to the UHI, potentially mitigating changes to the water and energy balances caused by urbanization. Our results indicate that local-scale land cover decisions (increases in impervious cover) can significantly impact evapotranspirative demand, with likely implications for water and carbon cycling in urban ecosystems.

1. Introduction

As the proportion of the global population living in cities increases from 54% today to an estimated 66% by 2050 [*United Nations*, 2014], creating sustainable cities has emerged as a global challenge [*United Nations*, 2010; *Ramaswami et al.*, 2016]. Maintaining healthy urban ecosystems is critical to this goal; urban vegetation is associated with benefits including improved health [*Kardan et al.*, 2015; *Hartig and Kahn*, 2016] and reduced air and water pollution [*Nowak et al.*, 2014; *Denman et al.*, 2016]. This has led to widespread and large-scale investment in urban vegetation worldwide, epitomized by “million tree” initiatives in cities including Denver, Los Angeles, London (Canada), Miami, New York City, and Shanghai. However, the urban environment poses unique challenges for ecosystems, leading to a low survival rate of urban trees [*Sjoman and Nielsen*, 2010; *Koeser et al.*, 2013]. Among these, urban ecosystems typically experience elevated temperatures relative to their rural surrounding, a phenomenon known as the urban heat island (UHI) effect which has been documented worldwide [*Oke*, 1973, 1982, 1988; *Gazal et al.*, 2008; *Georgescu et al.*, 2013; *Zhou et al.*, 2015]. UHIs occur due to anthropogenic modification of the land surface. Key factors in temperate climates include the replacement of vegetation with impervious surfaces which typically increases the Bowen Ratio and heat storage while reducing albedo, reduced export of heat from the land surface to the atmosphere due to a decreased sky-view factor, and increased waste heat generation due to energy use [*Oke*, 1982, 1988; *Stewart and Oke*, 2012; *Georgescu et al.*, 2015].

While it is well known that urban vegetation can reduce the magnitude of the UHI [*Feyisa et al.*, 2014; *Declet-Barreto et al.*, 2016; *Sharma et al.*, 2016; *Shiflett et al.*, 2016; *Vaz Monteiro et al.*, 2016], little is known about the converse of this relationship: UHI impacts on evapotranspiration are complex and poorly understood [*Peters et al.*, 2011; *Wang et al.*, 2011; *Georgescu et al.*, 2012, 2013; *Ugolini et al.*, 2012]. Recent work has demonstrated that urban design decisions which reduce the magnitude of the UHI, such as cool roofs, may also lead to a reduction in evapotranspiration (ET) [*Georgescu et al.*, 2012] and that urban vegetation can adapt to reduced water supplies in urban areas [*Ugolini et al.*, 2012]. However, despite the global prevalence of the UHI and biophysical models capable of simulating the urban water cycle with ever-increasing detail [*Bhaskar et al.*, 2015; *Shields and Tague*, 2015], the drivers of variability in evapotranspiration within urban areas remain poorly constrained [*Peters et al.*, 2010].

Understanding the impacts of the UHI on evapotranspirative demand can guide the sustainable design of cities by improving estimates of urban ecosystem water requirements and stress regimes. As the global urban population and footprint continue to expand, research must focus on mechanistic understanding of the processes unique to urban ecosystems [*McDonnell and MacGregor-Fors*, 2016]. Previous research on vegetative response to the UHI has primarily focused on temperature-induced shifts in phenology [*Buyantuyev and Wu*, 2012; *Jochner and Menzel*, 2015; *Melaas et al.*, 2016; *Zipper et al.*, 2016], the cooling effects of urban green space [*Chang et al.*, 2007; *Declet-Barreto et al.*, 2013; *Feyisa et al.*, 2014; *Shiflett et al.*, 2016], and the water balance at a site or local scale [*Peters et al.*, 2010; *Vico et al.*, 2014; *Jacobs et al.*, 2015]. To date, neither the magnitude nor drivers of intraurban variability in evapotranspirative demand have been quantified. Here we address three fundamental questions regarding the UHI impacts on plant water requirements:

1. Does the UHI increase evapotranspirative demand, and through what mechanism?
2. How do changes in phenology and micrometeorological conditions interact to impact total growing season evapotranspirative demand?
3. Do UHI impacts on evapotranspirative demand counteract or reinforce urbanization-induced reductions in ET?

We hypothesize that the UHI increases vegetative water requirements through two mechanisms: increases in the growing season length (length-of-season changes) and increases in water demand during the growing season (within-season changes). Combined, these mechanisms are enabled by increased temperature and decreased air moisture content in urban areas relative to their rural surroundings. Additionally, we hypothesize that changes in potential ET rates of the vegetation remaining within UHIs could partially counteract decreases in total ET caused by urbanization.

2. Materials and Methods

We test these hypotheses using a network of 151 HOBO U23 v2 automated temperature/relative humidity sensors distributed in and around the city of Madison, WI (USA), over 4 years (2012–2015) (Figure S1 in the supporting information), described in detail in *Schatz and Kucharik* [2014]. As a midsized urban area (population 402,000) Madison has broad global relevance; worldwide, approximately half of urban residents live in urban areas of less than 500,000 people [*United Nations*, 2014]. Furthermore, Madison's UHI is well characterized by past studies, providing useful context for interpretation of results. Our sensor network is among the densest currently deployed [*Schatz and Kucharik*, 2014; *Smoliak et al.*, 2015; *Shiflett et al.*, 2016] and has been previously used to study the spatial patterns and drivers of Madison's UHI [*Schatz and Kucharik*, 2014], the impact of the UHI on extreme heat and cold [*Schatz and Kucharik*, 2015], and the UHI impacts on growing season heating/cooling degree days [*Schatz and Kucharik*, 2016] and plant phenology [*Zipper et al.*, 2016].

Sensors were installed beginning in March 2012 at a height of 3.5 m on light and utility poles and classified as rural, park, urban, wetland, and lake at the time of installation; in this study, we analyze only urban, park, and rural sensors. Sensors logged instantaneous temperature and relative humidity at 15 min intervals, which were aggregated to daily maximum and minimum temperature and used to calculate daily mean actual vapor pressure (e_a), saturation vapor pressure (e_s), and vapor pressure deficit (VPD).

Evapotranspirative demand was calculated as Penman-Monteith reference ET (RET; see abbreviation legend in Table 1) [*Allen et al.*, 1998]. RET is standardized to reference canopy conditions, and therefore does not reflect actual plant water use, but rather the atmospheric demand for water. Wind speed and incoming shortwave solar radiation were measured at the University of Wisconsin-Madison Arboretum, a large urban park (Figure S1). To calculate total growing season RET (RET_{GS}), we estimated the start and end of the growing season (SOS and EOS, respectively) using a degree-day approach modified from *Richardson et al.* [2006] and described in *Zipper et al.* [2016]. We defined the daily RET anomaly (RET_{Anom}) at each urban sensor as the difference between RET at that sensor and mean rural RET for that date, and the total growing season RET anomaly ($RET_{Anom:T}$) as the cumulative RET_{Anom} from SOS to EOS for each urban sensor. We subdivided $RET_{Anom:T}$ into contributions from changes in growing season length ($RET_{Anom:L}$) and changes in within-season meteorological conditions ($RET_{Anom:W}$) (Figure S3). To separate the impacts of air temperature and moisture content on RET, we separately calculated daily RET_{Anom} combining urban e_a and rural e_s (RET_{+V}) to isolate the impacts of changes in air moisture content in urban areas, and rural e_a with urban e_s (RET_{+T}) to isolate the changes in air temperature.

While RET represents a standardized atmospheric demand for water, potential ET (PET) is a function of atmospheric demand, vegetation type, and plant growth stage. To assess spatial dynamics in RET and PET, we interpolated daily RET over the City of Madison and surroundings using impervious cover, lake proximity, and topographic relief as covariates. We used a crop coefficient approach [*Allen et al.*, 1998] to estimate

Table 1. Abbreviation Legend, in Order Introduced in Text

Abbreviation	Definition	Units
e_a	Air vapor pressure	kPa
e_s	Air saturation vapor pressure	kPa
VPD	Air vapor pressure deficit (VPD = $e_s - e_a$)	kPa
RET	Reference evapotranspiration	mm d^{-1}
RET_{GS}	Total growing season RET	mm
RET_{Anom}	Daily RET anomaly, defined as the difference between RET at a sensor and mean rural RET for that date (urban sensors only)	mm d^{-1}
$RET_{Anom:T}$	Total growing season RET_{Anom} (urban sensors only)	mm
$RET_{Anom:L}$	$RET_{Anom:T}$ due to changes in growing season length (urban sensors only)	mm
$RET_{Anom:W}$	$RET_{Anom:T}$ due to changes in within-season meteorology (urban sensors only)	mm
RET_{+V}	RET_{Anom} due only to UHI-induced changes in atmospheric moisture content. Calculated using e_a from each urban sensor and mean rural e_s (urban sensors only).	mm d^{-1}
RET_{+T}	RET_{Anom} due only to UHI-induced changes in temperature. Calculated using e_s from each urban sensor and mean rural e_a (urban sensors only).	mm d^{-1}
PET	Potential evapotranspiration, calculated using crop coefficient method and remotely sensed land cover estimates.	mm d^{-1}
PET_{GS}	Total growing season PET	mm
$PET_{GS\text{-}UHI}$	Total growing season PET with UHI effect removed	mm

total growing season PET (PET_{GS}) at 30 m resolution based on the growing season length and the relative proportion of impervious, tree, and grass cover. To quantify the direct impact of the UHI on PET, we perform similar calculations using mean rural daily RET and observed land cover to remove the UHI effect ($PET_{GS\text{-}UHI}$). The difference between PET_{GS} and $PET_{GS\text{-}UHI}$ at each pixel is attributable to the UHI effect, and we define a metric called the UHI PET enhancement as $100 * (PET_{GS} - PET_{GS\text{-}UHI})/PET_{GS\text{-}UHI}$ to quantify UHI-induced increases in PET for vegetation remaining within the city. Finally, to assess whether a subdaily temporal mismatch between peak UHI intensity (nighttime) and peak radiation intensity (daytime) impacted our results, we performed a parallel set of analyses for the 2015 growing season using 15 min temporal resolution data. Additional methodological details are provided in the supporting information [*Anselin*, 1988, 1995, 2002; *Hengl et al.*, 2007; *Zipper and Loheide*, 2014; *Homer et al.*, 2015].

3. Results

3.1. UHI Impacts on RET

Daily RET is highest in urban areas, intermediate in parks, and lowest in rural areas (Figure 1). Urban RET is up ~0.4 mm/d higher than rural RET, with a maximum daily increase of 7.36% within the growing season (not shown), while park RET falls between urban and rural end-members. Overall, urban RET is significantly higher than rural RET ($p < 0.05$) on 98.2% of days, while park RET is significantly higher than rural RET on 34.6% of days and lower on 0.1% of days, and 15 day moving average urban and park RET are consistently above rural RET throughout the growing season. Differences between classes are comparable from year to year, with maximum moving average urban increases of 0.12 to 0.20 mm/d (2.6–4.8%) during the growing season and minimal differences during the winter, corresponding to seasonal changes in UHI intensity [*Schatz and Kucharik*, 2014]. Urban-rural differences are largest during the peak growing season; across all years, the maximum moving average urban-rural difference occurs between 9 July and 28 July, a period when RET rates are the highest and therefore plants more likely to be water stressed.

At a subdaily scale, UHI impacts on RET are largest during daylight hours, with peak effects during middle to late summer (Figure S5). Mean park RET is consistently between mean urban and rural RET, although peak park-rural differences occur in midmorning while peak urban-rural differences occur in early afternoon. When aggregated to a daily scale, RET calculated using subdaily data is slightly higher than RET calculated using daily data (Figure S6), although differences between classes are comparable throughout the year (Figure S7). Additional information regarding subdaily RET is provided in the supporting information.

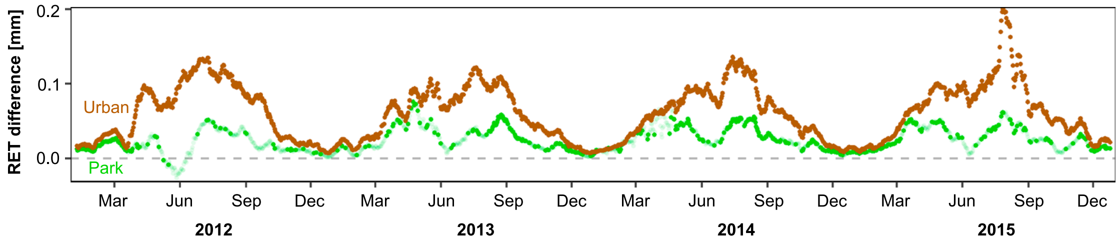

Figure 1. The 15 day moving average difference between urban (red) and park (green) RET and rural baseline RET. Semitransparent points are not significantly different from rural sensors for that measurement date ($p > 0.05$).

RET_{GS}, which represents the total atmospheric water demand from the start to end of the vegetative growing season, is a function of impervious cover within 500 m of each sensor (Figures 2a–2d): a 10% increase in impervious cover leads to a 4.9–6.7 mm increase in RET_{GS}, with similar relationships across all growing seasons. This leads to consistent urban-park-rural rank ordering when aggregated to land cover types (Figures 2e–2h). Median urban RET_{GS} is 19.2–25.5 mm larger than median rural RET_{GS} across our four growing seasons, a 3.6–4.6% increase in evapotranspirative demand. Park sensors are intermediate between urban and rural classes, with median park RET_{GS} 7.6–14.4 mm (1.1%–2.6%) higher than rural RET_{GS}, indicating that the UHI is being buffered by a park cool island effect [*Chang et al.*, 2007; *Cao et al.*, 2010; *Doick et al.*, 2014; *Feyisa et al.*, 2014; *Zipper et al.*, 2016].

3.2. Drivers of RET Changes

VPD, the difference between e_s and e_a, represents the ability of the atmosphere to hold additional water that is evaporated or transpired and is the primary mechanistic pathway through which air temperature and air moisture content influence RET. Our results reveal that differences in VPD between urban and rural areas are almost entirely due to UHI-induced increases in e_s (a function of temperature) rather than decreased e_a (a function of air moisture content). While daily urban e_s is significantly higher than rural e_s ($p < 0.0001$; Figure 3a), urban and rural e_a are statistically identical ($p = 0.15$; Figure 3b). The observed differences in e_a and e_s lead to a mean increase in urban VPD of 1.10 kPa for every 1.0 kPa increase in rural VPD, with 83.4% of the urban-rural difference driven by temperature-induced changes in e_s.

Similarly, changes in temperature explain the vast majority ($R^2 = 0.94$) of total variability in daily urban RET anomalies while changes in air moisture content at urban sensors have a minor influence. Urban air temperature effects on daily RET are an order of magnitude larger than the effects of changes in air moisture content (0.081 mm versus 0.008 mm; Figure 3c). Combined, these results mechanistically explain the escalating demand during the peak growing season observed in Figure 1, as Madison's UHI is largest during the

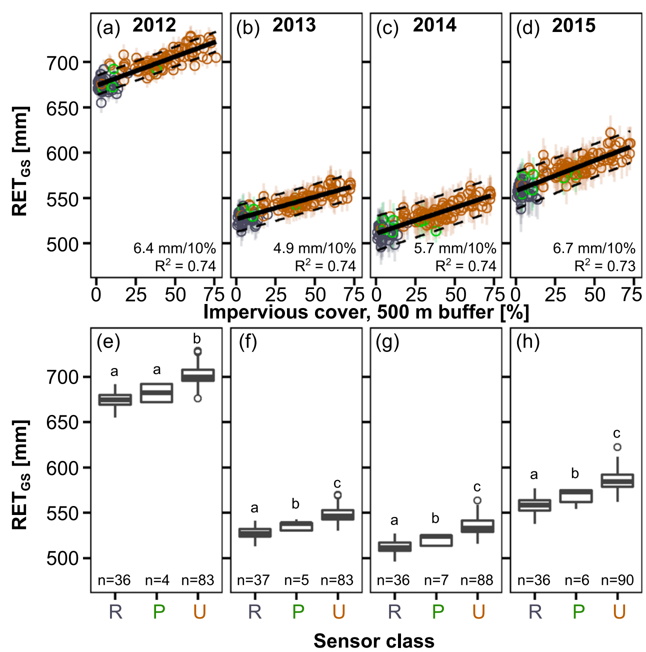

Figure 2. Total growing season RET_{GS} as a function of (a–d) impervious cover and (e–h) sensor class. Colors in Figures 2a–2d correspond to label colors in Figures 2e–2h, which denote rural (R), park (P), and urban (U) classes. In Figures 2a–2d, points are the mean of all permutations for each sensor, and the vertical lines represent one standard deviation above and below mean; the solid and dashed lines show linear fits to mean and upper/lower standard deviations, respectively. In Figures 2e–2h, different letters above boxplots denote statistically significant differences between the means of sensor classes within each year using a pairwise *t* test ($p < 0.05$).

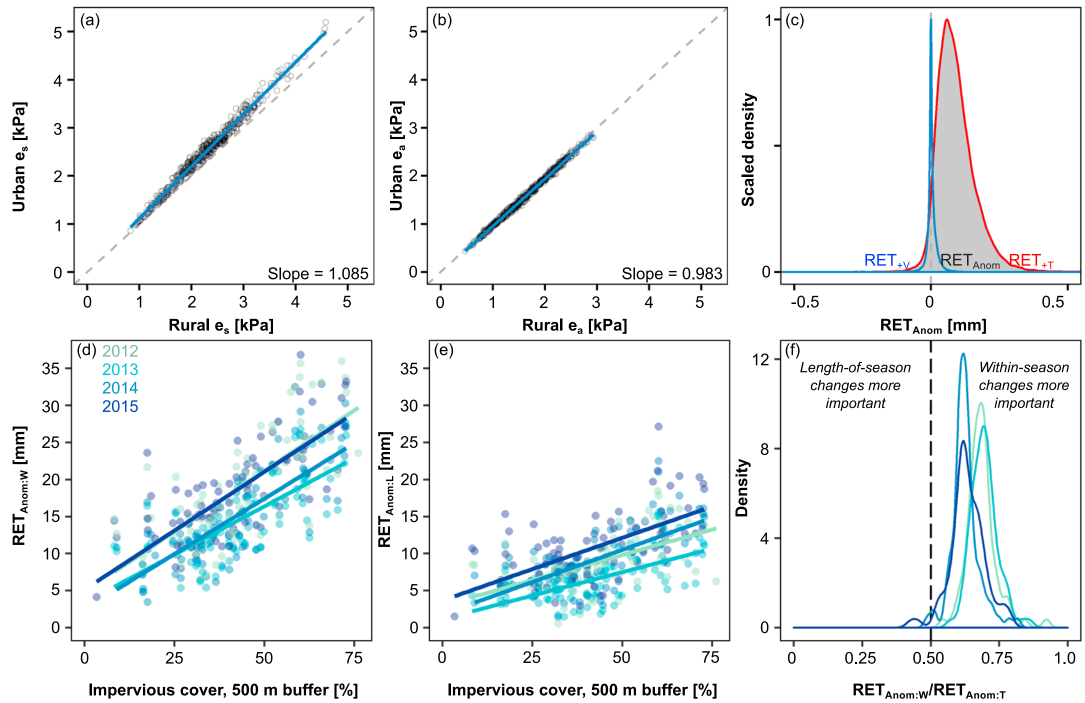

Figure 3. Effects of UHI-induced changes in air temperature and moisture on RET anomalies, and relative contribution of within-season and length-of-season changes. (a and b) Comparison between mean daily urban and rural e_s and e_a, respectively. (c) The lake effect-corrected relative contribution of changes in temperature only (RET$_{+T}$), vapor pressure only (RET$_{+V}$), and both temperature and vapor pressure (RET$_{Anom}$, gray shading). (d and e) Urban RET anomalies due to within-season and length-of-season changes, respectively, as a function of impervious cover for all urban sensors where RET$_{Anom:T}$ is greater than 5 mm. (f) A density plot demonstrating the proportion of RET$_{Anom:T}$ due to within-season changes. Colors noted in Figure 3d apply to Figures 3d–3f.

warm summer months [*Schatz and Kucharik*, 2014] and during extreme heat events [*Schatz and Kucharik*, 2015].

Both changes in within-season meteorological conditions and the length of the growing season contribute to larger RET$_{GS}$ in urban areas. RET$_{Anom:W}$ and RET$_{Anom:L}$ vary at fine spatial scales in response to impervious cover. Across our study years, impervious cover explains 53%–61% of RET$_{Anom:W}$ (Figure 3d) and 33–44% of RET$_{Anom:L}$ (Figure 3e). Slopes for both RET$_{Anom:W}$ and RET$_{Anom:L}$ are statistically indistinguishable ($p > 0.05$) among years. In tandem, observed patterns of RET$_{Anom:W}$ and RET$_{Anom:L}$ indicate that both length-of-season and within-season anomalies represent a consistent interannual source of enhanced atmospheric water demand strongly correlated with urban density, although within-season anomalies make up a larger component of total anomalies across all growing seasons (median RET$_{Anom:W}$/RET$_{Anom:T}$ of 62–70%; Figure 3f).

3.3. Potential ET and Interactions With Land Cover

Due to the strong correlation with impervious cover, RET$_{GS}$ is largest at the center of Madison (Figure 4). The decrease of RET$_{GS}$ as a function of distance from the city center can be described as a logarithmic decay ($R^2 = 0.91$–0.94) with no effects at distances $>$ ~12 km from the city center, a distance corresponding approximately to the edge of Madison's urban extent [*Zipper et al.*, 2016]. In contrast, PET$_{GS}$ is ~500 mm lower at the city center relative to the rural surroundings and increases rapidly within 5 km of the city center. This decrease in PET$_{GS}$ within Madison's urban core is due to increased impervious (nontranspiring) cover in the city center (Figure S1) and is the primary driver of the UHI effect. Across all growing seasons, the UHI PET enhancement approaches ~10% closest to the city center with an average of 4.6–6.4% within the 5 km closest to the city center.

4. Discussion

Our results reveal significantly higher water requirements in urban areas relative to their rural surroundings. Mechanistically, air temperature in urban areas is higher, leading to increased e_s, VPD, and RET. Rather than a

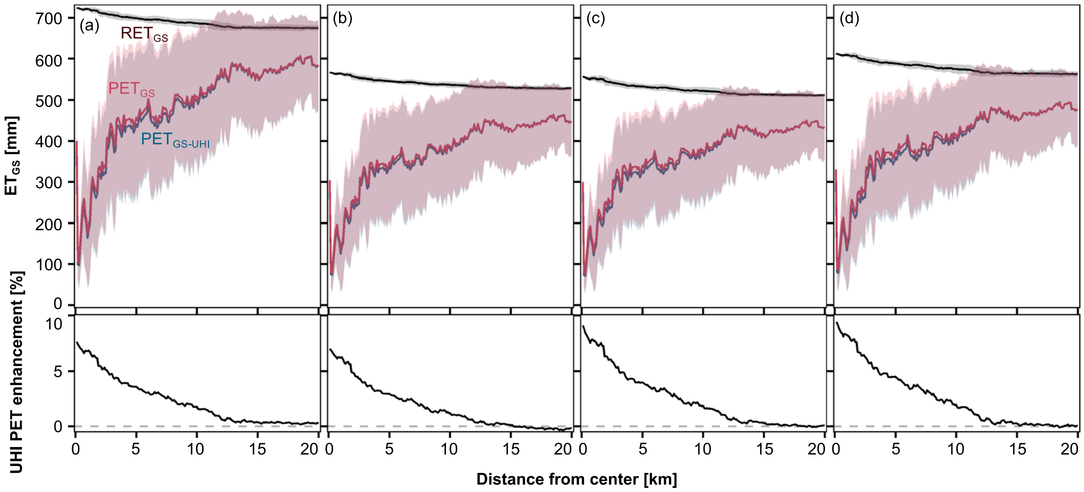

Figure 4. ET as a function of distance from city center for (a–d) 2012–2015. Mean (lines) and standard deviation (shading) RET_{GS}, PET_{GS}, and $PET_{GS\text{-}UHI}$ are shown for buffers ranging from 0.1 to 20 km from the city center (top). In the bottom plots, the UHI PET enhancement (the additional PET within urban areas occurring due to the UHI) is shown as a function of distance from the city center.

homogeneous increase across the year, the UHI increases water demand most strongly during midsummer when soil moisture supplies are most likely to be depleted, thus potentially exacerbating existing water stress or increasing the probability of entering a water stress regime. These results indicate that design efforts focused on reducing summer urban temperatures, such as cool roofs or urban parks, are likely to be most effective at reducing urban VPD and plant water stress [*Georgescu et al.*, 2012; *Melaas et al.*, 2016; *Sharma et al.*, 2016; *Vaz Monteiro et al.*, 2016], although care must be taken to select solutions appropriate to the scale of the problem and well suited to local climatic conditions [*Georgescu et al.*, 2015]. The strong association between fine-scale variability in impervious cover and increased RET highlights a management opportunity, as increased urban green space may reduce water stress in urban ecosystems at local scales in addition to well-known benefits such as stormwater management, improved human health, and biodiversity enhancement [*Davies et al.*, 2009; *Wadzuk et al.*, 2010; *Kardan et al.*, 2015; *Versini et al.*, 2015; *Hartig and Kahn*, 2016]; identifying such synergies between ecosystem services is key to creating sustainable urban socio-environmental systems [*Lundy and Wade*, 2011; *Ziter*, 2016].

These results indicate that hydrological models, which have recently made major advances in representing surface and subsurface flow in urban environments [*Bhaskar et al.*, 2015; *Shields and Tague*, 2015], must consider the UHI impacts on evapotranspirative demand to accurately simulate urban ecohydrological processes. By demonstrating substantial and predictable variability in RET_{GS} as a function of impervious cover within urban areas, we show that land cover must be considered as a driver of variability in evapotranspirative demand over fine spatial scales. While much previous urban ecological research has focused on the impacts of the UHI on growing season length [*Buyantuyev and Wu*, 2012; *Jochner and Menzel*, 2015; *Schatz and Kucharik*, 2016; *Zipper et al.*, 2016], our results suggest that studies focusing solely on UHI-induced changes in growing season length are inadequate to understand UHI effects on plant water requirements.

While our results focus only on Madison, WI, we anticipate that our conclusions may be broadly true across a range of temperate urban areas. Although lakes are an important component of Madison's local geography, lake effects on the UHI are relatively minor and confined to nearshore areas [*Schatz and Kucharik*, 2014]. Furthermore, we find that temperature drives virtually all UHI impacts on RET; urban increases in temperature have been documented worldwide, and the effects of the UHI on RET and PET would likely be greater in larger cities which have stronger UHI effects [*Oke*, 1973]. Therefore, the traditional view of a one-way relationship between urban vegetation and UHI (i.e., decreases in vegetation cover lead to proportionally reduced ET and the formation of an UHI) may not adequately capture bidirectional feedbacks (the formation of an UHI then increases the ET of the remaining vegetation). However, increases in ET due to the UHI will only occur where sufficient soil moisture exists to sustain higher ET levels, for example, in temperate climates or irrigated areas; this highlights the need to consider climate- and management-specific factors when assessing the UHI impacts on the urban energy and water balance [*Georgescu et al.*, 2015].

More broadly, these results have implications for carbon, energy, and water cycles at city to watershed scales. UHI-induced increases in RET are a physical mechanism for previously observed increases in water demand [*Rahman et al.*, 2014] and decreases in water use efficiency [*van Rensburg et al.*, 1997] in urban areas and may explain why vegetation-induced air cooling has been observed to be strongest in hotter areas within cities [*Jenerette et al.*, 2016] and during heat waves [*Shiflett et al.*, 2016]. Furthermore, increases in total growing season evapotranspirative demand associated with increased temperature (due to the UHI and also observed due to climate change) are a potential causal mechanism for increases in the strength of the relationship between growing season length and carbon storage during wet years [*White and Nemani*, 2003; *Penuelas et al.*, 2009], as increased VPD has been shown to reduce both photosynthesis and transpiration [*Sulman et al.*, 2016]. This implies that recent observations of decreased carbon storage associated with urban warming may result from enhanced VPD and water stress [*Meineke et al.*, 2016], indicating that urban carbon sequestration will not increase due to a longer growing season unless accompanied by additional precipitation.

5. Conclusions

Quantifying the water requirements of urban ecosystems requires a mechanistic understanding of urban ecohydrology. Here we make a contribution to our understanding of evapotranspiration in the urban environment, one of the most challenging hydrological fluxes to quantify. We demonstrate that the UHI significantly increases evapotranspirative demand associated with fine-scale spatial variability in impervious cover, with consistent relationships from year to year. These changes in RET are due primarily to changes in temperature and most strongly impact peak RET rates within the growing season. Changes in growing season length associated with the UHI cause a smaller but still important increase in RET, while differences in air moisture content between urban and rural areas have a negligible effect. When accounting for reduced vegetation within urban areas, we find that total growing season potential ET is enhanced by up to 10% compared to rural areas, which may partially counteract reductions in actual ET due to reduced transpiring cover. Therefore, we expect UHI-induced changes in RET to significantly impact patterns of water, energy, and carbon cycling within cities.

Acknowledgments
We thank Aditya Singh and Philip A. Townsend for their assistance with atmospheric correction of the Landsat imagery used for species composition estimates and feedback from Paul Block, Michael Cardiff, and Philip A. Townsend on an earlier version of this manuscript. We further appreciate the helpful comments from two anonymous reviewers and Editor M. Bayani Cardenas, which improved the manuscript. This work was funded by the National Science Foundation Water Sustainability & Climate Program (DEB-1038759) and the North Temperate Lakes Long-Term Ecological Research Program (DEB-0822700). Data are available by contacting the authors.

References

Allen, R. G., L. S. Pereira, D. Raes, and M. Smith (1998), Crop evapotranspiration: Guidelines for computing crop water requirements, FAO Irrigation and Drainage Paper No. 56, United Nations Food and Agriculture Organization, Rome.

Anselin, L. (1988), Lagrange multiplier test diagnostics for spatial dependence and spatial heterogeneity, *Geogr. Anal.*, *20*(1), 1–17, doi:10.1111/j.1538-4632.1988.tb00159.x.

Anselin, L. (1995), Local indicators of spatial association—LISA, *Geogr. Anal.*, *27*(2), 93–115.

Anselin, L. (2002), Under the hood: Issues in the specification and interpretation of spatial regression models, *Agric. Econ.*, *27*(3), 247–267, doi:10.1016/S0169-5150(02)00077-4.

Bhaskar, A. S., C. Welty, R. M. Maxwell, and A. J. Miller (2015), Untangling the effects of urban development on subsurface storage in Baltimore, *Water Resour. Res.*, *51*, 1158–1181, doi:10.1002/2014WR016039.

Buyantuyev, A., and J. Wu (2012), Urbanization diversifies land surface phenology in arid environments: Interactions among vegetation, climatic variation, and land use pattern in the Phoenix metropolitan region, USA, *Landscape Urban Plann.*, *105*(1–2), 149–159, doi:10.1016/j.landurbplan.2011.12.013.

Cao, X., A. Onishi, J. Chen, and H. Imura (2010), Quantifying the cool island intensity of urban parks using ASTER and IKONOS data, *Landscape Urban Plann.*, *96*(4), 224–231, doi:10.1016/j.landurbplan.2010.03.008.

Chang, C.-R., M.-H. Li, and S.-D. Chang (2007), A preliminary study on the local cool-island intensity of Taipei city parks, *Landscape Urban Plann.*, *80*(4), 386–395, doi:10.1016/j.landurbplan.2006.09.005.

Davies, Z. G., R. A. Fuller, A. Loram, K. N. Irvine, V. Sims, and K. J. Gaston (2009), A national scale inventory of resource provision for biodiversity within domestic gardens, *Biol. Conserv.*, *142*(4), 761–771, doi:10.1016/j.biocon.2008.12.016.

Declet-Barreto, J., A. J. Brazel, C. A. Martin, W. T. L. Chow, and S. L. Harlan (2013), Creating the park cool island in an inner-city neighborhood: Heat mitigation strategy for Phoenix, AZ, *Urban Ecosyst.*, *16*(3), 617–635, doi:10.1007/s11252-012-0278-8.

Declet-Barreto, J., K. Knowlton, G. D. Jenerette, and A. Buyantuev (2016), Effects of urban vegetation on mitigating exposure of vulnerable populations to excessive heat in Cleveland, Ohio, *Weather Clim. Soc.*, *8*(4), 507–524, doi:10.1175/WCAS-D-15-0026.1.

Denman, E. C., P. B. May, and G. M. Moore (2016), The potential role of urban forests in removing nutrients from stormwater, *J. Environ. Qual.*, *45*(1), 207–214, doi:10.2134/jeq2015.01.0047.

Doick, K. J., A. Peace, and T. R. Hutchings (2014), The role of one large greenspace in mitigating London's nocturnal urban heat island, *Sci. Total Environ.*, *493*, 662–671, doi:10.1016/j.scitotenv.2014.06.048.

Feyisa, G. L., K. Dons, and H. Meilby (2014), Efficiency of parks in mitigating urban heat island effect: An example from Addis Ababa, *Landscape Urban Plann.*, *123*, 87–95, doi:10.1016/j.landurbplan.2013.12.008.

Gazal, R., M. A. White, R. Gillies, E. Rodemaker, E. Sparrow, and L. Gordon (2008), GLOBE students, teachers, and scientists demonstrate variable differences between urban and rural leaf phenology, *Global Change Biol.*, *14*(7), 1568–1580, doi:10.1111/j.1365-2486.2008.01602.x.

Georgescu, M., A. Mahalov, and M. Moustaoui (2012), Seasonal hydroclimatic impacts of Sun Corridor expansion, *Environ. Res. Lett.*, *7*(3), 34026, doi:10.1088/1748-9326/7/3/034026.

Georgescu, M., M. Moustaoui, A. Mahalov, and J. Dudhia (2013), Summer-time climate impacts of projected megapolitan expansion in Arizona, *Nat. Clim. Change*, *3*(1), 37–41.

Georgescu, M., W. T. L. Chow, Z. H. Wang, A. Brazel, B. Trapido-Lurie, M. Roth, and V. Benson-Lira (2015), Prioritizing urban sustainability solutions: Coordinated approaches must incorporate scale-dependent built environment induced effects, *Environ. Res. Lett.*, *10*(6), 61001, doi:10.1088/1748-9326/10/6/061001.

Hartig, T., and P. H. Kahn (2016), Living in cities, naturally, *Science*, *352*(6288), 938–940, doi:10.1126/science.aaf3759.

Hengl, T., G. B. M. Heuvelink, and D. G. Rossiter (2007), About regression-kriging: From equations to case studies, *Comput. Geosci.*, *33*(10), 1301–1315, doi:10.1016/j.cageo.2007.05.001.

Homer, C., J. Dewitz, L. Yang, S. Jin, P. Danielson, G. Xian, J. Coulston, N. Herold, J. Wickham, and K. Megown (2015), Completion of the 2011 National Land Cover Database for the conterminous United States—Representing a decade of land cover change information, *Photogramm. Eng. Remote Sens.*, *81*(5), 345–354, doi:10.14358/PERS.81.5.345.

Jacobs, C., J. Elbers, R. Brolsma, O. Hartogensis, E. Moors, M. T. R.-C. Marquez, and B. van Hove (2015), Assessment of evaporative water loss from Dutch cities, *Build. Environ.*, *83*, 27–38, doi:10.1016/j.buildenv.2014.07.005.

Jenerette, G. D., S. L. Harlan, A. Buyantuev, W. L. Stefanov, J. Declet-Barreto, B. L. Ruddell, S. W. Myint, S. Kaplan, and X. Li (2016), Micro-scale urban surface temperatures are related to land-cover features and residential heat related health impacts in Phoenix, AZ USA, *Landscape Ecol.*, *31*(4), 745–760, doi:10.1007/s10980-015-0284-3.

Jochner, S., and A. Menzel (2015), Urban phenological studies—Past, present, future, *Environ. Pollut.*, *203*, 250–261, doi:10.1016/j.envpol.2015.01.003.

Kardan, O., P. Gozdyra, B. Misic, F. Moola, L. J. Palmer, T. Paus, and M. G. Berman (2015), Neighborhood greenspace and health in a large urban center, *Sci. Rep.*, *5*, 11610, doi:10.1038/srep11610.

Koeser, A., R. Hauer, K. Norris, and R. Krouse (2013), Factors influencing long-term street tree survival in Milwaukee, WI, USA, *Urban For. Urban Green.*, *12*(4), 562–568, doi:10.1016/j.ufug.2013.05.006.

Lundy, L., and R. Wade (2011), Integrating sciences to sustain urban ecosystem services, *Prog. Phys. Geogr.*, *35*(5), 653–669, doi:10.1177/0309133311422464.

McDonnell, M. J., and I. MacGregor-Fors (2016), The ecological future of cities, *Science*, *352*(6288), 936–938, doi:10.1126/science.aaf3630.

Meineke, E., E. Youngsteadt, R. R. Dunn, and S. D. Frank (2016), Urban warming reduces aboveground carbon storage, *Proc. R. Soc. B*, *283*(1840), 20161574, doi:10.1098/rspb.2016.1574.

Melaas, E. K., J. A. Wang, D. L. Miller, and M. A. Friedl (2016), Interactions between urban vegetation and surface urban heat islands: A case study in the Boston metropolitan region, *Environ. Res. Lett.*, *11*(5), 54020, doi:10.1088/1748-9326/11/5/054020.

Nowak, D. J., S. Hirabayashi, A. Bodine, and E. Greenfield (2014), Tree and forest effects on air quality and human health in the United States, *Environ. Pollut.*, *193*, 119–129, doi:10.1016/j.envpol.2014.05.028.

Oke, T. R. (1973), City size and the urban heat island, *Atmos. Environ.*, *7*(8), 769–779, doi:10.1016/0004-6981(73)90140-6.

Oke, T. R. (1982), The energetic basis of the urban heat island, *Q. J. R. Meteorol. Soc.*, *108*(455), 1–24.

Oke, T. R. (1988), The urban energy balance, *Prog. Phys. Geogr.*, *12*(4), 471–508, doi:10.1177/030913338801200401.

Penuelas, J., T. Rutishauser, and I. Filella (2009), Phenology feedbacks on climate change, *Science*, *324*(5929), 887–888, doi:10.1126/science.1173004.

Peters, E. B., J. P. McFadden, and R. A. Montgomery (2010), Biological and environmental controls on tree transpiration in a suburban landscape, *J. Geophys. Res.*, *115*, G04006, doi:10.1029/2009JG001266.

Peters, E. B., R. V. Hiller, and J. P. McFadden (2011), Seasonal contributions of vegetation types to suburban evapotranspiration, *J. Geophys. Res.*, *116*, G01003, doi:10.1029/2010JG001463.

Rahman, M. A., D. Armson, and A. R. Ennos (2014), Effect of urbanization and climate change in the rooting zone on the growth and physiology of Pyrus calleryana, *Urban For. Urban Green.*, *13*(2), 325–335, doi:10.1016/j.ufug.2013.10.004.

Ramaswami, A., A. G. Russell, P. J. Culligan, K. R. Sharma, and E. Kumar (2016), Meta-principles for developing smart, sustainable, and healthy cities, *Science*, *352*(6288), 940–943, doi:10.1126/science.aaf7160.

Richardson, A. D., A. S. Bailey, E. G. Denny, C. W. Martin, and J. O'Keefe (2006), Phenology of a northern hardwood forest canopy, *Global Change Biol.*, *12*(7), 1174–1188, doi:10.1111/j.1365-2486.2006.01164.x.

Schatz, J., and C. J. Kucharik (2014), Seasonality of the urban heat island effect in Madison, Wisconsin, *J. Appl. Meteorol. Climatol.*, *53*(10), 2371–2386, doi:10.1175/JAMC-D-14-0107.1.

Schatz, J., and C. J. Kucharik (2015), Urban climate effects on extreme temperatures in Madison, Wisconsin, USA, *Environ. Res. Lett.*, *10*(9), 94024, doi:10.1088/1748-9326/10/9/094024.

Schatz, J., and C. J. Kucharik (2016), Urban heat island effects on growing seasons and heating and cooling degree days in Madison, Wisconsin USA, *Int. J. Climatol.*, doi:10.1002/joc.4675.

Sharma, A., P. Conry, H. J. S. Fernando, A. F. Hamlet, J. J. Hellmann, and F. Chen (2016), Green and cool roofs to mitigate urban heat island effects in the Chicago metropolitan area: Evaluation with a regional climate model, *Environ. Res. Lett.*, *11*(6), 64004, doi:10.1088/1748-9326/11/6/064004.

Shields, C., and C. Tague (2015), Ecohydrology in semiarid urban ecosystems: Modeling the relationship between connected impervious area and ecosystem productivity, *Water Resour. Res.*, *51*, 302–319, doi:10.1002/2014WR016108.

Shiflett, S. A., L. L. Liang, S. M. Crum, G. L. Feyisa, J. Wang, and G. D. Jenerette (2016), Variation in the urban vegetation, surface temperature, air temperature nexus, *Sci. Total Environ.*, doi:10.1016/j.scitotenv.2016.11.069.

Sjoman, H., and A. B. Nielsen (2010), Selecting trees for urban paved sites in Scandinavia—A review of information on stress tolerance and its relation to the requirements of tree planners, *Urban For. Urban Green.*, *9*(4), 281–293, doi:10.1016/j.ufug.2010.04.001.

Smoliak, B. V., P. K. Snyder, T. E. Twine, P. M. Mykleby, and W. F. Hertel (2015), Dense network observations of the Twin Cities canopy-layer urban heat island, *J. Appl. Meteorol. Climatol.*, *54*(9), 1899–1917, doi:10.1175/JAMC-D-14-0239.1.

Stewart, I. D., and T. R. Oke (2012), Local climate zones for urban temperature studies, *Bull. Am. Meteorol. Soc.*, *93*(12), 1879–1900, doi:10.1175/BAMS-D-11-00019.1.

Sulman, B. N., D. T. Roman, K. Yi, L. Wang, R. P. Phillips, and K. A. Novick (2016), High atmospheric demand for water can limit forest carbon uptake and transpiration as severely as dry soil, *Geophys. Res. Lett.*, *43*, 9686–9695, doi:10.1002/2016GL069416.

Ugolini, F., F. Bussotti, G. M. Lanini, A. Raschi, C. Tani, and R. Tognetti (2012), Leaf gas exchanges and photosystem efficiency of the holm oak in urban green areas of Florence, Italy, *Urban For. Urban Green.*, *11*(3), 313–319, doi:10.1016/j.ufug.2012.02.006.

United Nations (2010), *Growing Greener Cities*, UN Food and Agriculture Organization, Rome.

United Nations (2014), *World Urbanization Prospects: The 2014 Revision*, Department of Economic and Social Affairs (DESA), Population Division, New York.

van Rensburg, L., G. H. J. Kruger, B. Ubbink, J. Stassen, and H. van Hamburg (1997), Seasonal performance of Quercus robur L along an urbanization gradient, *South Africa J. Bot.*, *63*(1), 32–36.

Vaz Monteiro, M., K. J. Doick, P. Handley, and A. Peace (2016), The impact of greenspace size on the extent of local nocturnal air temperature cooling in London, *Urban For. Urban Green.*, *16*, 160–169, doi:10.1016/j.ufug.2016.02.008.

Versini, P.-A., D. Ramier, E. Berthier, and B. de Gouvello (2015), Assessment of the hydrological impacts of green roof: From building scale to basin scale, *J. Hydrol.*, *524*, 562–575, doi:10.1016/j.jhydrol.2015.03.020.

Vico, G., R. Revelli, and A. Porporato (2014), Ecohydrology of street trees: Design and irrigation requirements for sustainable water use, *Ecohydrology*, *7*(2), 508–523, doi:10.1002/eco.1369.

Wadzuk, B. M., M. Rea, G. Woodruff, K. Flynn, and R. G. Traver (2010), Water-quality performance of a constructed stormwater wetland for all flow conditions1, *J. Am. Water Resour. Assoc.*, *46*(2), 385–394, doi:10.1111/j.1752-1688.2009.00408.x.

Wang, H., Z. Ouyang, W. Chen, X. Wang, H. Zheng, and Y. Ren (2011), Water, heat, and airborne pollutants effects on transpiration of urban trees, *Environ. Pollut.*, *159*(8–9), 2127–2137, doi:10.1016/j.envpol.2011.02.031.

White, M. A., and A. R. Nemani (2003), Canopy duration has little influence on annual carbon storage in the deciduous broad leaf forest, *Global Change Biol.*, *9*(7), 967–972, doi:10.1046/j.1365-2486.2003.00585.x.

Zhou, D., S. Zhao, L. Zhang, G. Sun, and Y. Liu (2015), The footprint of urban heat island effect in China, *Sci. Rep.*, *5*, doi:10.1038/srep11160.

Zipper, S. C., and S. P. Loheide (2014), Using evapotranspiration to assess drought sensitivity on a subfield scale with HRMET, a high resolution surface energy balance model, *Agric. For. Meteorol.*, *197*, 91–102, doi:10.1016/j.agrformet.2014.06.009.

Zipper, S. C., J. Schatz, A. Singh, C. J. Kucharik, P. A. Townsend, and S. P. Loheide (2016), Urban heat island impacts on plant phenology: Intra-urban variability and response to land cover, *Environ. Res. Lett.*, *11*(5), 54023, doi:10.1088/1748-9326/11/5/054023.

Ziter, C. (2016), The biodiversity–ecosystem service relationship in urban areas: A quantitative review, *Oikos*, *125*(6), 761–768, doi:10.1111/oik.02883.