

## White oak and red maple tree ring analysis reveals enhanced productivity in urban forest patches



Nancy F. Sonti<sup>a,\*</sup>, Richard A. Hallett<sup>b</sup>, Kevin L. Griffin<sup>c</sup>, Joe H. Sullivan<sup>d</sup>

<sup>a</sup> USDA Forest Service Northern Research Station, Baltimore, MD, USA

<sup>b</sup> USDA Forest Service Northern Research Station, Bayside, New York, USA

<sup>c</sup> Department of Earth and Environmental Sciences, Lamont-Doherty Earth Observatory, Columbia University, Palisades, New York, USA

<sup>d</sup> Department of Plant Science and Landscape Architecture, University of Maryland – College Park, College Park, MD, USA

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### ABSTRACT

Many aspects of the urban environment have the potential to affect tree growth and physiology in forest patches, including higher temperatures, elevated CO<sub>2</sub> concentrations, and modified soil biogeochemistry from increased nutrient and heavy metal inputs. However, these changes in tree growth are likely to vary by species and across urban areas, reflecting the local environmental conditions associated with the idiosyncratic trajectory of development in a city. Here, we examine growth rates of two dominant native tree species (white oak (*Quercus alba* L.) and red maple (*Acer rubrum* L.)) across urban and reference forest sites of three major cities in the eastern United States (New York, NY (NYC); Philadelphia, PA; and Baltimore, MD). We also characterized soil physical and chemical properties around each tree and monitored air temperature over three growing seasons at each site. Throughout the growing season, the urban sites had consistently warmer daytime and nighttime temperatures than reference sites. Urban forest patch soils also had elevated calcium, magnesium, and heavy metal concentrations compared to reference forest soils. Urban vs. reference tree growth rates varied by species, by city, and over time. Despite differences in the two native species' ecophysiology, both grew more rapidly in the urban environment than at nearby reference sites, particularly in recent decades. Over the entire 145-year tree ring record analyzed, white oak basal area increment was significantly higher in urban trees compared to reference trees. Changes in the relative production of white oak earlywood and latewood between site types in each city and over time may relate to precipitation patterns. Perhaps due to their ability to persist in a wide variety of environmental conditions, mature trees of both species appear to be acclimating to urban forest patch conditions of the eastern U.S. and in some cases are experiencing enhanced growth rates compared to trees in nearby reference forests. An understanding of past and present growth rates of trees in urban forest patches can provide insight into future ecosystem functioning of these urban green spaces as well as that of more rural ecosystems experiencing environmental change factors similar to those associated with urbanization.

### 1. Introduction

The environmental conditions of urban ecosystems shape the health and well-being of all organisms living there. The growth and vitality of urban trees in the eastern United States (U.S.) is of particular interest as they are dominant organisms in urban landscapes and provide valuable biophysical and socio-cultural ecosystem services to urban residents. An understanding of past and present growth rates of trees in urban forest patches may provide insight into future ecosystem functioning of these urban green spaces as well as that of more rural ecosystems experiencing environmental change factors similar to those associated with urbanization (Ziska et al., 2003; Lahr et al., 2018). However, these

changes in tree growth are likely to vary by species and across urban areas, reflecting the local environmental conditions associated with the trajectory of development in a city. Here, we examine growth rates of two native tree species (white oak (*Quercus alba* L.) and red maple (*Acer rubrum* L.)) across urban and reference forest sites of three major cities in the eastern U.S. (New York, NY (NYC); Philadelphia, PA; and Baltimore, MD).

Urban forest patches are woodlands contained within a city, whether completely surrounded by urban development or on the urban fringe (Kowarik, 2005). These forests vary in their land use history, ownership, and management, but they tend to be less intensively managed than other more landscaped urban green spaces. In the

\* Corresponding author.

E-mail address: [nancy.f.sonti@usda.gov](mailto:nancy.f.sonti@usda.gov) (N.F. Sonti).

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eastern U.S., where cities have generally replaced temperate deciduous forest, the minimal management of urban forest patches allows for aboveground and belowground ecosystem function that is similar to the forested ecosystems of the surrounding region (Pregitzer et al., 2019). Biophysical ecosystem services provided by these urban green spaces include carbon sequestration, stormwater regulation, nutrient cycling, and temperature regulation, for which it is important to understand tree growth and physiology (Livesley et al., 2016).

Urban trees experience warming air temperatures over time due to both global climate change and the urban heat island (UHI) effect. The UHI refers to increased urban air temperatures compared to surrounding suburban and rural temperatures that tends to develop in areas that contain a high percentage of non-reflective, water-resistant surfaces and a low percentage of vegetated and moisture trapping surfaces relative to the surrounding rural areas (Rosenzweig et al., 2005). Decades of UHI effects have been well-documented in many cities, including NYC, Philadelphia, and Baltimore (Brazel et al., 2000; Rosenzweig et al., 2005; Gaffin et al., 2008). In an exploration of the different sources of warming urban air temperatures, Gaffin et al. (2008) found that the UHI effect was responsible for approximately one third of the total warming of NYC over the twentieth century. Research from NYC and Baltimore suggests that elevated air temperatures, and particularly nighttime temperatures, may be impacting plant growth (George et al., 2009; Searle et al. 2011, 2012). However, most urban forest patches are not located near downtown weather stations used to calculate UHI effects, which vary across the urban landscape depending on land use context (Gaffin et al., 2008; Heisler et al., 2016; Scott et al., 2017). In addition to growth responses to UHI effects, temperate deciduous trees are likely to have a positive growth response to warming temperatures due to global climate change, as respiration has been shown to acclimate more strongly than photosynthesis in these tree populations (Way and Oren, 2010).

Human activity is also known to impact drivers of soil biogeochemical cycles, with implications for the soil properties of urban forest patches (Kaye et al., 2006). Urban air contains increased concentrations of pollutants such as CO<sub>2</sub>, nitrogen oxides, sulfur oxides, O<sub>3</sub> and other volatile organic compounds, as well as atmospheric dust arising from roadways and from construction activities, resulting in more nitrogen and less acidity in urban forests (Lovett et al., 2000; Rao et al., 2014). However, previous studies of forest nitrogen dynamics on urban to rural gradients have shown that urban land use change has complex effects on soil nitrogen cycling that are difficult to predict, and published findings on urban nitrogen cycling are inconsistent (Pouyat et al., 2009). Groffman et al. (2006) suggest that natural soil factors and changes in species composition in both plant and soil communities are ultimately stronger determinants of soil nutrient dynamics than urban atmospheric conditions. Similarly, Pouyat et al. (2008) find that although forest soil chemistry responds to urbanization gradients in three different cities, the nature of the response varies depending on spatial patterns of development, parent material, and pollution sources of each particular city. These factors interact to affect physical and chemical properties of urban soils, which are not uniform across a city (Groffman et al., 2006). As a result, the status of the urban soil biogeochemistry and resulting impacts on tree growth are dynamic, complex, and difficult to predict.

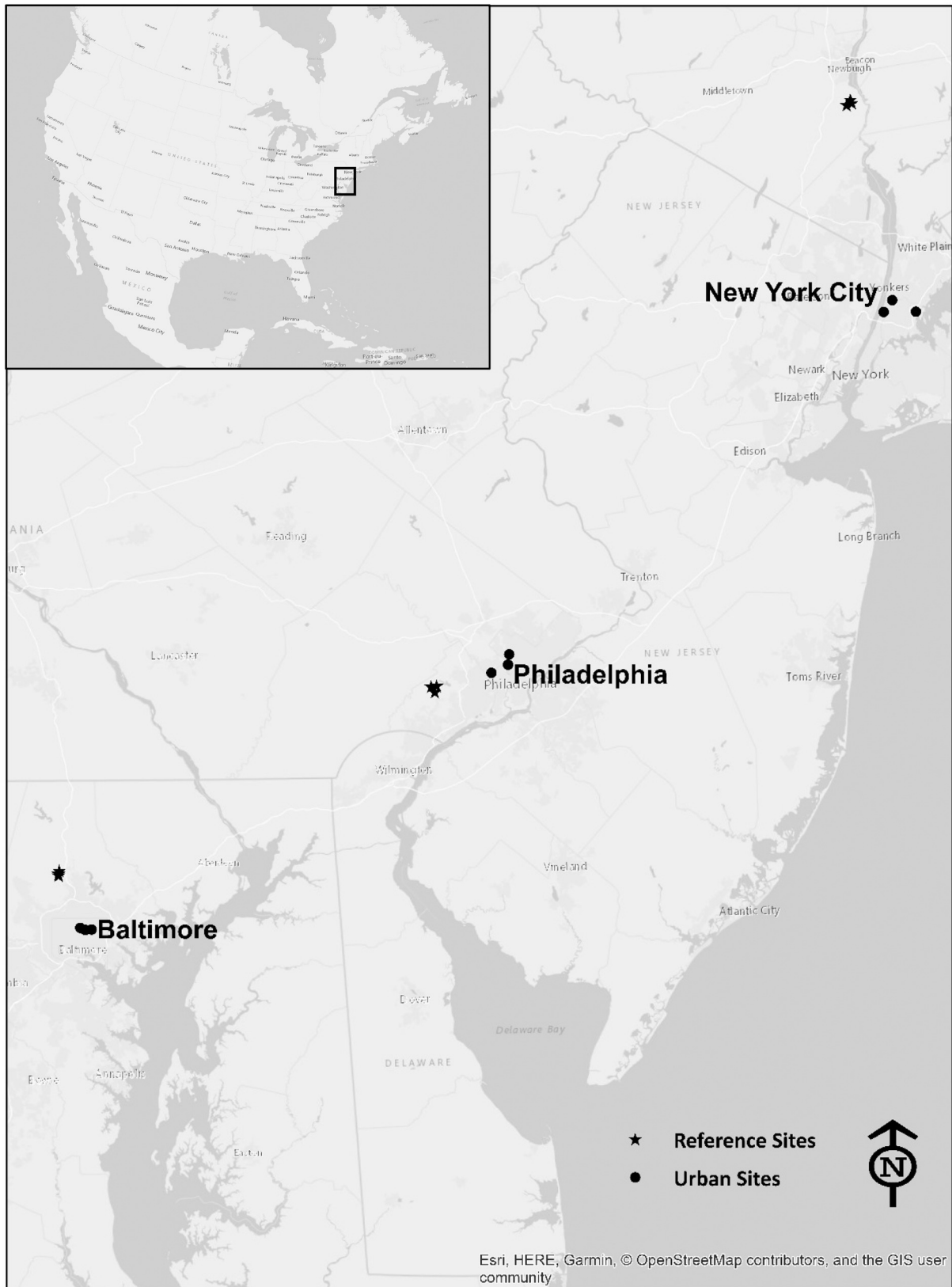
Urban forest patches of the eastern U.S. are largely made up of native species (Groffman et al., 2006; Templeton, 2016; Pregitzer et al., 2019). White oak and red maple trees are two of the most common and ecologically important species in urban and rural forests of the region but occupy different social-ecological roles. Oaks provide critical ecosystem services in forests of the eastern U.S. and their failure to regenerate is causing concern to forest managers (Schuster et al., 2008). In particular, the slow-growing white oak has declined since European settlement of the eastern U.S. relative to more disturbance adapted oak species (Abrams, 2003). White oak historically benefitted from low levels of disturbance caused by understory fires, which minimized the

occurrence of thin-barked, shade tolerant species such as red maple. However, as fire has been controlled in the eastern U.S. for over a century, red maple has become one of the most common tree species in urban and rural forests of the region, often replacing oak species (Abrams, 1998, 2003; McDonald et al., 2002; McDonald et al., 2003; Thompson et al., 2013). Although considered an important urban tree species for its ability to thrive in harsh streetscape conditions, red maple is less valued in the context of forest conservation and restoration (Cralle, 2003; Highfield, 2018). Despite these long-term changes in forest composition, eastern populations of white oak are not expected to experience declines in growth based on anthropogenic climate change, as they are adapted to drought conditions and appear relatively insensitive to climate fluctuations (Abrams, 2003; Goldblum, 2010). In fact, as climate change creates warmer and drier forest conditions, white oak may again become more dominant (Goldblum, 2010). However, red maple is also well-adapted to a variety of ecological conditions and may benefit more from increased nitrogen deposition in urban areas due to its arbuscular mycorrhizal associations (Abrams, 1998; Thomas et al., 2010; Phillips et al., 2013).

Recent research examining urban tree growth rates in highly developed urban settings in comparison with rural forest tree growth has yielded mixed results. Depending on the city and tree species studied, research from the midwestern U.S. has found that trees in rural forests may grow more slowly or more rapidly than trees on maintained sites in nearby urban communities (Iakovoglou et al., 2002; Quigley, 2004). Briber et al. (2015) found red oak basal area increment to double following recent land use conversion from forest to urban land use in the northeastern U.S. In a study examining a century of urban tree growth across multiple cities, Pretzsch et al. (2017) found open grown urban trees to have undergone accelerated growth in recent decades and compared to their rural counterparts. Although we know that the urban environment may be stressful for trees planted in highly developed sites (Cregg and Dix, 2001; Gillner et al., 2014), it is not clear that trees in urban forest patches experience the same environmental impacts. In one study of tree growth within forest fragments along an urbanization gradient, McClung and Ibáñez (2018) found that the negative impacts of impervious surface cover varied by species. The urban forest encompasses trees growing across a wide range of land uses, site types, and environmental conditions, which are known to impact tree growth rates. Trees in urban forest patches may have different growth rates than those in more open park or built environment settings, although this response is species dependent (Fahey et al., 2013; Bialecki et al., 2018).

In addition to basal area increment, the proportion of earlywood and latewood in annual tree rings is an important property of radial growth. The ring-porous wood found in oak species is characterized by the division of each annual ring into low-density earlywood containing larger vessels and high-density latewood containing smaller vessels (Genet et al., 2013). Because earlywood typically forms before leaf expansion (Fritts, 1976), it should mainly be controlled by endogenous factors, and is relatively constant within individuals and species. By contrast, latewood is set during the growing season (Fritts, 1976) and thus should be more sensitive to exogenous factors of the current season. As such, latewood is more likely to vary according to annual growing conditions such as climate and water-related factors (Bergès et al., 2008; Sousa et al., 2016). Therefore, the proportion of latewood tends to be more highly correlated with overall ring width (Bergès et al., 2000; Sousa et al., 2016). However, it is not known whether the urban environment impacts the amount of earlywood and latewood of ring-porous trees in urban forest patches by altering local climate conditions during the growing season.

In this study, we compare growth rates of mature white oak and red maple trees located in urban forest patches and nearby rural reference sites across three metropolitan areas of the eastern United States. We hypothesized that differences in growth rates between trees in urban and reference forests would vary by species and by city. We also



**Fig. 1.** Map of the eastern United States showing the location of urban and reference forest patch study sites within each metropolitan area: New York, NY; Philadelphia, PA; and Baltimore, MD.

hypothesized that temperature and soil chemistry would vary between urban and rural forests and across cities. This examination of tree growth, temperature and soil chemistry in multiple cities allows us to assess the generality of urban environmental effects versus the effects of

the idiosyncratic patterns of growth and development in each city. In addition, tree ring records provide insight into the ways that urbanization and climate change can impact existing forests in less developed areas.

## 2. Materials and methods

### 2.1. Study sites

Field data for this study were collected from secondary growth oak-hickory forests found in New York, NY; Philadelphia, PA; and Baltimore, MD, as well as at reference forest sites outside each metropolitan area (Fig. 1, Table S1). The climate of these eastern U.S. cities is strongly seasonal with warm summers and cold winters and all three cities receive ~1100 mm precipitation annually (NOAA).

In all three metropolitan areas, urban forest patches and reference forest sites were selected based on the presence of red maple and white oak canopy dominant trees in patches of at least 1.5 ha with slopes < 25%, and well-drained soils of similar soil series within each metropolitan area (USDA NRCS Web Soil Survey; Table S1). Within each city, three separate urban forest patches were selected to capture the variation in forest patch site conditions across an individual city. All urban sites were located on public parkland or other institutional grounds, with the exception of one Baltimore forest patch that was located on vacant undeveloped land. Available information about urban site histories indicates that most were private estates prior to these current land uses and were influenced by agricultural and/or horticultural practices before being allowed to revert to native forest cover. All reference sites were located in protected areas outside of the city and within intermix wildland-urban interface landscapes, in order to target similar contexts of surrounding land use and population density (Martinuzzi et al., 2015). Reference ecosystems were dominated by mature native trees with minimal management by humans (Reisinger et al., 2016). Three reference sites were selected for each city, located within the same protected area considered representative of rural forests of the region. Although there was some variation in reference site history, all reference sites were protected areas established during the early- to mid-twentieth century allowing forest regeneration to occur after extensive clearcutting for logging and/or agricultural activities ended in the late 1800s - early 1900s (Maher, 1996; Baltimore County, 2007; Rhoads and Block, 2008). We recognize the potential importance of additional local environmental variables, such as surrounding land use and impervious surfaces, which can change over time; however, these dynamic variables are difficult to characterize historically and are not considered here.

### 2.2. Data collection

One Thermochron iButton air temperature sensor (model DS1921G; Maxim/Dallas Semiconductor, Dallas, TX) was installed in the forest understory at each site to establish urban-reference forest temperature differences within each metropolitan area. The sensors were installed from June to September in 2015, and from April to September 2016 and 2017. Hourly air temperature was recorded and summarized to determine daily maximum and minimum values throughout the growing season.

Five mature white oak and five mature red maple trees were selected at each site, for a total of 15 reference and 15 urban trees of each species for each city ( $n = 180$  trees). White oaks were at least 38.1 cm diameter at breast height (DBH), red maples were at least 25.4 cm DBH,

and all trees were dominant or co-dominant canopy trees. The trees had no major trunk cavities and had crown vigor scores of 1 or 2 (< 25% overall canopy damage; Pontius and Hallett, 2014). When there were more than five suitable trees at a given site, the trees were mapped and five were selected at random.

Four mineral soil cores (0–10 cm depth) were collected at a two-meter distance from each tree in each cardinal direction. The soil cores were combined to produce one composite sample per tree, which was air-dried and screened to remove particles > 2 mm. Samples were analyzed for pH in 0.01 M CaCl<sub>2</sub>, organic content using loss-on-ignition (LOI), and soil texture using the hydrometer method (Day 1965). Plant available elements, including Al, Ca, Cd, Cr, Cu, K, Mg, N, Na, P, Pb, and Zn were determined by extracting soil using a modified Morgan's solution (NH<sub>4</sub>OAC, pH 4.8, McIntosh, 1969) and then measuring soil extracts with inductively coupled plasma spectroscopy. Depth of organic horizon was also measured at each soil core location, but there was no organic horizon present at any site except for the three NYC reference sites.

From Fall 2015 to Summer 2016, increment cores were collected from each tree using a 5.15 mm diameter increment borer (Haglöf, Sweden). Cores were collected at a height of 1.4 m and attempted to capture as many years of growth as possible. Permission was not granted to core trees at one urban site in NYC, so increment cores from 170 trees were used in the analysis (Table S2). Two cores were removed from the cross-slope sides of each tree and stored in plastic trays for transport. After being mounted and sanded, the cores were scanned at high resolution, and ring width measurements as well as earlywood/latewood boundaries were made using the CooRecorder software program (Larsson, 2003). Visual cross-dating of cores within each site was validated using the COFECHA program (Grissino-Mayer, 2001). BAI was calculated using the dplR package in R (R Development Core Team, 2008; Bunn et al., 2016). Conversion of ring width to BAI helps to remove variation in radial growth attributable to increasing circumference (Duchesne et al., 2002; Long et al., 2009). LWR was also calculated for each white oak tree ring, as latewood ring width/total ring width x 100 and indicates the amount of latewood contained in each tree ring (Bergès et al., 2000).

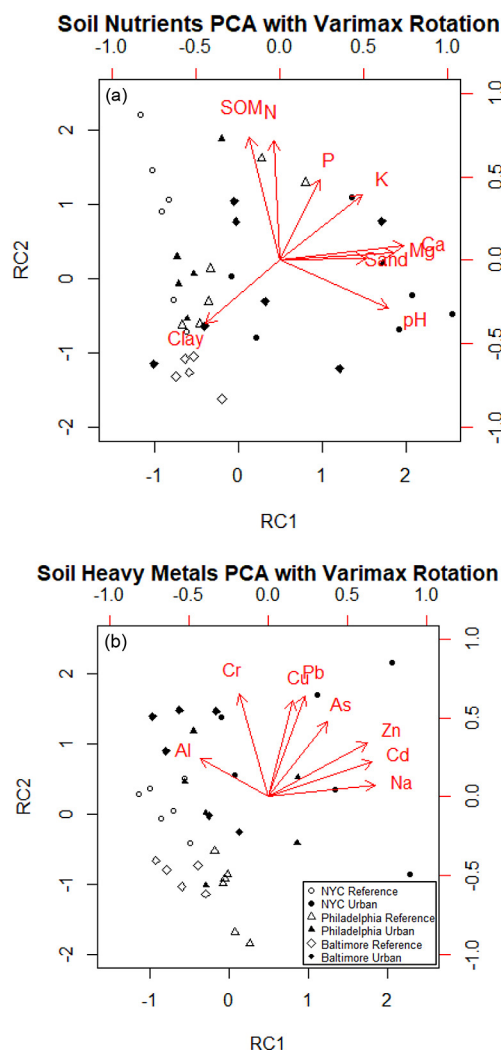
### 2.3. Statistical analyses

Soil data were averaged by site and submitted to principal component analysis (PCA) using the R function `prcomp` in the package `stats` v. 3.5.1 (R Core Team, 2018). Separate PCAs were conducted for soil nutrients and heavy metals. Here, 'nutrients' indicate soil elements and properties essential to plant growth, including ppm of N, P, K, Ca, and Mg; percent sand and clay; pH; and soil organic matter (SOM). 'Heavy metals' indicate elements that may inhibit plant growth and function at high levels, including Al, As, Cd, Cr, Cu, Na, Pb, and Zn (Appenroth, 2010). Al concentrations in soil samples from most sites were found to be at a potentially toxic levels (Amacher et al., 2007), and Na was thought to be potentially toxic due to urban pollution, and so both elements were included as heavy metals. Eigenvalues were used to determine that two principal components should be retained for each soil PCA. Varimax rotation was used to aid in interpretation of the principal components in both analyses using the function `varimax`. The

**Table 1**

Mean daily maximum and minimum air temperatures ( $\pm$  SEM) and urban-reference differences averaged across three years of data collection: June–September 2015, April–September 2016, April–September 2017. Values are averaged for one reference site and three urban sites within each city.

City	Mean Daily Maximum Air Temperature (°C)			Mean Daily Minimum Air Temperature (°C)		
	Urban Sites	Reference Site	Difference (Urban – Reference)	Urban Sites	Reference Site	Difference (Urban – Reference)
New York City	24.59 ( $\pm$ 0.14)	21.64 ( $\pm$ 0.23)	2.95	16.97 ( $\pm$ 0.13)	14.13 ( $\pm$ 0.23)	2.84
Philadelphia	24.82 ( $\pm$ 0.13)	23.56 ( $\pm$ 0.22)	1.27	16.73 ( $\pm$ 0.13)	15.38 ( $\pm$ 0.22)	1.35
Baltimore	25.48 ( $\pm$ 0.14)	24.31 ( $\pm$ 0.22)	1.16	17.15 ( $\pm$ 0.12)	16.35 ( $\pm$ 0.21)	0.81



**Fig. 2.** (a) Biplot of Soil Nutrients Principal Components 1 and 2 after Varimax Rotation, accounting for 40% and 28% of the variation, respectively; (b) Biplot of Soil Heavy Metals Principal Components 1 and 2 after Varimax Rotation, accounting for 45% and 21% of the variation, respectively.

first and second rotated principal components (RC1 and RC2) from each soil PCA were used as response variables in generalized least squares (GLS) models, fit using the R function `gls` in the `nlme` package (Pinheiro et al., 2010). Nine GLS models were constructed for each soil response variable using a combination of city, site type, species, and interaction effects, and AIC values were used to determine the best fit model.

Linear mixed effects models in the `nlme` package (Pinheiro et al., 2010) were used to analyze the effects of city and site type (urban vs. reference) and their interaction effect on basal area increment (BAI) of white oak and red maple trees and latewood ratio (LWR) of white oak trees over time. The `lme` function allowed the use of tree as random variable in each model to account for repeated measurements on the same tree over time. In addition, different variance structures were incorporated into the models using the `varIdent` function to meet assumptions of homoscedasticity (Table 3). Differences between means were considered significant at  $\alpha = 0.05$ .

### 3. Results

#### 3.1. Air temperature

Average daily minimum and maximum temperatures (°C) at the urban and reference field sites are shown in Table 1. Each city's urban

**Table 2**

Generalized least squares model summaries for soil nutrients and heavy metals varimax rotated principal components (RC) response variables. Soil samples were collected from beneath mature red maple and white oak trees at urban and reference forest sites in New York, NY; Philadelphia, PA; and Baltimore, MD. F-values and p-values (bold) are listed for each fixed effect, and grey cells indicate parameters excluded from the best-fit model for each response variable. Bold values indicate significant differences ( $\alpha = 0.05$ ).

Response Variable	Species	Site Type	City	City*SiteType	Variance Structure
Soil Nutrients RC1		18.60	4.79	10.94	Species and Site Type
		<b>0.0002</b>	<b>0.02</b>	<b>0.0003</b>	
Soil Nutrients RC2	4.69	0.092	4.78	5.63	
		<b>0.04</b>	<b>0.02</b>	<b>0.009</b>	
Soil Heavy Metals RC1		9.47	17.74	3.99	Species and Site Type
		<b>0.004</b>	<b>&lt;.0001</b>	<b>0.03</b>	
Soil Heavy Metals RC2		25.66	17.55		Site Type
		<b>&lt;.0001</b>	<b>&lt;.0001</b>		

forest patch sites were consistently warmer than nearby reference forest sites during the day and night over the three years of the study. The largest air temperature differences were found between New York City's urban and reference forest sites, while the smallest differences were found between Baltimore's urban and reference forest sites. In general, maximum daily temperature differences were slightly larger than minimum daily temperature differences between urban and reference sites of each city. Urban-reference temperature differences tended to increase during the growing season and were generally greater during the summer months (June–September) than in spring (April–May).

#### 3.2. Soils

Clear relationships were discernible in the soil nutrients and heavy metals PCAs for the three cities and two site types (Figs. 2a&b). In the soil nutrients PCA, 68% of the variation was explained by the first two varimax rotated components, with RC1 accounting for 40% of the variation and RC2 accounting for 28%. Positive loadings on RC1 corresponded most strongly to Ca, Mg, pH and % sand, whereas positive loadings on RC2 corresponded to SOM, N, and P (Fig. 2; Table S3). Inspection of the scatterplot for soil nutrients RC1 and RC2 showed reference sites predominantly associated with more negative loadings of RC1 (i.e., lower concentrations of Ca, Mg, sand and pH) than urban sites. In particular, Baltimore reference site soils appear to be high in clay content and low in nutrients, consistent with previous findings from this site (Table S4; Groffman et al., 2006). Generalized least squares (GLS) models revealed differences in soil nutrients RC1 and RC2 by city and site type to be significant (Table 2). Species by site type interaction effects were excluded from all four models. A significant city \* site type interaction effect revealed higher values of soil nutrients RC1 (Ca, Mg, sand and pH) associated with NYC and Baltimore urban sites compared to reference sites, whereas there was no difference between Philadelphia urban and reference sites. In the GLS for soil nutrients RC2, there was again a significant city by site type interaction effect, where Baltimore urban sites had greater SOM, N, and P compared to reference sites, but NYC urban and reference sites showed the opposite trend, and Philadelphia urban and reference sites were not different from one another. The GLS for soil nutrients RC2 also had a significant species effect, with greater values of RC2 (positive loadings of SOM, N, and P) under white oak trees compared to soils under red maples.

In the soil heavy metals PCA, 66% of the variation was explained by the first two varimax rotated components, with RC1 accounting for 45% of the variation and RC2 21%. Positive loadings on RC1 corresponded most strongly to Na, Cd, and Zn, whereas positive loadings on RC2 corresponded to Cr, Pb, and Cu (Table S3). RC1 and RC2 of the soil heavy metals PCA largely separated reference from urban sites, with reference sites clustering to the negative ends of both axes, corresponding to lower amounts of heavy metals. Overall, urban soils had higher concentrations of heavy metals than reference soils (Table S4). A

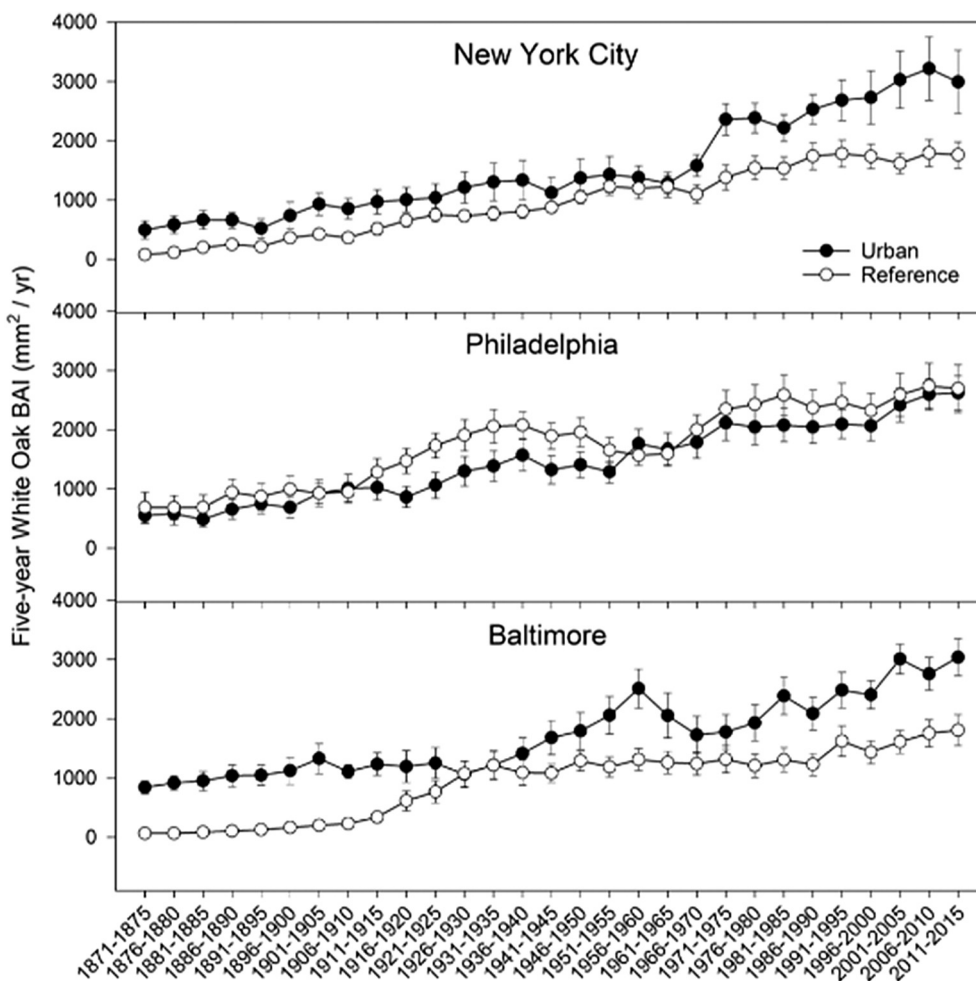


Fig. 3. Mean five-year basal area increment for white oak trees in urban and reference forest patches of New York City, Philadelphia, and Baltimore. Error bars show mean  $\pm$  SEM.

significant city by site type interaction effect in the GLS model for soil heavy metals RC1 showed a clear separation of urban and reference soils in NYC, that was lacking for the other cities (Fig. 2b, Table 2). In the GLS model for soil heavy metals RC2, heavy metal concentration was consistently higher in urban soils of all three cities with no significant interaction effect.

### 3.3. Tree growth

Tree diameters for white oak ranged between 38 and 153 cm (mean  $\pm$  SE: 69.9  $\pm$  2.3 cm) and red maple ranged between 25 and 91 cm (46.5  $\pm$  1.5 cm) (see Table S1). For all tree populations investigated except NYC red maples, there was a trend of increasing BAI with time (Figs. 3, 4). White oak BAI was generally greater than red maple BAI, with larger differences between species than between site types from 1935 to 1970 (Fig. 5). However, red maple BAI at urban sites has increased to the level of reference white oak BAI in the twenty-first century, while urban white oak and reference red maple growth currently have the greatest and least BAI, respectively.

Across the entire period, white oak BAI was 42.5% greater in urban forest patches than reference forest patches across all cities combined. Although NYC and Baltimore white oaks showed this pattern consistently over time, Philadelphia white oaks showed the opposite trend, leading to a significant city by site type interaction effect (Table 3, Fig. 3). These differences by city and site type also varied over time, as indicated by the significant three-way interaction. Beginning around 1970, BAI of urban white oaks in NYC increased suddenly compared to

reference BAI. In Baltimore, reference white oak BAI increased starting in 1915 and continued until 1930 before leveling out; urban BAI had a similar increasing trend that didn't start until 1940 and continued to 1960.

White oak LWR also had a significant three-way interaction effect but did not differ significantly among site types (Table 3, Fig. 6). In Baltimore and New York City, reference LWR was initially greater, but this trend was not apparent after 1940 and 1970 in Baltimore and New York City, respectively. In Philadelphia, mean urban LWR is greater than reference LWR after 1950.

Red maple BAI did not vary significantly by site type but did show a significant three-way interaction effect (Table 3, Fig. 4). As with the white oaks, each city showed different urban and reference red maple growth trajectories over time. Both Baltimore and Philadelphia red maples had greater reference BAI initially, with a subsequent switch to greater urban BAI, which occurs during the 1960s and 1990s in Baltimore and Philadelphia, respectively. NYC red maples showed the opposite pattern, with urban BAI greater than reference BAI until the late 1960s.

## 4. Discussion

### 4.1. Current environmental conditions of urban forest patches

Throughout the growing season, the urban sites in our study were consistently warmer during the day and night. Thus, urban forest patches in all three cities show evidence of an UHI effect, although they

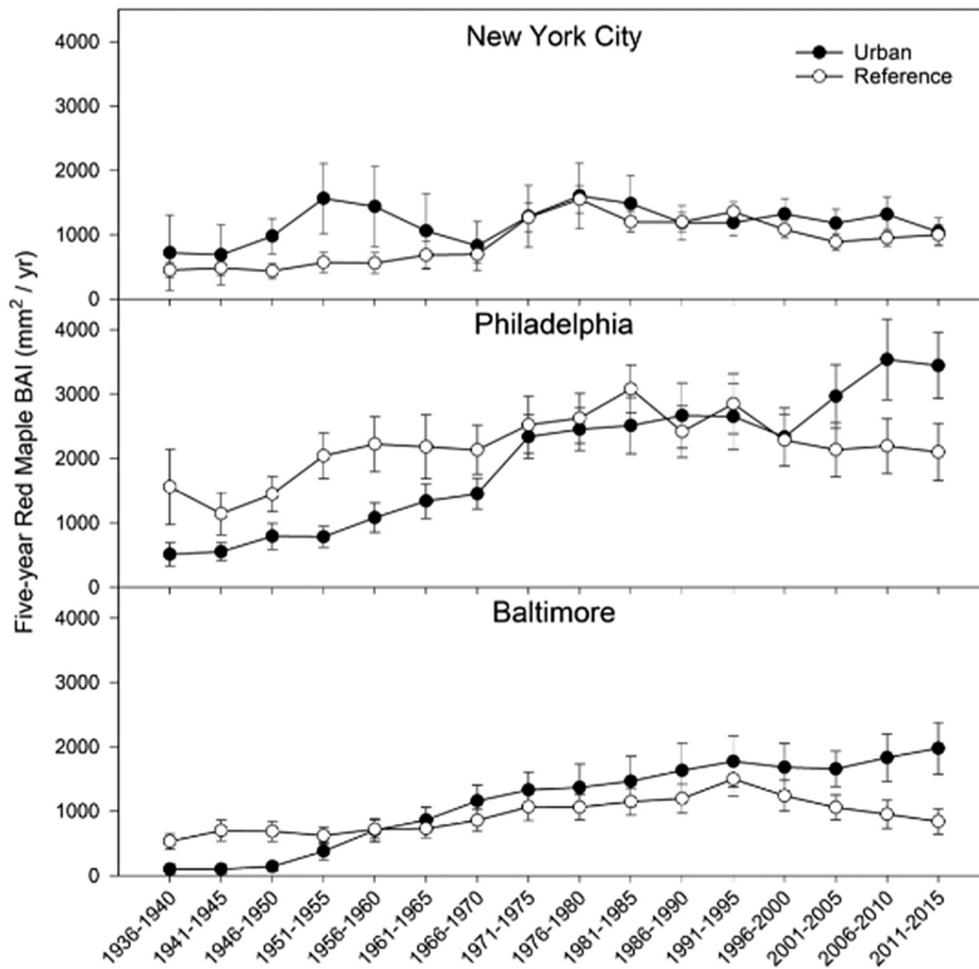


Fig. 4. Mean five-year basal area increment for red maple trees in urban and reference forest patches of (A) New York City, (B) Philadelphia, and (C) Baltimore. Error bars show mean  $\pm$  SEM.

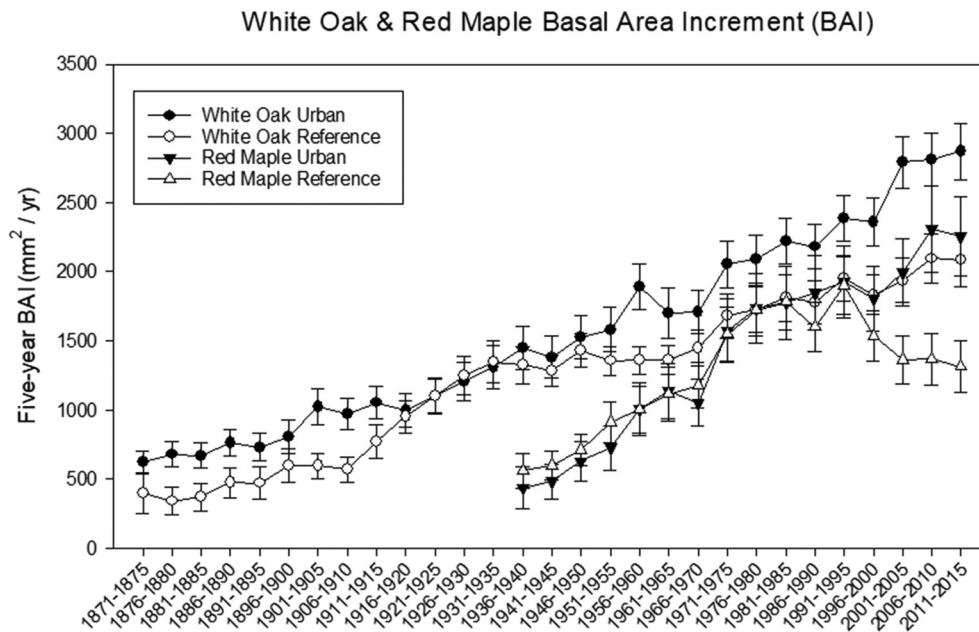


Fig. 5. Mean five-year basal area increment for white oak and red maple trees in urban and reference forest patches (includes data from NYC, Philadelphia, and Baltimore sites). Error bars show mean  $\pm$  SEM.

**Table 3**

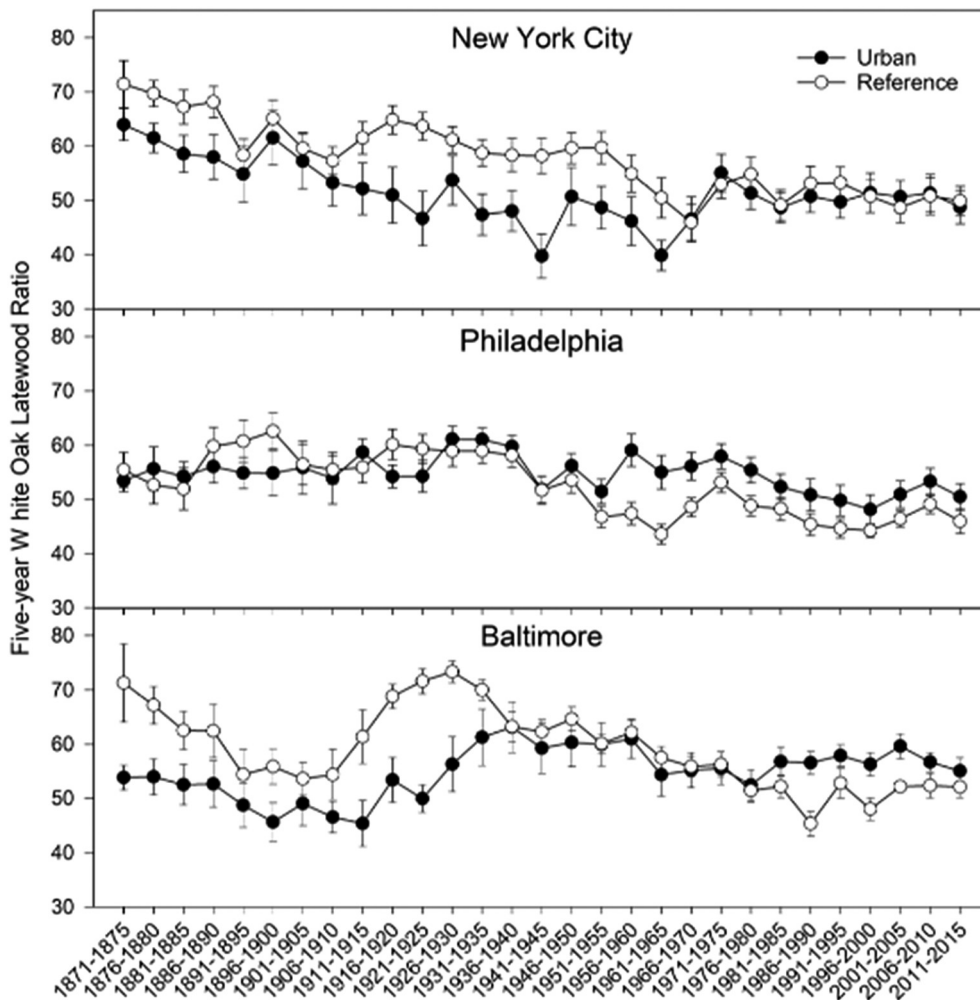
Results from linear mixed effects models analyzing the effects of city and site type and their interactions on basal area increment (BAI) of white oak and red maple trees and latewood ratio (LWR) of white oak trees over time. F-values and p-values (bold) are listed for each fixed effect and bold values indicate significant differences ( $\alpha = 0.05$ ).

Response Variable	Site Type	City	Time	City* Site Type	Time* City	Time* Site Type	Time*City*Site Type	Variance Structure
White Oak BAI	6.59	3.76	15.20	4.08	3.76	2.82	2.31	City and Site Type
	<b>0.01</b>	<b>0.03</b>	<b>&lt; 0.0001</b>	<b>0.02</b>	<b>&lt; 0.0001</b>	<b>&lt; 0.0001</b>	<b>&lt; 0.0001</b>	
Red Maple BAI	0.39	9.04	10.83	0.32	2.48	2.05	1.52	City and Site Type
	0.54	<b>0.0003</b>	<b>&lt; 0.0001</b>	0.73	<b>&lt; 0.0001</b>	<b>0.01</b>	<b>0.04</b>	
White Oak LWR	0.50	4.52	13.05	2.56	3.75	3.27	1.73	City
	0.48	<b>0.01</b>	<b>&lt; 0.0001</b>	0.08	<b>&lt; 0.0001</b>	<b>&lt; 0.0001</b>	<b>0.0007</b>	

may remain cooler than the surrounding urban matrix (Heisler et al., 2016; Scott et al., 2017). The UHI effect observed in these forest patches is less than that observed in previous urban-rural gradient studies from NYC and Baltimore, where increased plant growth was at least partially attributed to elevated air temperatures monitored in a downtown location (George et al., 2009; Searle et al., 2012). We did not find as large of a difference in nighttime air temperatures between urban and reference sites as previous studies, likely because the forest patches provide a buffer from the release of heat stored by urban surfaces warmed by daytime solar energy (Rosenzweig et al., 2005).

Soil characteristics also varied between urban and reference forest patches, but the results were less consistent across the three cities. In Baltimore, urban forest patch soils tend to have more nutrients (Ca, Mg, N, P, K) and SOM than reference soils (Table S4). These differences may

indicate that soils of Baltimore's urban forest patches are better able to support tree growth than nearby reference forest sites, which may be deficient in Ca, Mg, and P (Amacher et al., 2007). Although Baltimore's urban forest patch soils are sandier, the higher SOM may increase their water holding capacity and nutrient supply. Furthermore, our Baltimore reference site is particularly low in nutrients and high in clay content. However, the difference in pH between Baltimore reference and urban soils (3.8 vs. 4.0) may not be large enough to have a meaningful impact on nutrient availability or tree growth. In NYC, the difference between urban and reference soil nutrients is more nuanced. Reference forest soils may be deficient in Ca and Mg (Amacher et al., 2007), but have more N, P, and SOM than urban forest patch soils. There is also more sand in NYC urban soils and the pH is 4.3 compared to 3.1 in reference soils. This result is not surprising given the potential of construction fill and debris to contribute sand and concrete to



**Fig. 6.** Mean five-year latewood ratio for white oak trees in urban and reference forest patches of New York City, Philadelphia, and Baltimore. Error bars show mean  $\pm$  SEM.



urban forest soils, which may lead to Ca deposition from cement dust (Pregitzer et al., 2016). Finally, Philadelphia urban and reference forest soils are not well separated on either axis of the soil nutrients PCA. All sites except for one Philadelphia urban forest patch had soil C:N ratios below the threshold indicating the onset of nitrification (Ollinger et al., 2002). Although the soils sampled in this study are likely to capture impacts of urban deposition, we recognize that tree roots access much deeper soils, which may have different chemical and physical properties.

Urban forest soils tended to have higher heavy metal concentrations than reference forest soils. This was particularly true in NYC, where urban soils had higher values on both axes of the heavy metals PCA (Fig. 2). According to Amacher et al.'s (2007) forest soil quality index, urban forest soils of all cities had toxic levels of Cu and Pb, and NYC urban forest soils had toxic levels of Na and Zn and moderate levels of Cd. High levels of soil Na may be a result of road de-icing salt, while sources of Cu, Pb, and Zn include nearby vehicle traffic and industrial activity (Amacher et al., 2007). It is possible that trees in these urban forest patches may suffer adverse physiological effects from these pollutants, but there is little available research relating vegetation response to concentrations of plant available heavy metals (Amacher et al., 2007). Also, urban forests have been shown to persist on brownfield sites with much higher concentrations of heavy metals than those found in this study (Gallagher et al., 2008).

#### 4.2. Forest patch tree growth

The overall trend of increasing BAI found here is consistent with the findings of Johnson and Abrams (2009), where radial growth slowed in older trees but BAI continued to increase over time. Phipps and Whiton (1988) also found that BAI of individual canopy white oak trees increases at a nearly constant rate with increasing age. Looking at average growth across all cities, both urban white oak and red maple BAI increase more sharply during the period of study than BAI of reference trees (Fig. 5). Despite differences in the species' ecophysiology, both are able to grow more rapidly in the urban environment than at nearby reference sites, particularly in recent decades. This is true for both isohydric red maples and anisohydric white oaks, despite concerns that climate change and the urban heat island effect may create increasingly stressful drought conditions for urban trees. Although white oak has the reputation of being a slow-growing tree, the white oak trees in this study had greater overall BAI, particularly at the urban sites (Fig. 5). While white oak is generally considered to grow more slowly than red maple (Burns and Honkala, 1990), others have found that white oak may grow faster than red maple depending on site type or environmental conditions (Hanson et al., 2001; Quigley, 2004).

We also found variation in the growth rates of urban vs. reference forest trees depending on species and city. Urban white oak trees of NYC and Baltimore have had consistently higher BAI than reference trees, indicating that the elevated levels of heavy metals at these sites are either not impacting tree growth or the impacts are balanced by the increased availability of nutrients, warmer air temperatures, and other urban environmental conditions. By contrast, white oaks from Philadelphia urban forest patches do not have greater BAI than reference trees and there was also no difference in soil nutrients between Philadelphia's urban and reference forest soils. The forests at the Philadelphia urban sites appear to be older than those in NYC or Baltimore based on the age of the trees (see Table S1), and so the soils may have been less disturbed over time, leading to more similarity with relatively undisturbed reference forests of the region.

Over the almost 150-year white oak tree ring record analyzed, we did find some fluctuations in urban vs. reference BAI for each city. There is a sudden increase in NYC urban white oak BAI relative to reference trees that begins around 1970. In Baltimore, reference white oak BAI shows a sudden increase beginning in the 1920s, followed by a subsequent increase in urban BAI beginning in the 1940s. These periods of change do not appear to reflect changes in regional temperature (NOAA GHCN, 2018), nor in population growth (U.S. Census Bureau, 2009–2013), which may be associated with environmental changes caused by urban development. Because of the age of these cities, they have experienced the urban heat island effect for

over a century due to a long history of development (Brazel et al., 2000; Gaffin et al., 2008). Historical precipitation records from NYC indicate that there is an increase in annual precipitation over time which appears particularly pronounced after 1970 when urban white oak BAI also increases relative to reference white oak BAI (Fig. S1). However, this same increase in annual precipitation is not observed at the reference site. Historical precipitation records do not exist for both urban and reference sites in Baltimore or Philadelphia, so it is difficult to determine whether they may be similarly affected by changes in precipitation. A sudden increase in oak growth may also be due to a "release" effect when thinning or other disturbances to the forest make additional resources available for the remaining trees. In New York City, some of these forests experienced understory fires as a result of arson during the 1970s, which may have led to a release of the remaining white oak trees. The rapid increase in Baltimore reference white oak BAI is driven by a few individuals which may have been similarly released due to nearby disturbances. In addition, there is more variation in NYC urban white oak BAI compared to reference values (Fig. 3), which may be a result of heterogeneous patterns of disturbance across the urban landscape.

Surprisingly, white oak LWR does not follow the same pattern as BAI over time. In the early part of the NYC and Baltimore tree ring records, urban LWR is lower than reference LWR, despite higher levels of BAI. Differences in annual precipitation may account for the higher LWR at the NYC reference site compared to the urban sites (Fig. S1). Around 1970, annual precipitation in NYC increases to become more similar to reference site conditions, with a resulting increase in urban LWR to levels similar to that of the reference site. Higher white oak BAI despite lower LWR prior to 1970 suggests that the enhanced urban tree growth in NYC is temperature-driven and occurs despite the drier urban conditions. Warmer springs at the NYC urban sites may be driving higher BAI and lower LWR. However, once annual precipitation begins to increase, urban white oaks can increase their BAI even more relative to reference forest trees.

In the Baltimore region, and to a lesser extent in Philadelphia, urban LWR has also increased over time relative to reference site LWR. Changes in precipitation may also explain variation in Baltimore LWR relative to BAI over time if urban precipitation has increased over time relative to reference site precipitation, and warmer spring temperatures may explain higher BAI and lower LWR at urban sites. Besides providing insight into the mechanisms of changes in growth rates between urban and reference trees, LWR has additional implications for biomass and carbon storage of urban trees. Ring-porous trees with higher LWR will have greater wood density (Souza et al., 2018), which may affect modeled estimates of urban tree ecosystem services such as carbon sequestration (Westfall et al., 2019). Changes in LWR over time reveal that the white oak trees in these urban forest patches once produced rings with lower wood density than trees in nearby reference forest sites. However, this trend has changed over time and urban white oak trees now appear to be producing wood with a density that is equivalent to or greater than nearby rural trees.

Different from the white oaks, urban red maple BAI increases over time in both Philadelphia and Baltimore relative to reference site BAI, with increasingly large differences since 2000. Elevated heavy metal concentrations in urban vs. reference sites are not likely to explain these differences in growth rates given that red maple is not known to be particularly sensitive to heavy metals (Heale and Ormrod, 1982). In addition, since the urban heat island effect predates the red maple tree ring records studied here, it is not clear that the relative increase in urban vs. reference BAI would be due to changes in air temperature alone. However, it is possible that regional trends of increasing temperatures and flashier precipitation events caused by climate change may be adversely affecting reference forests, particularly in Baltimore where the reference site has lower SOM than the urban forest patch soils.

Regional and local air pollution may also be a factor influencing urban vs. reference tree growth, with varying outcomes depending on species and city. Red maple shows increased growth and survival rate under nitrogen deposition across the northeastern and north-central U.S. during the 1980s and 1990s, while white oak does not show any significant response in

growth or survival (Thomas et al., 2010). However, previous research has found an increase in nitrogen deposition in NYC compared to the surrounding area, but not in the Baltimore region (Lovett et al., 2000; Ziska et al., 2004). White oak does appear to show growth decline due to regional air pollution (Long and Davis, 1999; Davis and Long, 2003), but it is not clear that nitrogen or sulfur oxide pollution would vary between the urban and reference sites of each city. Previous studies have found increased rates of plant growth in both NYC and Baltimore, with these trends attributed to lower levels of ozone exposure or elevated air temperature in NYC (Gregg et al., 2003; Searle et al., 2012) and to elevated temperature and atmospheric CO<sub>2</sub> in Baltimore (Ziska et al., 2004; George et al., 2009). White oak is insensitive to ozone while red maple is only moderately sensitive (Davis and Skelly, 1992), making this factor unlikely to cause the differences in tree growth found here. Each city may have different environmental conditions that cause increased growth rates of some tree species compared to rural conspecifics, while other species are not affected.

Previous research on open grown urban tree growth shows varying results by species and city (Iakovoglou et al., 2002; Quigley, 2004; Monteiro et al., 2017). However, trees in urban forest patches may not suffer the same drought stress as open grown urban trees, possibly leading to more benefits than drawbacks of urban environmental conditions. Research from rural forests has found that tree growth does not necessarily increase as expected with increased atmospheric CO<sub>2</sub> and resulting increases in water use efficiency, if growth is constrained by nutrient limitation or drought conditions (Oren et al., 2001; Peñuelas et al., 2011). If urban forest patch trees experience greater nutrient or water availability than trees in nearby reference forests, they may be able to take advantage of the elevated temperatures and CO<sub>2</sub> in urban areas in a way that trees from rural forests are not. Although this study addresses growth rates of mature canopy trees, there is evidence that urban environmental and edaphic conditions can enhance seedling growth as well (Searle et al., 2012; Sonti, 2019). However, urban environmental factors may simultaneously stimulate tree growth and reduce longevity, since fast growing trees are less likely to obtain maximum longevity for the species (Johnson and Abrams, 2009).

## 5. Conclusions

Many aspects of the urban environment have the potential to affect tree growth and physiology in forest patches, including higher temperatures, altered CO<sub>2</sub> and O<sub>3</sub> concentrations, and modified soil biogeochemistry from increased nutrient and heavy metal inputs. However, trees in urban forest patches do not experience the same conditions as trees in more heavily developed or managed urban sites. It appears that soil conditions and elevated air temperature may contribute to enhanced BAI of urban forest patch trees compared to conspecifics of nearby reference forests. In addition, changes in precipitation over time may alter the relative production of earlywood and latewood, with implications for both growth rates and wood density of urban trees. However, urban vs. reference growth rates vary by species, by city, and over time, demonstrating that it is necessary to study multiple tree species across different urban areas in order to draw conclusions about urban tree physiology.

In this study, we chose to examine the growth of two important native tree species of the eastern U.S. White oak is considered to be a slow growing tree species but can persist on a wide range of sites and does well with low level disturbance, tolerating understory fire and drought conditions (Abrams, 2003). The shade tolerant red maple is more fire sensitive but is a generalist species that has low resource requirements and also does well in a wide variety of conditions (Abrams, 1998). Perhaps due to their ability to persist in a wide variety of environmental conditions, mature trees of both species appear to be acclimating to urban forest patch conditions of the eastern U.S. In fact, contrary to the paradigm of cities as a stressful environment for native plants, we observed enhanced growth rates in urban forest patch trees compared to trees in nearby reference forests. The continued growth of white oak and red maple trees in urban forest patches will ensure the provision of valuable ecological habitat and important ecosystem services for urban residents in the decades to come. However, the

regeneration of these native trees in urban areas is being impeded by both a lack of natural disturbance processes and increasing pressure from deer browse and invasive plants, leading to even-aged urban forest stands lacking recruitment of new canopy trees (Simmons et al., 2016; Pregitzer et al., 2019). Forestry and silvicultural management practices may be used to help address these issues so that urban forest patch tree growth and productivity continue to provide social and ecological benefits to urban communities.

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## Appendix A. Supplementary data

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