

ABSTRACT

Title of Dissertation: ECOPHYSIOLOGICAL AND SOCIAL
 FUNCTIONS OF URBAN FOREST
 PATCHES

 Nancy Falxa Sonti, Doctor of Philosophy, 2019

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 Science and Landscape Architecture

Urban trees provide many ecosystem services to cities: alleviating the urban heat island effect, absorbing stormwater runoff, and contributing to residents' social and psychological well-being. The production of these benefits is influenced by tree growth and physiological function within the urban ecosystem, and also by the social-ecological context in which urban forest patches exist. This dissertation investigates the ecophysiological and social functions of urban forest patches of the eastern United States using a multidisciplinary approach that combines diverse empirical methods across varied timescales and geographies.

Using data collected from urban and reference forest patch sites, this dissertation begins by addressing the following questions: How does native tree growth and physiology vary between urban and reference forest patches? Are there differences in

ecophysiological responses by tree species and by city within the eastern United States? Air temperature and soil data from each field site are analyzed alongside tree ring and leaf-level physiological data. Next, results from a controlled growth chamber experiment are presented to examine how Baltimore's urban vs. reference forest soils and air temperature interact to affect seed germination and seedling growth of white oak (*Quercus alba*), a dominant species in the region. Finally, the social functions of forest patches are investigated using qualitative data from semi-structured interviews conducted with Baltimore residents.

Overall, urban forest patches were found to support robust growth and physiological function of white oak (*Quercus alba* L.) and red maple (*Acer rubrum* L.) trees, with differences by species and site type (urban vs. reference), and over time. In particular, urban soils appear to support greater biomass and photosynthesis rates than reference soils. Regardless of the favorable ecophysiological conditions of urban forest patches, community awareness and engagement with these sites will be critical to their continued protection and management. Qualitative interview data revealed local residents' strongly ambivalent attitudes towards urban wilderness, with only limited differences by homeownership and property management regime. In sum, dissertation documents important ecophysiological and social functions of urban forest patches, with implications for the continued provision of benefits to urban and rural communities.

ECOPHYSIOLOGICAL AND SOCIAL FUNCTIONS OF URBAN FOREST
PATCHES

by

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Dedication

I dedicate this work to the memory of my grandparents, Katie Raymond, George Raymond, Marjorie Falxa, and John Falxa, Sr., for teaching me to love learning and to get my hands dirty in the great outdoors. And to Rohak, for the adventures ahead.

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Chapter 1: Introduction

Urban forests provide many benefits to local residents: alleviating the urban heat island effect, improving water quality, and contributing to community members' social and psychological well-being. Production of these benefits is influenced by tree growth and physiological function within the urban ecosystem, and also by the social-ecological context in which urban forest patches exist. As cities around the world invest millions of dollars into tree-planting initiatives (Oldfield et al. 2013; Pincetl 2013; Campbell 2014), it is important to take a comprehensive social-ecological approach to understanding urban forest ecology.

Urban forest patches are woodlands contained within a city, whether completely surrounded by urban development or on the urban fringe (Kowarik 2005). These forests may 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 eastern United States (U.S.), where cities have generally replaced temperate deciduous forest, the minimal understory 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. 2018). Urban woodlands and other natural areas have been categorized in a variety of ways, based primarily on land use history, vegetation type, and the extent to which they are actively managed. (Kowarik 2005, 2018; Threlfall and Kendal 2016). Patches of these different types of forest exist within a complex urban social-ecological system (Pickett et al. 1997, 2001). In this system, urban natural resources influence and are

influenced by socioeconomic and cultural resources, as well as social institutions, cycles, and aspects of social order inherent in any society (Machlis et al. 1997).

Urban forest patches are part of the mosaic of urban land cover types, and they provide many important ecosystem services, as well as habitat for humans and non-human species. Biophysical ecosystem services include carbon sequestration, stormwater regulation, nutrient cycling, and temperature regulation, for which it is important to understand tree physiological processes like carbon fixation, evapotranspiration, and nitrogen uptake (Livesley et al. 2016). Equally important socio-cultural ecosystem services are provided by urban forest patches. However, these benefits of urban natural areas are not as well-documented, particularly in areas that are not formally managed as parkland. Such socio-cultural ecosystem services may include spiritual and religious values, inspiration, aesthetic values, social relations, sense of place, and recreation (MEA 2005). In this dissertation, I investigate the ecophysiological and social functions of urban forest patches through: (1) tree physiology and soils data collected from field sites in New York, NY, Philadelphia, PA, and Baltimore, MD; (2) a growth chamber experiment based on urban and rural soils and air temperatures of Baltimore, MD; and (3) in-depth qualitative field research in communities surrounding Baltimore forest patches.

1.1 Urban Forest Ecology and Urban Tree Ecophysiology

Urbanization is an important force of environmental change impacting ecosystem functioning on continental and global scales. Forests embedded within an

urban matrix are directly affected by environmental impacts associated with urban land use, and thus provide a useful context for investigating the effects of these anthropogenic factors on forest ecosystems (Groffman et al. 2006). Habitat destruction, invasive species, air and water pollution, and the urban heat island effect are all associated with increased urbanization (Oke 1989; Nowak 2000; Baxter 2002). These processes impact the climate, soil biogeochemistry, and biotic communities of urban forest patches, potentially affecting tree growth and physiology in these urban ecosystems (Pouyat and McDonnell 1991; Groffman et al. 2006; Nowak et al. 2006; Pavao-Zuckerman 2008). In addition, these environmental factors are sure to impact forest tree species differently according their inherent growth strategies, nutrient use, and tolerance for environmental stress. However, few comparative investigations have been undertaken.

Soil biogeochemical cycles are driven by hydrology, atmospheric chemistry, climate, nutrients, vegetation composition, and land use (Kaye et al. 2006). Human activity may impact all of these drivers, with implications for the soil properties of urban forest patches. Cities are sources of gaseous sulfur and nitrous oxides resulting from fossil fuel combustion, and are also sources of atmospheric dust arising from roadways and from excavation, construction, and demolition activities (Lovett et al. 2000). Urban forests may be receiving excess deposition of nutrients and pollutants, resulting in more nitrogen and less acidity than nonurban forests (Lovett et al. 2000). 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 differences in species composition in both plant and soil communities are ultimately stronger determinants of soil nitrogen dynamics than urban atmospheric conditions. Similarly, Pouyat et al. (2008) found 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 is dynamic, complex, and difficult to predict.

Cities also experience elevated temperature and CO₂ compared to surrounding areas. As a result, urban systems have been proposed as laboratories to study climate change (Ziska et al. 2003; Youngsteadt et al. 2014). The urban heat island effect has been documented in multiple cities, including Baltimore, Philadelphia, and New York City (Brazel et al. 2000; Rosenzweig et al. 2005; Gaffin et al. 2008). Gaffin et al. (2008) found that the urban heat island effect was responsible for approximately one third of the total warming of New York City over the twentieth century. This warming is also associated with a drop in windspeed that likely occurred as the city's skyscrapers created a larger boundary layer. However, most urban forest patches are not located near downtown weather stations, and urban heat island effects vary across the urban landscape depending on land use context (Gaffin et al. 2008; Heisler et al. 2016; Scott et al. 2017). Atmospheric CO₂ and ozone gradients of varying strengths

have also been documented in Baltimore and New York City (Gregg et al. 2003; George et al. 2007; Hsueh 2009).

Despite the attention paid to urban climate and soil biogeochemistry, there is relatively little research examining the physiological response of plants to the urban environment compared to reference ecosystems outside of urban areas (Cadenasso et al. 2007; Calfapietra et al. 2015). Gregg et al. (2003) found hybrid poplars (*Populus deltoides*) to grow faster in New York City than surrounding rural areas, attributing the difference to ozone effects. Similarly, Searle et al. (2012) found that New York City's urban environment promoted northern red oak (*Quercus rubra* L.) seedling growth, while Rahman et al. (2014) found urbanization to increase tree growth, but not LAI in Callery pear trees (*Pyrus calleryana* Decne). In Baltimore, George et al. (2009) measured greater aboveground plant biomass and faster rate of secondary succession from fallow soil at urban compared to rural sites. Native tree growth, health, and nitrogen use have been found to vary across different New York City forest soils (Falxa-Raymond et al. 2014; Pregitzer et al. 2016). In addition, increased leaf nitrogen and respiration rates in New York City northern red oak trees compared to rural trees suggest the interaction between temperature and biogeochemistry (Searle et al. 2011).

The altered soil characteristics and air temperature of cities may have significant impacts on tree growth in urban forest patches, but it is unclear whether these impacts will be similar across multiple urban areas. The differences between urban and reference forest conditions across metropolitan areas may cause changes in tree growth and physiology that are similar in magnitude and direction. However,

urban forest patch conditions of different regions vary according to the unique sets of interacting human and biophysical variables at each location, including latitudinal gradient, physiography, and land use history. A greater understanding of these ecophysiological processes will provide insight into carbon, water, and nutrient dynamics as large scale urban tree planting initiatives become increasingly widespread and the impact of their changes to the urban landscape is borne out (Campbell 2014).

1.2 Social Functions of Urban Forest Patches

Urban green spaces, including parks and vacant land, have been framed as sources of socio-cultural ecosystem services (Bolund and Hunhammar 1999; McPhearson et al. 2013), including recreation, social relations, and sense of place (Campbell et al. 2016). However, there is limited research focusing specifically on urban woodlands and other natural areas. Perceptions of and engagement with natural areas inside public parks have been studied using rapid survey methods combining site observations, behavioral observations, and interviews with park users in New York City (Campbell et al. 2016), Chicago (Feldman 2007), and Belgium (Roovers et al. 2002). These studies recorded many activities taking place in urban “natural areas,” including active recreation, engagement with nature, and opportunities for reflection while sitting, reading, walking, or gazing. Outside of formal parkland, the ecological and social value of urban “wildscape” fragments and large vacant areas have been described in case studies from the U.S. (Gobster 2011) and Germany (Keil

2005). A large variety of activities have been recorded in these unmanaged spaces, including active recreation and many instances of “lingering.”

The study of urban woodlands and other ‘natural areas’ outside of formal parkland has received more attention in Europe than in the U.S. Konijnendijk (2005, 2008) has reviewed the historical context of urban forests in Europe and the complications around the concepts of “natural” or “wild” urban forests as perceived by urban residents. Forest patches existing on vacant or abandoned land have been termed “urban wild woodlands” by European scholars (Kowarik and Körner 2005), and may perform ecological and social functions, though they are not viewed favorably by all urban residents. Rupprecht and Byrne (2014) have identified a typology of “informal urban greenspace (IGS)” and have reviewed the literature on these liminal spaces. The typology is based on the built environmental characteristics (street verge, retaining wall, vacant lot) rather than vegetation type, so forest patches may occur on several of them. They find that surveys and photographic methods have been most commonly used to study human-IGS interactions, and that there is a need for more qualitative approaches such as interviews and participant observation.

Much of the literature on social functions of urban forest patches and woodlands comes from the field of landscape architecture and has focused on the aesthetic experience of these green spaces rather than the social and psychological benefits they provide to urban residents. Perceptions of forest patches on urban parkland and vacant land may be tied to the presence or absence of “cues to care” that indicate whether or not the forest is actively stewarded (Nassauer 1995a). Nassauer (1995b) has stated that “the appearance of landscapes communicates cultural values”

and that “human landscape perception, cognition, and values directly affect the landscape and are affected by the landscape.” Therefore, the socio-cultural benefits of urban trees and green space may be context dependent. Forest patches are a particular type of urban green space, existing within a variety of land use, ownership, and management regimes. Urban residents may perceive and use forest patches differently from landscaped park areas or community gardens, for example. Social functions of urban forest patches may also vary depending on their particular neighborhood context and physical characteristics.

Studies have generally explored perceptions or uses of urban forests and natural areas either within formal parkland or within the context of more informal urban greenspace, but have not compared the two settings. Furthermore, although the psychosocial benefits of engaging with urban nature are well documented (Dwyer et al. 1991; Hull 1992; van den Berg et al. 2007; Barton and Pretty 2010), less attention has been paid to the ways in which these benefits are impacted by green space management regime or neighborhood characteristics such as homeownership rates.

The social benefits of urban natural areas within the U.S. are increasingly recognized by municipal governments and organizations such as Trees for Seattle and New York City’s Natural Areas Conservancy (City of Seattle 2013; Natural Areas Conservancy 2016). Although Baltimore City lacks a similarly comprehensive citywide plan to manage its natural areas, the city provides a compelling context in which to study the varying social functions of forest patches across an urban landscape. Thirty-four percent of Baltimore’s tree canopy is made up of forest patches, defined as areas of tree canopy greater than 10,000 ft² (Avins 2013). More

than half of this area exists outside of municipal park boundaries, creating a varied mosaic of neighborhood characteristics, land uses, and ownership of the city's forest patches. There is a need to document the socio-cultural functions of these spaces and the extent to which resident perceptions and interactions with forest patches vary across Baltimore neighborhoods.

1.3 Project Overview

In the following chapters I present an investigation of the ecophysiological and social functions of urban forest patches of the eastern U.S. A combination of field work across three urban areas, and a controlled growth chamber experiment are used to examine ecophysiology of native trees in urban forest patch environments compared to reference forest conditions. To understand the social functions of urban forest patches in Baltimore, Maryland, I employ semi-structured interviews with local residents. Thus, this dissertation uses a multidisciplinary approach that combines diverse empirical methods across varied timescales and geographies.

In Chapter 2 I compare growth rates of white oak (*Quercus alba* L.) and red maple (*Acer rubrum* L.) trees in urban forest patches with forested reference sites in three major cities of the eastern United States (New York, NY; Philadelphia, PA; Baltimore, MD). White oak and red maple are two ecologically important tree species which are common throughout the region and have been found to thrive in a wide variety of environmental conditions. 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 the more disturbance-adapted northern red oak and chestnut oak (*Quercus prinus* L.) (Abrams 2003). Red maple is one of the most common native species in urban and rural forests of the region and is replacing oaks in some cases (Abrams 1998, 2003). Growth rates and physiological function of both species would be expected to vary between urban and reference forest sites due to soil physical and chemical properties, nitrogen deposition, higher air temperatures, and elevated CO₂. Using tree ring data, I examine the effects of site type (urban vs. reference), city, and time on basal area increment and latewood ratio. Differences in current air temperatures and soil properties are also presented in this chapter, to provide some environmental context for the differences in tree growth at each site. In Chapter 3 I examine chlorophyll fluorescence parameters, leaf traits, and foliar nutrients in the same white oak and red maple trees across urban and reference sites of the three cities. Species differences in photosynthetic efficiency, thermal tolerance of photosynthesis, stomatal traits, and specific leaf area (SLA) across the urban and reference sites are presented and compared to foliar nutrient levels.

Building on these field studies, I then use a growth chamber experiment to examine urban soil and air temperature effects on white oak seedling growth and physiological function. In particular, I explore whether urban soil and temperature conditions of Baltimore, MD will enhance photosynthesis and growth of white oak seedlings, or whether the urban environment has negative effects on white oak physiology that may further contribute to the decline of this slow-growing and ecologically important species (Abrams 2003). The growth chamber study helps

elucidate interactions between the urban heat island effect and impacts of anthropogenic soil disturbance, using seedlings grown in soils from the same urban and reference sites as the Baltimore field study described above. Seedling biomass, foliar chemistry, chlorophyll fluorescence, and gas exchange data are used to determine rates of growth, photosynthesis, respiration, and nitrogen use efficiency across the different treatment combinations.

Finally, I examine social functions of Baltimore's urban forest patches. These woodland spaces may support physical activity, emotional and psychological restoration, sense of place, and a profound connection to the natural world. However, forest patch land management and neighborhood context may impact the provisioning of these ecosystem services across the urban landscape. Using semi-structured interviews with Baltimore residents, I compare the perception and use of forest patches across four case study neighborhoods selected for differences in homeownership and forest patch management. This approach allows me to assess the degree to which these urban green spaces are viewed by nearby residents as amenities or disamenities, as well as variation in resident perceptions of forest patches across different neighborhood and forest patch contexts. Qualitative analysis of resident attitudes and reported behaviors provides a better understanding of the ways in which the local community values and engages with formal and informal urban green space, and the positive and negative social meanings ascribed to them. In the final chapter, I present a synthesis of my results and implications for management that will ensure continued social-ecological functioning of urban forest patches into the future.

Chapter 2: White oak and red maple growth in urban forest patches

2.1 Abstract

Many aspects of the urban environment have the potential to affect tree growth and physiology in forest patches, including higher temperatures, elevated CO₂ 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, I 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). I 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

2.2 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, I 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 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, and UHI effects 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).

Soil biogeochemical cycles are driven by hydrology, atmospheric chemistry, climate, nutrients, vegetation composition, and land use (Kaye et al. 2006; Decina et al. 2017). Human activity may impact all of these drivers, with implications for the soil properties of urban forest patches. Urban air contains increased concentrations of pollutants such as CO₂, nitrogen oxides, sulfur oxides, O₃ 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). 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; Trammell 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 2002, 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 (Craib 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 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), 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 Ibanez (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 (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, I use measurements of basal area increment (BAI) and latewood ratio (LWR) to determine how the urban environments of NYC, Philadelphia, and Baltimore affect the growth of white oak and red maple trees in urban forest patches

compared to trees in nearby reference forest sites. Long-term changes in tree growth rates indicated by tree ring records can provide insight into the ways that urbanization and climate change will impact existing forests in less developed areas. Comparing long-term tree ring growth in multiple cities allows us to assess the generality of urban environmental effects versus the effects of the idiosyncratic patterns of growth and development of each city. I also examine contemporary air temperature and soil properties of the urban and reference forests, which may impact tree growth at these sites.

2.3 Methods

2.3.1 Study Area

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 (Figure 2.1). The climate of these eastern U.S. cities is strongly seasonal with warm summers and cold winters and all three cities receive ~1,100 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 trees and similarity of soil types within each metropolitan area (NRCS soil survey). All urban sites were within official city limits, and all reference sites were located in protected areas outside of the city surrounded by intermix wildland-urban interface landscapes (Martinuzzi et al. 2015). Reference ecosystems were dominated by mature native trees with minimal management by humans (Reisinger et al. 2016). 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 except for one Baltimore forest patch were on public parkland or other institutional grounds, and most were private estates prior to these current land uses. Three reference sites were selected for each city, located within the same protected area considered representative of rural forests of the region. Reference sites were all protected areas, established in during the twentieth century after extensive clearcutting for agricultural activities through the late 1800s - early 1900s.



Figure 2.1 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.

2.3.2 Data Collection

One Thermochron iButton air temperature sensor 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 (less than 25% overall canopy damage; Pontius and Hallett 2014). When there were more than five suitable trees in a given site, the trees were mapped and five were selected at random.

Four 10 cm mineral soil cores 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 CaCl₂, organic content using loss-on-ignition (LOI), and soil texture using the hydrometer method (Day 1965). Plant available elements, including Al, As, Ca, Cd, Cr, Cu, K, Mg, N, Na, P, Pb, and Zn were determined by extracting soil using a modified Morgan's solution (NH₄OAC, 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. 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 proportion of latewood contained in each tree ring (Bergès et al. 2000).

2.3.3 Data Analysis

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. 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, 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 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 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 `gls()` function in R. 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 R package 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 (Pinheiro et al. 2010). 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 2.4). Differences between means were considered significant at $\alpha = 0.05$.

2.4 Results

2.4.1 Air Temperature

Average daily minimum and maximum temperatures at the urban and reference field sites are shown in Table 2.1. Each city's urban 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).

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 (±0.14)	21.64 (±0.23)	2.95	16.97 (±0.13)	14.13 (±0.23)	2.84
Philadelphia	24.82 (±0.13)	23.56 (±0.22)	1.27	16.73 (±0.13)	15.38 (±0.22)	1.35
Baltimore	25.48 (±0.14)	24.31 (±0.22)	1.16	17.15 (±0.12)	16.35 (±0.21)	0.81

Table 2.1 Mean daily maximum and minimum air temperatures (±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.

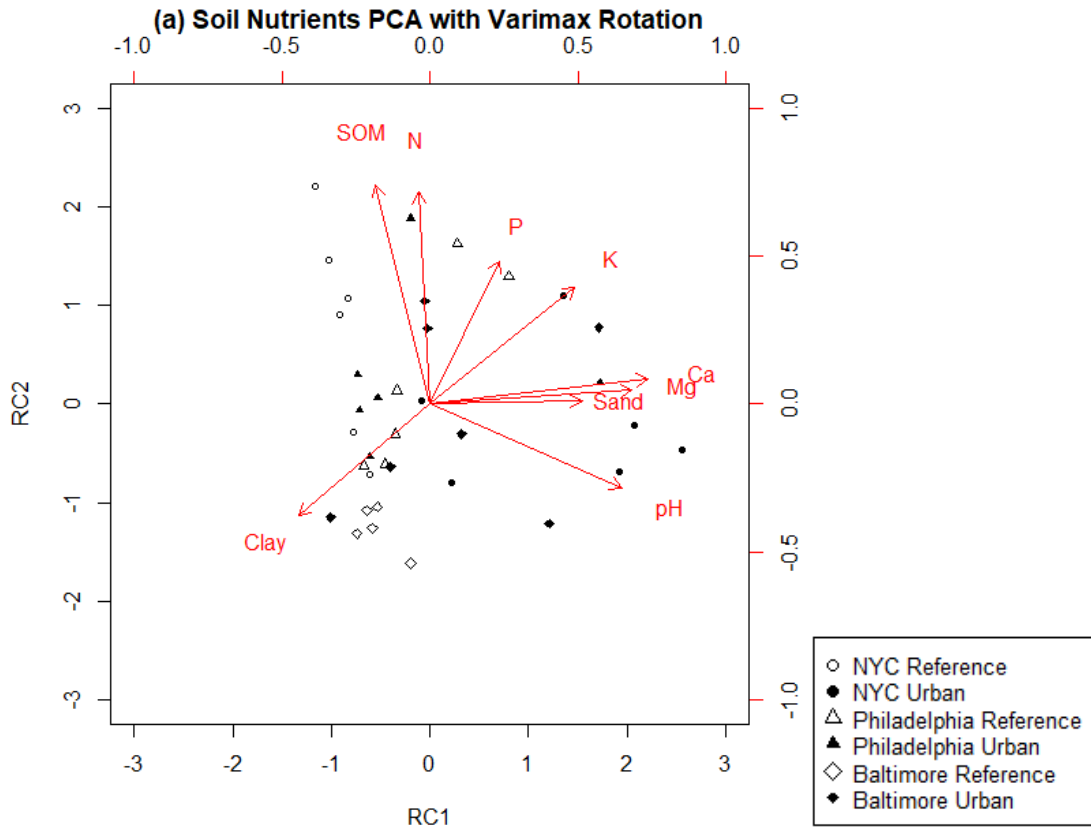
2.4.2 Soil

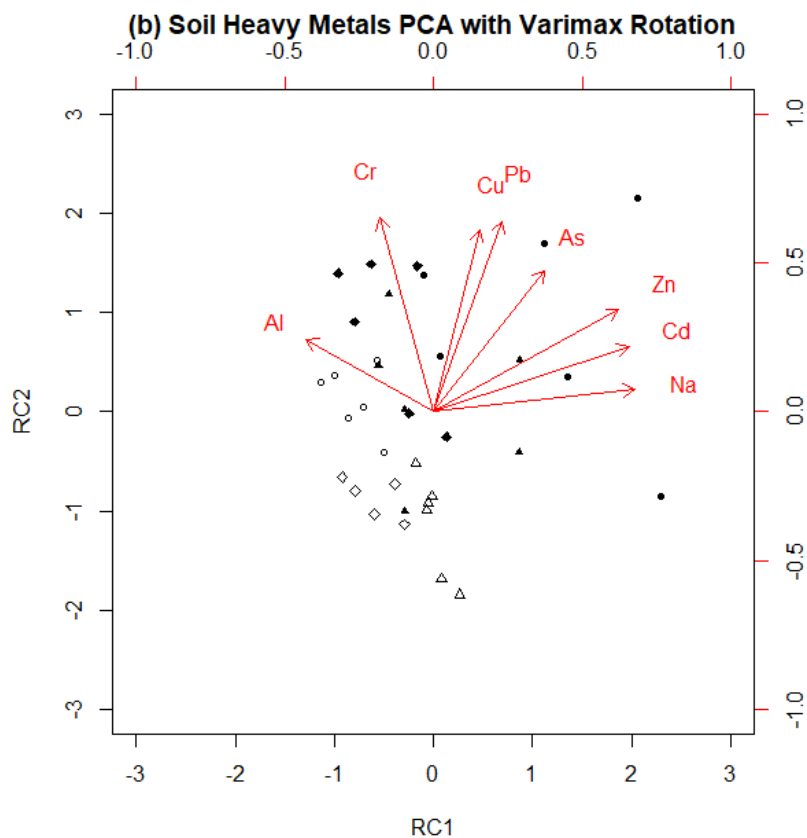
Clear relationships were discernible in the soil nutrients and heavy metals PCAs for the three cities and two site types (Figs 2.2 a&b). In the soil nutrients PCA, 68% of the variation was explained by the first two varimax rotated components, with RC1 accounting for 40% of the variation and RC2 28%. Positive loadings on RC1 corresponded most strongly to Ca, Mg, pH and % sand, whereas positive loadings on RC2 corresponded to SOM, N, and P (Figure 2.2; Table 2.2). 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 2.3; 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 2.4). Species * 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 * 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 2.3). 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 2.2). A significant city * 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 (Figure 2.2b, Table 2.4). 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.

Figure 2.2 (a) Biplot of Soil Nutrients Principal Components 1 and 2 after Varimax Rotation, **(b)** Biplot of Soil Heavy Metals Principal Components 1 and 2 after Varimax Rotation. Soils from urban forest patches are represented by solid symbols and soils from reference forest sites are represented by clear symbols; shapes are used to differentiate soils from NYC (circles), Philadelphia (triangles), and Baltimore (diamonds).





Soil Nutrients	RC1 Loading	RC2 Loading
pH	0.811	-0.354
SOM	-0.231	0.925
Sand	0.642	
Clay	-0.556	-0.468
N		0.897
P	0.294	0.605
K	0.611	0.494
Ca	0.920	0.108
Mg	0.854	
Soil Heavy Metals	RC1 Loading	RC2 Loading
Cu	0.192	0.764
Zn	0.780	0.432
Pb	0.286	0.801
Na	0.846	
Cr	-0.227	0.813
Cd	0.822	0.277
Al	-0.537	0.305

Table 2.2 Loadings for soil nutrients and heavy metals varimax rotated principal components (RC).

	pH	SOM	Sand	Clay	N	P	K	Ca	Mg
				(%)				(ppm)	
New York City									
Reference	3.14 (2.28-4.03)	13.92 (5.67-34.63)	50.24 (37.99-70.00)	21.66 (15.28-29.90)	0.37 (0.14-0.82)	17.75 (1.76-41.37)	86.34 (36.25-219.09)	73.70 (5.68-459.41)	26.44 (8.29-68.12)
Urban	4.33 (3.27-6.49)	9.18 (3.19-19.63)	59.35 (28.65-82.44)	18.03 (11.63-28.65)	0.29 (0.10-0.63)	16.89 (1.89-81.32)	119.00 (36.27-257.19)	1292.54 (33.94-7431.17)	206.12 (19.86-2007.42)
Philadelphia									
Reference	3.93 (3.06-4.76)	12.09 (6.51-27.00)	46.47 (22.22-69.59)	20.27 (10.86-31.56)	0.38 (0.19-0.90)	6.53 (0.09-35.66)	114.44 (33.96-389.63)	563.89 (36.36-4509.29)	86.11 (12.67-320.49)
Urban	3.53 (2.85-4.92)	12.08 (7.34-26.54)	51.11 (35.08-64.42)	20.83 (12.07-25.10)	0.35 (0.20-0.87)	8.50 (2.24-43.35)	124.19 (40.67-441.44)	520.57 (54.37-2160.35)	85.33 (13.8-360.94)
Baltimore									
Reference	3.75 (3.33-4.62)	7.51 (5.04-11.36)	53.18 (42.48-75.51)	23.78 (16.35-33.17)	0.19 (0.06-0.32)	3.29 (1.32-10.49)	61.00 (40.59-118.73)	120.75 (5.68-753.17)	37.23 (9.84-151.81)
Urban	4.00 (2.62-7.78)	10.31 (4.65-18.09)	53.80 (27.04-73.23)	20.07 (12.34-30.04)	0.35 (0.19-0.70)	9.53 (2.09-40.85)	124.01 (41.94-341.75)	875.22 (60.82-3834.83)	101.81 (19.09-355.33)

	Cu	Zn	Pb	Na	Cr	Cd	Al
				(ppm)			
New York City							
Reference	0.46 (0-0.96)	3.69 (0.41-16.81)	32.33 (3.99-306.45)	6.88 (2.00-14.78)	0.28 (0.07-0.51)	0.08 (0.01-0.18)	654.49 (177.56-1133.63)
Urban	1.61 (0.38-4.55)	10.89 (2.26-96.0)	52.99 (6.45-401.05)	36.66 (9.73-113.51)	0.37 (0.04-1.07)	0.15 (0.03-0.42)	291.21 (23.4-801.29)
Philadelphia							
Reference	0.16 (0-0.48)	2.49 (0.27-6.13)	9.17 (1.08-78.53)	11.27 (1.37-27.49)	0.16 (0.01-0.46)	0.09 (0.02-0.16)	262.76 (70.77-847.08)
Urban	1.21 (0.37-13.13)	5.99 (1.59-16.58)	37.98 (14.57-88.48)	9.73 (2.76-21.85)	0.28 (0.03-0.96)	0.10 (0.03-0.26)	277.75 (24.62-500.81)
Baltimore							
Reference	0.32 (0.01-0.85)	1.10 (0-3.31)	5.98 (1.99-15.34)	3.34 (0.65-6.87)	0.21 (0.04-0.91)	0.04 (0.01-0.11)	258.31 (98.65-416.39)
Urban	1.14 (0.10-3.32)	9.34 (0.94-42.97)	30.71 (7.64-87.23)	20.76 (7.06-122.1)	0.48 (0.07-1.25)	0.09 (0-0.47)	218.07 (13.81-560.89)

Table 2.3 Mean and range of soil characteristics from urban and reference forest sites in New York, NY; Philadelphia, PA; and Baltimore, MD. Soil nutrients and heavy metals are shown in parts per million (mg kg⁻¹) unless otherwise indicated.

Response Variable	Species	Site Type	City	City*SiteType	Variance Structure
Soil Nutrients RC1		18.60 0.0002	4.79 0.02	10.94 0.0003	Species and Site Type
Soil Nutrients RC2	4.69 0.04	0.092 0.76	4.78 0.02	5.63 0.009	
Soil Heavy Metals RC1		9.47 0.004	17.74 <.0001	3.99 0.03	Species and Site Type
Soil Heavy Metals RC2		25.66 <.0001	17.55 <.0001		Site Type

Table 2.4 Generalized least squares model summaries for soil nutrients and heavy metals varimax rotated principal components (RC) response variables. 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 ($\alpha = 0.05$).

2.4.3 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 92 cm (46.5 \pm 1.5 cm). For all tree populations investigated except NYC red maples, there was a trend of increasing BAI with time (Figures 2.3, 2.4). White oak BAI was generally greater than red maple BAI, with larger differences between species than between site types from 1935-1970 (Figure 2.5). 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.

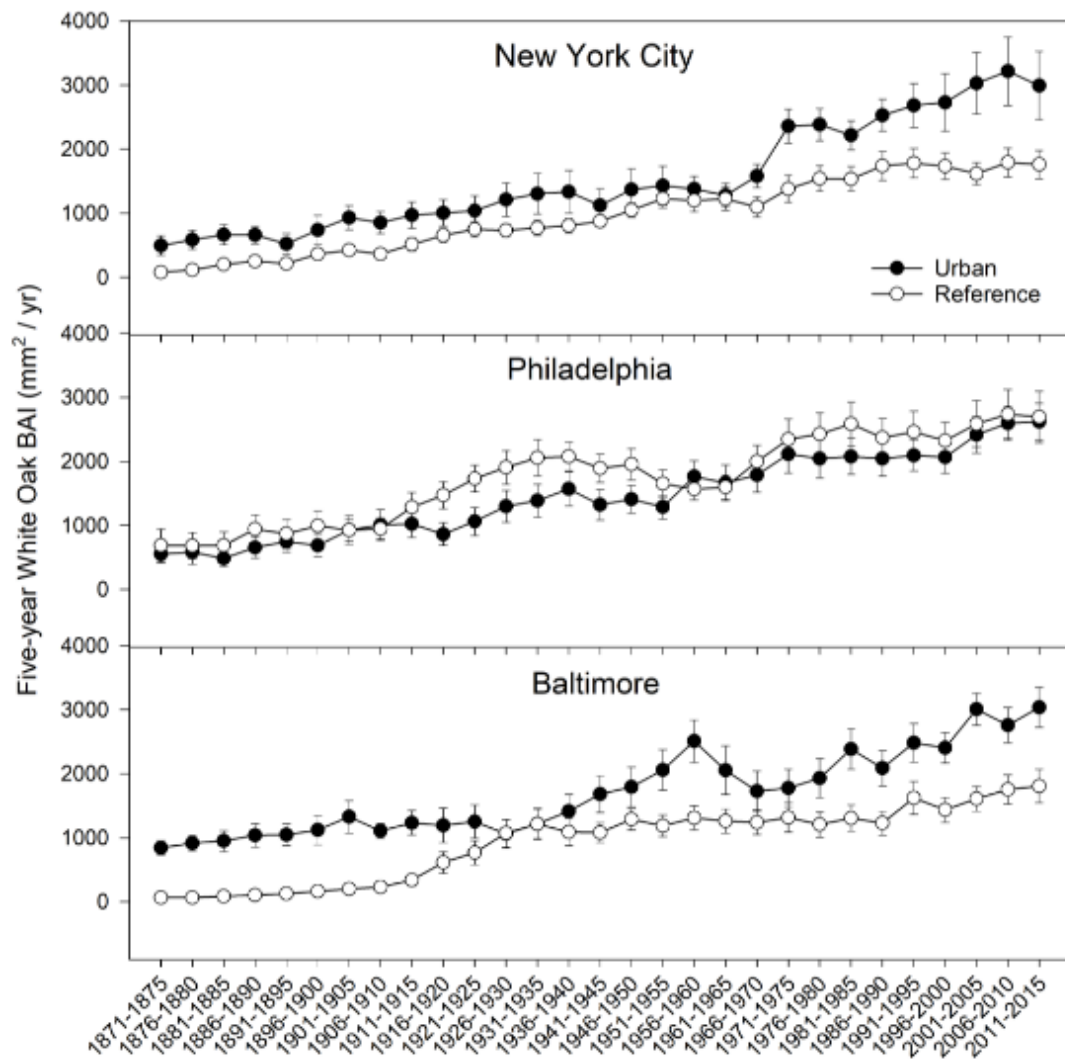


Figure 2.3 Mean five-year basal area increment for white oak trees in urban and reference forest patches of New York City, Philadelphia, and Baltimore. Error bars show mean \pm SEM.

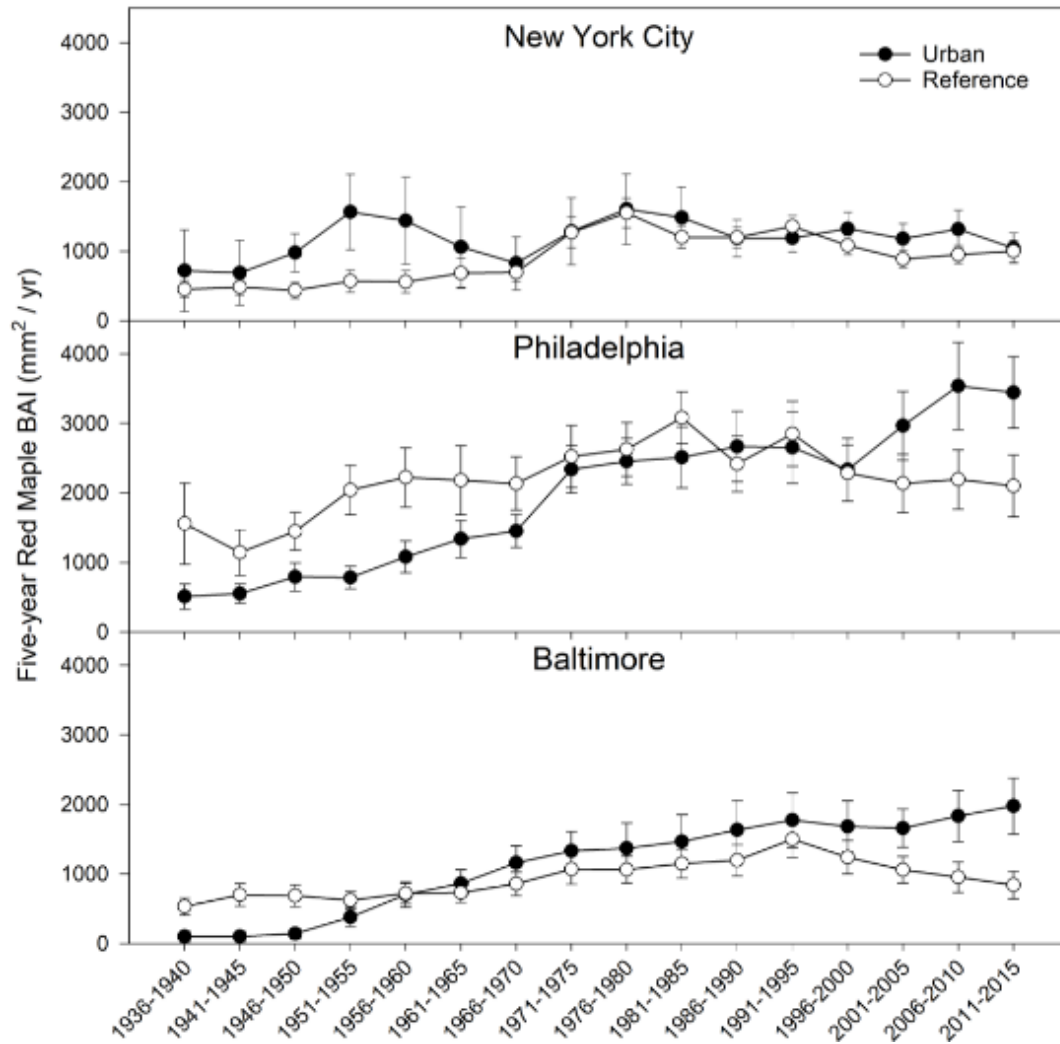


Figure 2.4 Mean five-year basal area increment for red maple trees in urban and reference forest patches of New York City, Philadelphia, and Baltimore. Error bars show mean \pm SEM.

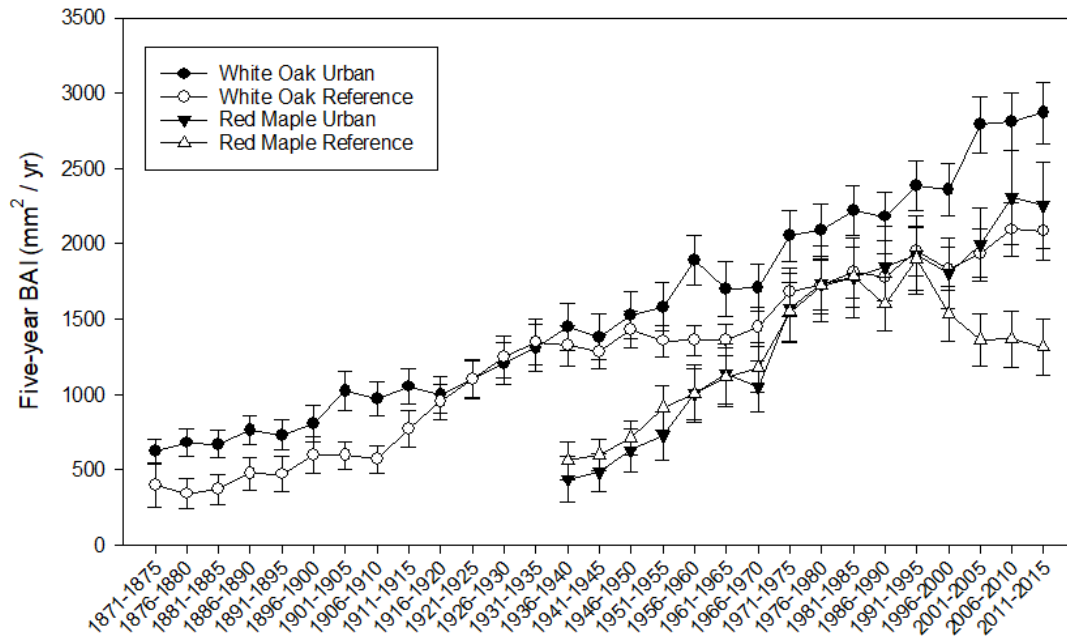


Figure 2.5 Mean five-year basal area increment for white oak and red maple trees in urban and reference forest patches (includes data from NYC, Philadelphia, and Baltimore sites). Error bars show mean \pm SEM.

Across the entire period, white oak BAI was 42.5% greater in urban forest patches than reference forest patches across all cities combined. Although New York City and Baltimore white oaks showed this pattern consistently over time, Philadelphia white oaks showed the opposite trend, leading to a significant city * site type interaction effect (Table 2.5, Figure 2.3). These differences by city and site type also varied over time, as indicated by the significant three-way interaction effect. Beginning around 1970, BAI of urban white oaks in New York City 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.

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 0.01	3.76 0.03	15.20 <0.0001	4.08 0.02	3.76 <0.0001	2.82 <0.0001	2.31 <0.0001	City and Site Type
Red Maple BAI	0.39 0.54	9.04 0.0003	10.83 <0.0001	0.32 0.73	2.48 <0.0001	2.05 0.01	1.52 0.04	City and Site Type
White Oak LWR	0.50 0.48	4.52 0.01	13.05 <0.0001	2.56 0.08	3.75 <0.0001	3.27 <0.0001	1.73 0.0007	City

Table 2.5 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 ($\alpha = 0.05$).

White oak LWR also had a significant three-way interaction effect but did not differ significantly among site types (Table 2.5, Figure 2.6). In Baltimore and New York City, reference LWR was initially greater, but this trend disappeared around 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 2.5, Figure 2.4). 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. New York City red maples showed the opposite pattern, with urban BAI greater than reference BAI until the late 1960s.

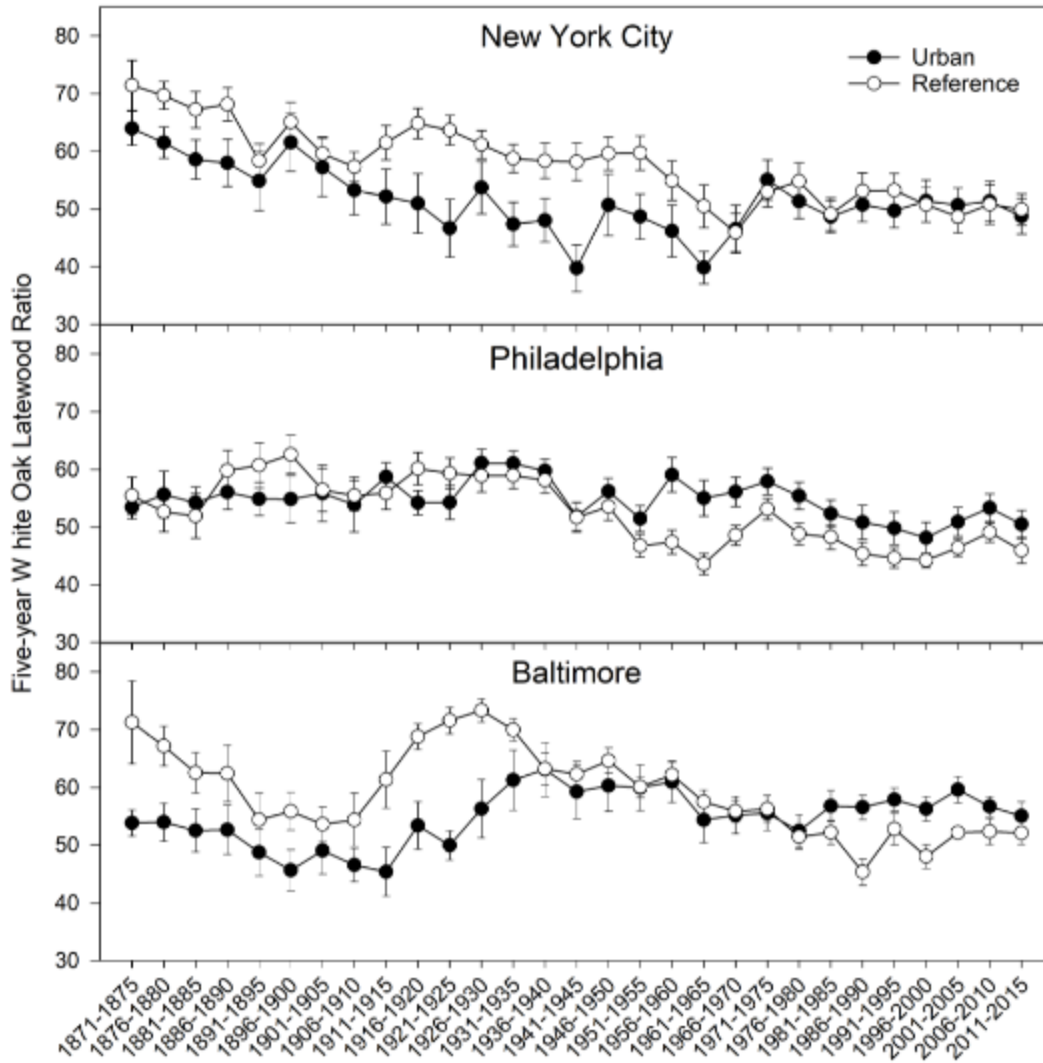


Figure 2.6 Mean five-year latewood ratio for white oak trees in urban and reference forest patches of New York City, Philadelphia, and Baltimore. Error bars show mean \pm SEM.

2.5 Discussion

2.5.1 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 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). I did not find as large of differences 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) and SOM than reference soils. 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 is relatively small (3.6 vs. 3.9) and may not 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 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).

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 (Figure 2.2). 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).

2.5.2 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 (Figure 2.5). 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. In addition, despite concerns that climate change and the urban heat island effect may create stressful drought conditions for urban trees, both the isohydric red maples and anisohydric white oaks have increasing growth rates that will allow them to provide valuable ecological habitat and ecosystem services for urban residents.

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, 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, I 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, nor in population growth, which may be associated with environmental changes caused by urban development (NOAA GHCN; U.S. Census). 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 (NOAA GHCN; Figure 2.7). 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 (Figure

2.3), which may be a result of heterogeneous patterns of disturbance across the urban landscape.

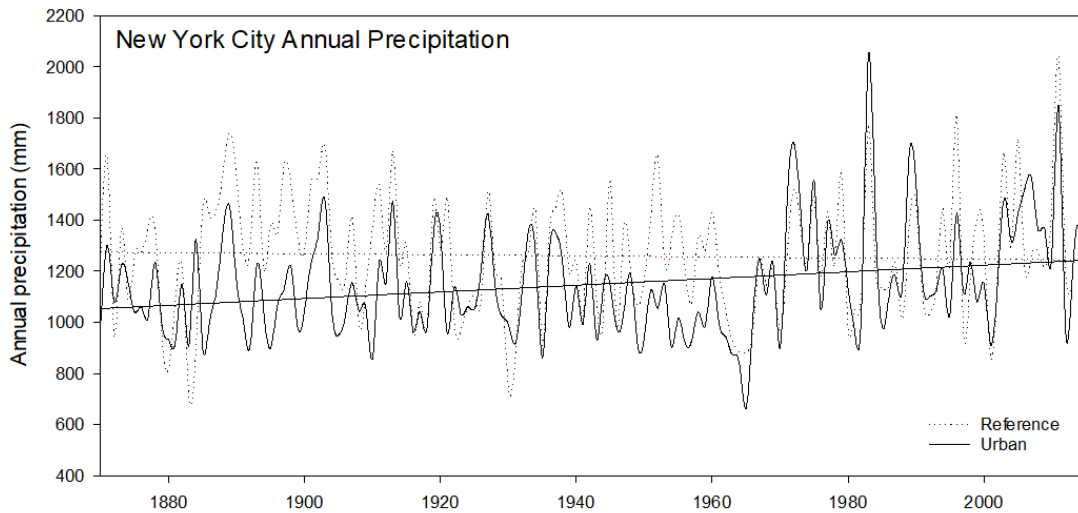


Figure 2.7 Historical annual precipitation for Central Park, NYC and West Point, NY. Data from NOAA Global Historical Climate Network (<https://www.ncdc.noaa.gov/cdo-web/>).

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. Higher annual precipitation may account for the higher LWR at the NYC reference site compared to the urban sites (NOAA GHCN; Figure 2.7). 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 (Sousa et al. 2018), which may affect modeled estimates of urban tree ecosystem services such as carbon sequestration (Westfall et al. in press). 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. It is possible that red maples are more sensitive to the heavy metal concentrations of NYC urban soils, and therefore do not show the same enhanced growth rates as white oaks in NYC urban forest patches. However, 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 overall trends of increasing temperatures and flashier precipitation events caused by regional 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 CO₂ 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 urban vs. rural tree growth shows varying results by species and city (Quigley 2004; Gillner et al. 2014). However, these studies compare urban street trees to rural conspecifics and trees in urban forest patches may not suffer the same drought stress as urban street 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 CO₂ 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 CO₂ in urban areas in a way that trees from rural forests are not. 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).

2.6 Conclusion

Many aspects of the urban environment have the potential to affect tree growth and physiology in forest patches, including higher temperatures, altered CO₂ and O₃ 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, I chose to examine the growth of two important native tree species of the eastern U.S. White oak is 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. and in some cases are experiencing enhanced growth rates compared to trees in nearby reference forests. The continued growth of white oak and red maple trees in urban forest patches will ensure that important ecosystem services are delivered to urban residents in the decades to come. However, the regeneration of these native trees in urban areas is being impeded by deer browse and invasive plants, threats which must be overcome in order to see continued benefits of urban tree growth and productivity.

Chapter 3: Chlorophyll fluorescence parameters, leaf traits, and foliar chemistry of white oak and red maple trees in urban forest patches

3.1 Abstract

The provisioning of critical ecosystem services to cities of the eastern United States depends on the health and physiological function of trees in urban woodlands and forest patches. Although we know that the urban environment may be stressful for trees planted in highly developed sites, it is not clear that trees in urban forest patches experience the same stressful environmental impacts. In this study, I examine chlorophyll fluorescence parameters, leaf traits, and foliar nutrients of urban forest patch trees compared to trees growing at reference forest sites, in order to characterize physiological response of these native tree species to the urban environment of three major cities of the eastern United States (New York, NY; Philadelphia, PA; Baltimore, MD). Overall, white oaks (*Quercus alba* L.) show more variation in chlorophyll fluorescence parameters and leaf traits by city and site type (urban vs. reference) than red maples (*Acer rubrum* L.). Across all sites, red maple trees in this study had higher thermal tolerance of photosynthesis (T_{crit}) than white oaks, suggesting a greater ability to withstand temperature stress from the urban heat island effect and climate change. However, the highest average values of T_{crit} were found in the Baltimore urban white oaks, suggesting that species suitability and response to the urban environment varies across a latitudinal gradient. Urban red maple foliage was

higher in calcium and magnesium than reference foliage, and red maple stomatal pore index (SPI) was higher in urban trees, while white oak SPI was significantly lower in urban trees. Overall, the results do not indicate that urban forest patches provide a more difficult growing environment than nearby reference forests. Urban environmental factors vary across different cities and are sure to impact tree species differently according to their inherent growth strategies, nutrient use, and tolerance for environmental stress.

3.2 Introduction

The ability of plants to persist in urban landscapes will determine their capacity to mitigate aspects of the urban environment that are stressful for human inhabitants, including increased air temperature, flooding, and pollution (Volder 2010; Calfapietra et al. 2015). In the eastern United States (U.S.), trees are dominant components of urban ecosystems, providing valuable biophysical and socio-cultural ecosystem services to urban residents. Urban woodlands or forest patches make up a significant proportion of many cities' tree canopy cover (e.g., Avins et al. 2013; Natural Areas Conservancy 2016), and may have a higher capacity to provide critical biophysical ecosystem services than more intensively managed park areas (Vieira et al. 2018). However, the provisioning of these ecosystem services depends on the health and physiological function of trees in urban forest patches, which may be impacted by the surrounding urban land use (Cadenasso et al. 2007). Although we know that the urban environment may be stressful for trees planted in highly developed areas (Cregg and Dix 2001; Sjöman & Nielsen 2010), it is not clear that

trees in urban forest patches experience the same stressful environmental impacts. Furthermore, native wildtype trees found in forest patches may exhibit different physiological function than planted tree cultivars found in highly managed settings (Lahr et al. 2018).

There is an assumption that the urban environment is inherently stressful to plants (Calfapietra et al. 2015), despite some research to the contrary (George et al. 2009; Zhao et al. 2016). Gregg et al. (2003) found that hybrid poplars grow faster in New York City than in surrounding rural areas, attributing the difference to reduced ozone exposure within urban sites. Similarly, Searle et al. (2012) found that New York City's urban environment promotes red oak seedling growth. However, lower rates of photosynthesis were found in urban trees compared to rural trees in eastern Texas (Lahr et al. 2015). Increased leaf nitrogen and respiration rates in New York City red oak trees compared to rural trees suggest an interaction between temperature and biogeochemistry (Searle et al. 2011). Urban forest patches may experience greater nitrogen availability, leading to higher foliar nitrogen than in nearby rural sites (Nikula et al. 2010; Falxa-Raymond et al. 2014). However, variation in urban forest patch soil quality may lead to differences in native tree growth and physiology within an urban area, making it important to capture forest patch heterogeneity within a city (Pregitzer et al. 2016). Although increases in CO₂, nitrogen, and temperature found in urban environments will generally enhance photosynthesis, warmer temperatures may also increase rates of respiration and of soil water evaporation, which can lower net photosynthetic carbon gain.

Foliar nutrients and leaf-level physiological traits related to the photosynthetic capacity of different tree species may relate to their ability to grow and thrive in urban forest patches. Chlorophyll fluorescence can be used to quickly and easily estimate photosynthetic capacity from recently collected tree foliage (Pontius & Hallett 2014). Performance Index (PI_{abs}) is an integrative chlorophyll fluorescence parameter that reflects the functionality of both photosystems I and II and can indicate stress in plants even before visible symptoms appear on the leaves (Strasser et al. 2000, 2004; Christen et al. 2007). Under water stress, PI_{abs} has been found to be more sensitive than the maximum quantum yield of photosystem II (F_v/F_m), and has been shown to be positively correlated with CO_2 assimilation capacity (Živčák et al. 2008; Van Heerden et al. 2007). T_{crit} is an indication of the thermal tolerance of photosynthesis determined from continuous chlorophyll fluorescence, and is the high temperature where minimal chlorophyll a fluorescence (F_0) rises rapidly indicating disruption of photosystem II (O'Sullivan et al. 2017). Plants in mid-latitude biomes are at the greatest risk of exposure to heat wave temperatures that exceed their thermal tolerance (O'Sullivan et al. 2017), and this risk may be even greater in urban areas. Specific leaf area (SLA) and stomatal pore index (SPI) are leaf traits related to resource use and drought response (Poorter et al. 2009; Ramírez-Valiente et al. 2018), which are likely to be affected by the urban environment. SPI is a good predictor of leaf hydrologic conductance and is associated with maximum rates of gas exchange, indicating potential for carbon assimilation (Sack et al. 2003; Kaproth & Cavender-Bares 2016).

Although we see some evidence of tree physiological responses to the environment of individual cities, it is not clear whether the same species responds similarly across different urban areas. Pouyat et al. (2008) have found that forest soil chemistry responds to urbanization gradients in three different cities, but the nature of response varies depending on spatial patterns of development, parent material, and pollution sources of each particular city. It is possible that native tree physiology has a similarly varied response. Elevated urban temperatures may switch from beneficial to harmful in the lower latitudes of a species' range (Ghannoum and Way 2011). In this study, I examine chlorophyll fluorescence parameters, leaf traits, and foliar nutrients of urban forest patch trees compared to trees growing at reference forest sites, in order to characterize physiological response of these native tree species to the urban environment of three major cities of the eastern U.S. (New York, NY; Philadelphia, PA; Baltimore, MD). I have selected white oak (*Quercus alba* L.) and red maple (*Acer rubrum* L.) trees, which are common species in the region and have been found to thrive in a wide variety of environmental conditions. However, white oak populations have declined over the past century while red maple has become increasingly dominant (Abrams 1998, 2003). The physiological response of these trees to urban environmental conditions may also provide insight into the future of native tree physiology in surrounding rural forests as they are subject to global change impacts such as elevated temperature, atmospheric CO₂, and nitrogen deposition (Ziska et al. 2003; Youngsteadt et al. 2014; Calfapietra et al. 2015).

3.3 Methods

3.3.1 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 (Figure 3.1). The climate of these eastern U.S. cities is strongly seasonal with warm summers and cold winters and all three cities receive ~1,100 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 trees and similar soil types within each metropolitan area (NRCS soil survey). All urban sites were within official city limits, and all reference sites were located in protected areas outside of the city surrounded by intermix wildland-urban interface landscapes (Martinuzzi et al. 2015). Reference ecosystems were dominated by mature native trees with minimal management by humans (Reisinger et al. 2016). 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 except for one Baltimore forest patch are on public parkland or other institutional grounds, and most were private estates during the decades prior. Three reference sites were selected for each city, located within the same protected area considered representative of rural forests of the region.



Figure 3.1 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.

Previous work has documented differences in air temperature and soil chemistry between these urban and reference forest sites (Sonti 2019). During the 2015-2017 growing seasons each city's urban forest patch sites were consistently warmer than reference sites. Daily maximum air temperatures were 3.0°C, 1.3°C, and 1.2°C warmer in urban vs. reference forest patches in New York City, Philadelphia, and Baltimore, respectively. Daily minimum air temperatures were 2.8°C, 1.4°C, and 0.8°C warmer in urban vs. reference forest patches in New York City, Philadelphia, and Baltimore, respectively. Soil characteristics also varied between urban and reference forest patches of each city. Baltimore and NYC had differences in urban vs. reference forest soil nutrients, while all three cities had higher heavy metal concentrations in urban soils.

3.3.2 Data Collection

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 with accessible sun leaves. The trees had no major trunk cavities, and had crown vigor scores of 1 or 2 (less than 25% overall canopy damage) (Pontius and Hallett 2014).

From early July to early August 2015, sun leaves were collected from the periphery of the crown of each tree with either a shotgun or slingshot. Within 24 hours of leaf collection, chlorophyll fluorescence was measured on five leaves from each tree using a Hansatech Pocket PEA continuous excitation chlorophyll

fluorometer. Leaves were kept in a cooler or refrigerator if measurements were not taken immediately following collection, and leaves were dark adapted for at least 30 minutes before measurements were taken. Preliminary trials demonstrated that chlorophyll fluorescence parameters did not change after additional dark adaptation beyond 30 minutes, and were not affected by several hours of storage in a cooler. The Performance Index (PI_{abs}) parameter calculated by the Pocket PEA was used for subsequent analysis.

Within 48 hours of leaf collection, chlorophyll fluorescence was assessed under rapidly increasing heat stress to quantify the critical temperature (T_{crit}) at which F_0 rapidly rises and the disruption of both electron transport in PSII and respiratory biochemical machinery occurs in organelles (O'Sullivan et al. 2013). Leaves were kept in a cooler or refrigerator until measurements were taken. F_0 was monitored continuously on one leaf from each tree with a portable Walz PAM-2000 pulse-amplitude-modulated chlorophyll fluorometer as temperature was increased at the rate of $1^{\circ}\text{C min}^{-1}$ using a heating element inside an insulated chamber. Temperature was recorded simultaneously with F_0 by the thermocouple in the PAM-2000 leaf clip. T_{crit} values were determined using a change-point analysis, which combines the methods of cumulative sum (CUSUM) and bootstrap analysis (Taylor 2000; Rahman et al. 2013).

Three leaves from each tree were pressed and dried immediately after collection. To calculate specific leaf area (SLA), one 10 mm disc was punched out of each dry and weighed after being dried for a week at 70°C . In addition, stomatal density and aperture length were assessed in three spots on each leaf using clear nail polish

impressions. To create the impressions, clear nail varnish was applied to three locations on the abaxial surface of the leaf between the mid-vein and the leaf margin to create impressions of the leaf epidermis. Stomatal density and aperture length of three stomata were measured in three locations on each leaf impression. Digital image recording and image analysis were performed with the INFINITY ANALYZE software (Lumenera Corporation, Ottawa, Canada). Stomatal pore index (SPI) was calculated as the stomatal density \times (pore length)² (Sack et al. 2003). All leaf physiology parameters were averaged for each individual tree.

Remaining oven-dried foliage was ground to create a composite sample for each tree, and was subsequently digested using a microwave-assisted acid digestion procedure (USEPA Method 3052) and analyzed for Ca, K, Mg, Mn, and P by ICP spectroscopy. Foliar N was determined by combustion with a PerkinElmer 2400 series II CHNS/O analyzer (PerkinElmer, Waltham, MA).

3.3.3 Data Analysis

Linear mixed effects models in the nlme R package were used to analyze the effects of city, site type (urban vs. reference), tree species, and associated interaction effects on tree physiology and foliar nutrients (R Development Core Team 2008; Pinheiro et al. 2010). The lme() function allowed the use of site as random effect in each model. Eight models were constructed for each physiological response variable (PI_{abs} , T_{crit} , SPI, and SLA) using a combination of city, site type, species, and interaction effects, and AIC values were used to determine the best fit model. In addition, different variance structures were incorporated into some of the models

using the `varIdent()` function in order to meet assumptions of homoscedasticity (Table 3.1). Differences between means were considered significant at $\alpha = 0.05$.

Foliar nutrients data for each species were submitted to principal component analysis (PCA) using the R function `prcomp()`. Eigenvalues were used to determine that two principal components should be retained for each foliar nutrients PCA. In the case of both PCAs, a varimax rotation was used to aid in interpretation of the principal components using the function `varimax()`. The first and second rotated principal components (RC1 and RC2) from each foliar nutrients PCA were used as response variables in linear mixed effects models with site as a random effect. Because foliar nutrients RCs were calculated separately for each species, four models were constructed for each RC response variable using a combination of city, site type, and their interaction, and AIC values were used to determine the best fit model.

3.4 Results

Tree species had a significant effect on all leaf-level physiological variables measured (Table 3.1). White oaks tended to have higher values of PI_{abs} , and lower values of SLA, T_{crit} , and SPI than red maples (Figure 3.2). Average PI_{abs} for white oak foliage was 13.0 compared to 9.2 for red maple, SLA was 133.6 cm²/g for white oak foliage compared to 155.0 cm²/g for red maple, T_{crit} was 46.6° for white oak compared to 47.1° for red maple, and SPI was 0.06 for white oak compared to 0.10 for red maple foliage. In addition, there was a significant city * site type interaction effect on T_{crit} , a significant species * site type interaction effect on SPI, and a significant effect of city on SLA (Table 3.1).

Response Variable	Species	Site Type	City	City* SiteType	Species*Site Type	Variance Structure
T _{crit}	6.49 0.01	0.31 0.59	0.91 0.43	5.19 0.02		Species
PI _{abs}	53.71 0.0001	2.38 0.15	0.19 0.83	0.02 0.98	0.53 0.47	Species and Site Type
SPI	581.99 0.0001	1.28 0.28	2.04 0.17	2.18 0.15	8.78 0.004	Species
SLA	26.22 0.0001	0.02 0.90	4.19 0.04	0.65 0.54	0.65 0.42	Site Type
WO Foliar Nutrients RC1		5.68 0.03	12.18 0.001	8.48 0.005		
WO Foliar Nutrients RC2		0.01 0.90	0.09 0.91	3.56 0.06		
RM Foliar Nutrients RC1		3.83 0.07	4.02 0.04			
RM Foliar Nutrients RC2		20.27 0.0005	4.12 0.04			

Table 3.1 Results from linear mixed effects models analyzing the effects of tree species, site type, and city and their interactions on foliar nutrients and physiology response variables. F-values and p-values are listed for each fixed effect, and grey cells indicate parameters excluded from the best-fit model for each response variable. Species effects are not relevant for foliar nutrients models, which were analyzed separately by tree species. Bold values indicate significant differences ($\alpha = 0.05$).

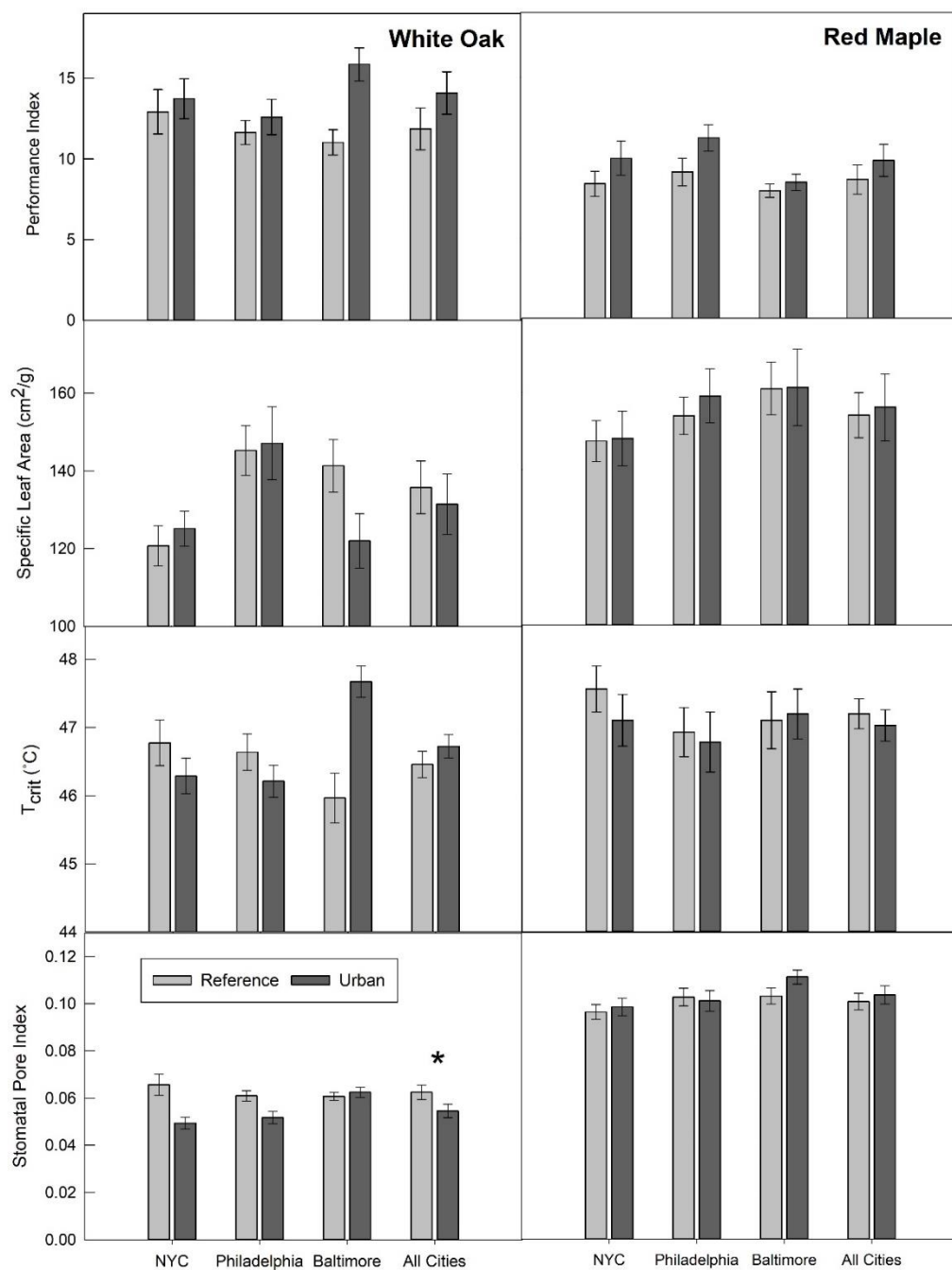
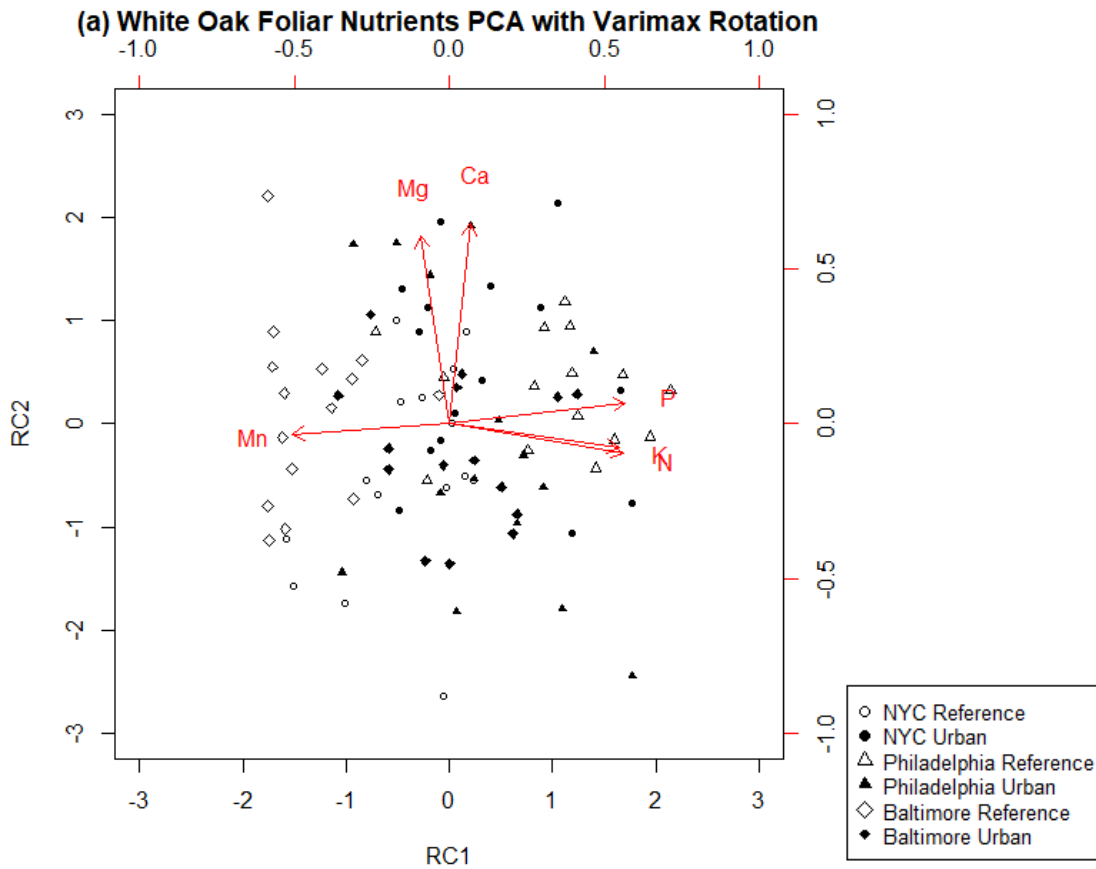


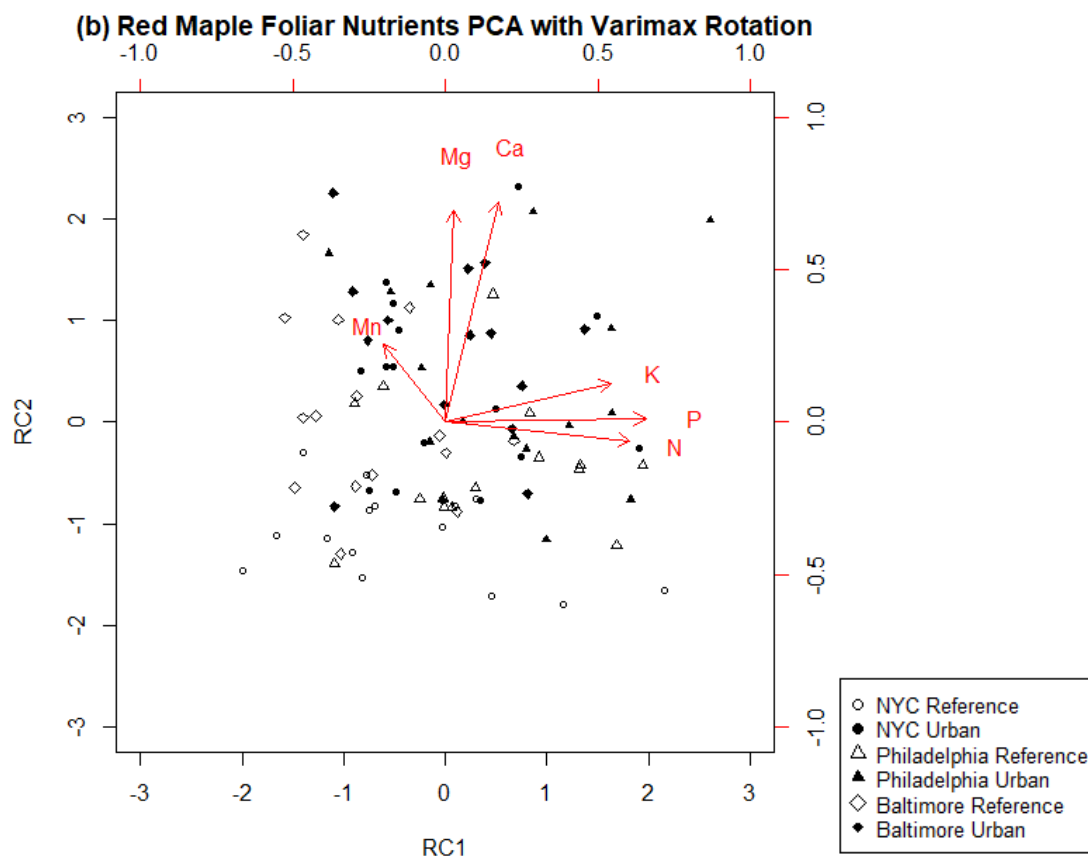
Figure 3.2 White oak and red maple foliar physiology variables by city and site type. Each variable was assessed on fifteen trees of each species within each city and site type combination ($n = 180$). Error bars show mean \pm SEM. Asterisk indicates significant difference between urban and reference stomatal pore index values averaged across all cities ($p = 0.02$).

In the white oak foliar nutrients PCA, 52% of the variation was explained by the first two varimax rotated components, with RC1 accounting for 31% of the variation and RC2 21% (Figure 3.3a). Positive loadings of RC1 corresponded to foliar N, P, and K and negative loadings to foliar Mn, whereas positive loadings of RC2 corresponded to foliar Ca and Mg (Table 3.2). Linear mixed effects models revealed a significant city * site type interaction effect on white oak foliar nutrients RC1 (Table 3.1). Inspection of the scatterplot for RC1 and RC2 shows a clustering of Baltimore reference sites to the left and Philadelphia reference sites to the right (negative vs. positive loadings of RC1 or concentrations of foliar N, P, K, and Mn). There appears to be less separation of white oak foliar nutrients RC2 by city and site type.

In the red maple foliar nutrients PCA, 59% of the variation was explained by the first two varimax rotated components, with RC1 accounting for 30% of the variation and RC2 29% (Figure 3.3b). Positive loadings of RC1 corresponded to foliar N, P, and K, whereas positive loadings of RC2 corresponded to foliar Ca and Mg (Table 3.2). Linear mixed effects models revealed a significant effect of city on red maple foliar nutrients RC1, and significant effects of site type and city on RC2 (Table 3.1). Inspection of the scatterplot for RC1 and RC2 shows a general clustering of urban sites to the upper right and reference sites to the bottom left, indicating that urban red maple foliage is higher in nutrients than reference foliage. In particular, NYC reference sites cluster towards the negative end of RC2 where concentrations of foliar Ca and Mg are low, while Baltimore reference sites cluster towards the negative end of RC1 where concentrations of foliar N, P, and K are low.

Figure 3.3 (a) Biplot of white oak foliar nutrients principal components 1 and 2 after varimax rotation (31% and 21% of variance), **(b)** Biplot of red maple foliar nutrients principal components 1 and 2 after varimax rotation (30% and 29% of variance). Trees from urban forest patches are represented by solid symbols and trees from reference forest sites are represented by clear symbols; shapes are used to differentiate trees from NYC (circles), Philadelphia (triangles), and Baltimore (diamonds).





White Oak	RC1 Loading	RC2 Loading
N	0.700	-0.119
K	0.683	
Ca		0.809
Mg	-0.118	0.760
Mn	-0.633	
P	0.711	
Red Maple	RC1 Loading	RC2 Loading
N	0.756	
K	0.681	0.161
Ca	0.215	0.905
Mg		0.870
Mn	-0.252	0.319
P	0.821	

Table 3.2 Loadings for white oak and red maple foliar nutrients varimax rotated principal components (RCs).

3.5 Discussion

In this study, I found variation in leaf-level physiological responses of white oaks and red maples in urban forest patches compared to reference forest sites across three cities. The physiological responses of these two native tree species may affect their ability to persist in urban forest patches. The findings here are consistent with previous work showing a modest response of red maple leaf physiology to changes in environmental conditions, and Abrams' (1998) assertion that the species' widespread expansion is not easily explained by its leaf physiology. Overall, white oaks show more variation in chlorophyll fluorescence parameters and leaf traits by city and site type than red maples. In particular, white oak physiology appears to respond more strongly to the urban environment of Baltimore forest patches compared to NYC and Philadelphia, which may be due to the warmer temperatures found in Baltimore. These results reveal the importance of local site conditions in influencing urban forest productivity.

Although urbanization can cause great disturbances in native ecosystems, it is not clear that urban forest patches provide a more difficult growing environment than nearby reference forests. Warmer air temperatures at urban sites may allow for higher rates of photosynthesis, without causing additional drought stress. Furthermore, the soils of urban forest patches may actually have greater nutrient availability than reference forest sites (Pouyat et al. 1995; Szlavecz et al. 2006; Falxa-Raymond et al. 2014; Sonti 2019). Previous research on growth rates of the same trees used in this study has shown higher recent basal area increment of urban white oaks compared to reference trees in NYC and Baltimore, and higher recent basal area increment of

urban red maples compared to reference trees in Philadelphia and Baltimore (Sonti 2019). These differences are not clearly reflected in the leaf-level physiology characterized here. Likely, trees in urban forest patches are putting on more aboveground biomass, increasing overall capacity for carbon fixation without necessarily altering leaf-level photosynthesis (Searle et al. 2012).

Overall, red maple trees in this study had higher values of T_{crit} than white oaks, possibly revealing higher photosynthetic tolerance for the warmer conditions of urban forest patches. When averaged across all cities, this difference between white oak and red maple T_{crit} was smaller in urban forest patches than at reference sites. I found that white oaks in Baltimore urban forest patches, which experience the warmest temperatures of any site in this study, actually have the highest average T_{crit} . Perhaps these warm temperatures have surpassed a threshold causing the oaks to acclimate and increase their thermal tolerance of photosynthesis (O’Sullivan et al. 2017). Overall growth and survival of urban trees and their photosynthetic thermal tolerance may not be correlated because photosynthesis occurs only when environmental conditions are favorable (Knight and Ackerly 2003). If favorable conditions occur frequently enough despite episodes of high temperature stress, there may not be selection pressures for increased T_{crit} . However, Baltimore urban white oaks also have a higher average PI_{abs} and lower SLA compared to reference trees, possibly indicating lower overall stress and higher photosynthetic capacity, while variation in these physiological variables between urban and reference trees are not as strong in NYC or Philadelphia. Finally, average SPI of urban white oaks in NYC and Philadelphia is lower than reference trees, while in Baltimore there is no difference

between urban and reference SPI. A lower SPI may indicate drought conditions (Ramírez-Valiente et al. 2018), so it is possible that Baltimore's urban white oaks are experiencing less drought stress relative to reference sites compared to trees of the other cities' urban forest patches. Using the closest available NOAA weather station data, I found that during 2015 NYC, Philadelphia, and Baltimore received 99, 117, and 49 mm more total annual precipitation than local reference sites, respectively (NOAA). However, urban forest patches may receive less infiltration and greater runoff of precipitation due to altered soils and surrounding impervious surfaces (Lahr et al. 2018). Urban study sites in Baltimore have more soil organic matter than the reference sites, while the opposite pattern was true for NYC and Philadelphia urban vs. reference soils (Sonti 2019). These differences in soil quality may affect water holding capacity and potential for drought stress in the trees studied here.

There is a significant species * site type interaction effect on SPI, where reference white oak trees tend to have higher values than urban white oak trees, whereas SPI in red maples did not differ by site type. This may mean that, in general, the reference white oaks are generally less drought stressed than urban white oaks (Ramírez-Valiente et al. 2018), despite the higher amount of precipitation within the urban areas. Higher levels of atmospheric CO₂ may also result in lower stomatal density, while elevated atmospheric ozone can lead to higher stomatal density (Paoletti and Grulke 2005). However, white oak is not sensitive to ozone (Davis and Skelly 1992), and previous research indicates that Baltimore has a stronger urban-rural gradient of elevated CO₂ than NYC (George et al. 2007; Hsueh 2009), which does not support the differences in white oak stomatal characteristics found here.

PI_{abs} did not vary significantly by city or site type, which is not surprising given that healthy trees were chosen for this study and PI_{abs} is an indication of stress. However, PI_{abs} is consistently higher in urban forest patch trees than reference trees across all combinations of species and cities. It is possible that including trees from a spectrum of stress levels would increase this difference between urban and reference trees, particularly if trees in urban forest patches are better able to avoid nutrient deficiencies than trees in nearby reference forest sites (Falxa-Raymond et al. 2014). I found a trend of increasing red maple SLA across the gradient of the three cities, which may be due to warmer air temperatures (Poorter et al. 2009). I also saw this trend across white oaks of NYC and Philadelphia, but Baltimore white oaks have surprisingly low SLA, particularly in the urban forest patches. There is a documented atmospheric CO_2 gradient in Baltimore that may contribute to lower urban vs. reference SLA in Baltimore white oaks, although I did not see this trend in red maples (George et al. 2007; Poorter et al. 2009).

Analyses of white oak and red maple foliar N, P, and K (RC1 in both PCAs) both show a significant effect of city. Both species have the highest values of RC1 in Philadelphia, but white oak RC1 is lowest in Baltimore and while red maple has the lowest values of RC1 in NYC. These differences may reflect variation in nutrient availability within each site relative to where white oak and red maple trees are found. White oak and red maple are likely to occupy slightly drier and wetter locations within a forest, respectively, and soil characteristics can be heterogeneous within a forest site. I observed particular separation between the reference sites of each city along the white oak RC1 axis, with a clustering of Baltimore reference sites

to the left and Philadelphia reference sites to the right. This reveals natural variation in nutrient availability to white oaks between reference forests of the region, which appears to be stronger than the variation between urban forest patches. Previous research has found evidence for convergence in urban soil pH, organic carbon, and total N values compared to reference sites, but divergence in urban soil P and K values (Pouyat et al. 2015).

White oak foliar nutrients RC1 has a significant city * site type interaction effect, where urban trees from Baltimore and NYC both have higher values of RC1 (corresponding to higher concentrations of foliar N, P, K and lower concentrations of Mn), but reference trees of Philadelphia have higher values of RC1. The separation is greatest between Baltimore urban and reference sites, which can be seen in the clustering of Baltimore reference sites to the left of the PCA plot. This pattern is driven by P deficiency in white oak foliage from the Baltimore reference site (Ward and Bowersox 1970; Davis et al. 1995; Mellert and Göttelein 2012). This P deficiency may be contributing to the higher PI_{abs} of Baltimore urban white oaks compared to reference trees. Previous research has in fact shown that these Baltimore urban forest patch soils have more Ca, Mg, N, P, and soil organic matter than the reference forest site (Sonti 2019). Both species foliar N concentrations are below the levels associated with maximum growth rates (Mitchell and Chandler 1939), making them both likely to benefit from the increased N availability of urban forest soils. Mn is more readily available at lower soil pH, and within each city I did find higher white oak foliar Mn at sites with lower pH (NYC reference, Philadelphia urban, and Baltimore reference)

(Sonti 2019) However, there does not appear to be Mn deficiency or toxicity in the trees studied here (St. Clair and Lynch 2005).

Although the city * site type interaction effect is not statistically significant for white oak RC2, I did see variation in urban vs. reference foliage between cities. The greatest separation by site type is seen in NYC, where urban white oak trees have higher values of foliar Ca and Mg than reference trees. Average urban values of red maple RC2 (Ca and Mg) are higher than reference values, largely driven by the high values of Baltimore foliage. It is possible that urban white oaks in NYC and urban red maples in Baltimore are located near construction debris leading to particularly high availability of Ca and Mg in surrounding soils. However, there do not seem to be Ca or Mg deficiencies in any of the trees studied here (Davis et al. 1995).

This study included healthy dominant or co-dominant canopy trees across urban and reference forest sites. However, biotic and abiotic stresses may impact the overall health of native tree populations in urban and reference forests, leading to differences in physiology that are not apparent when examining only healthy individuals. For example, mature red maple trees at these urban sites were frequently observed to have extensive trunk rot, making it difficult to find healthy canopy trees in some cases. Furthermore, evidence from a study of native seedling growth has shown that red oak survival is poor in some NYC urban forest soils (Pregitzer et al. 2016). Urban environmental conditions may also influence the susceptibility of native trees to damage from pests and pathogens, with implications for the long term health of urban forest patches.

3.6 Conclusion

Urbanization is an important force of environmental change impacting ecosystem functioning on continental and global scales (Ellis and Ramankutty 2008). As such, urban ecosystems are thought to mimic global change and provide insight into the future environmental conditions of rural ecosystems (Ziska et al. 2003; Lahr et al. 2018). Forest patches embedded within an urban matrix are directly impacted by environmental changes associated with urban land use, and thus provide a useful context for investigating the effects of these anthropogenic factors on forest ecosystems (Groffman et al. 2006). In this study, I examined urban tree physiology from three major cities in order to gain insight into the ecophysiological functioning of current urban forest patch trees as well as the future of surrounding rural forest trees as they are subject to the effects of climate change and increased regional development.

Both white oak and red maple are widespread species in the eastern U.S., but the past century has seen a decline in white oak and an increase in red maple throughout forests of the eastern United States (Abrams 1998, 2003). However, Abrams (2003) has predicted that warmer conditions associated with climate change may benefit white oak due to its drought tolerance relative to other hardwood species. Although both species are able to thrive in a variety of site conditions, red maple may be able to acclimate even more readily than white oak to the urban forest patch environment. The PI_{abs} of both tree species is higher in urban forest patches compared to reference sites, but red maple SPI is also higher in urban trees, while white oak SPI is significantly lower in urban trees. Together, these physiological results suggest that

the urban environment may allow for greater photosynthetic capacity in red maples, but not for white oaks. Furthermore, red maple T_{crit} is higher than that of white oaks in this study, which may mean that red maples will be better able to withstand temperature stress from the urban heat island effect and climate change. However, the high values of T_{crit} found in the Baltimore urban white oaks suggest that species suitability and response to the urban environment varies across a latitudinal gradient. Further research on the physiology of these species at their southern range limits may reveal greater temperature stress due to urban heat island effects.

The impacts of urbanization on native tree physiology and growth will result in changes to forest species composition, nutrient cycling, and hydrology. Furthermore, urban environmental factors may vary across different cities and are sure to impact tree species differently according to their inherent growth strategies, nutrient use, and tolerance for environmental stress. A greater understanding of these ecophysiological processes will provide insight into carbon, water, and nutrient dynamics of temperate urban ecosystems as large scale urban tree planting initiatives become increasingly widespread and the impact of their changes to the urban landscape is borne out (Campbell 2017).

Chapter 4: Photosynthesis, fluorescence, and biomass responses of white oak seedlings to urban soil and air temperature effects

4.1 Abstract

An increased understanding of the effects of urbanization on tree growth rates and physiological functions, such as photosynthesis, respiration, water use and nutrient use, will help ensure that urban forest patches continue to provide critical ecosystem services. In this growth chamber study, I examine the effects of urban soil and air temperatures on white oak (*Quercus alba* L.) acorn germination, and seedling growth and physiology. A split-plot design was used to test the effects of field collected soils from urban and reference forest sites in Baltimore, Maryland, and warm (urban) vs. cool (rural) growth chamber temperature regimes. Seedlings were harvested at the end of the 23 week experiment in order to assess foliar chemistry as well as biomass allocation. Seedling germination was high in both soil types and temperature regimes. Urban soils supported significantly higher total seedling biomass, but had no effect on root:shoot ratio. Soil type also had a significant effect on leaf-level physiological parameters, with seedlings grown in urban soils having greater A_{net} , V_{cmax} , ETR_{max} , J_{max} , PNUE, g_s , and Performance Index (PI_{abs} ; an integrated chlorophyll fluorescence parameter). PI_{abs} measurements taken throughout the experiment reveal a significant time * temperature interaction effect. Soil physical and chemical analysis revealed that Baltimore urban forest patch soils are higher in nutrients than reference soils, but also higher in heavy metals. Foliar chemistry

analysis also revealed that seedlings grown in reference soils may be experiencing manganese toxicity. Urban forest patch soils are able to support robust white oak seedling growth, but warming air temperatures may cause seedling stress and reduced growth.

4.2 Introduction

The demonstrated biophysical and cultural ecosystem services provided by urban forests have led many municipalities to implement large-scale conservation and reforestation projects (Oldfield et al. 2013). However, there is little published research addressing the impact of urban environmental factors on the success of native forest regeneration or restoration efforts (Oldfield et al. 2014; Doroski et al. 2018). Cities are also considered important ecological analogs for global change, as they produce abiotic conditions that are predicted to occur more widely in the future (Youngsteadt et al. 2014; Lahr et al. 2015). However, it can be challenging to disentangle the effects of co-occurring and confounding environmental variables and the responses of different focal species (Lahr et al. 2018). Cities experience both regional climate change and pollution effects, as well as locally elevated air temperatures and disturbed soil conditions. These aspects of the urban environment are likely to affect current and future urban tree growth and function. In particular, the impact of these environmental conditions on native seedling establishment and growth has implications for the success of reforestation initiatives and natural regeneration processes critical to the continuation of benefits provided by native forest ecosystems in urban areas.

Decades of urban heat island (UHI) effects have been well documented in many cities of the eastern United States where temperate deciduous forest would be the dominant native ecosystem type (Brazel et al. 2000; Rosenzweig et al. 2005; Gaffin et al. 2008). However, most urban woodlands or forest patches are not located near downtown weather stations, and UHI effects vary across the urban landscape depending on land use context and tree canopy cover (Gaffin et al. 2008; Heisler et al. 2016; Scott et al. 2017). Still, temperate deciduous trees are likely to have a positive growth response to warming temperatures, as respiration of these tree populations has been shown to acclimate more strongly than photosynthesis (Way and Oren 2010). The UHI effect is particularly pronounced at night, when artificial surfaces slowly radiate heat from solar radiation absorbed during the day (Arnfield 2003). Experiments from New York City and Baltimore, Maryland suggests that elevated nighttime air temperatures may enhance plant growth, resulting in greater biomass accumulation (George et al. 2009; Searle et al. 2012). In particular, red oak (*Quercus rubra* L.) seedlings grown at urban temperatures allocated more growth to leaves, resulting in greater photosynthetic area without any difference in photosynthetic capacity per unit area of foliage (Searle et al. 2012). However, Meineke et al. (2016) found that urban warming was associated with both reduced photosynthesis and growth of mature willow oak (*Quercus phellos* L.) trees, likely as a result of drought stress. These results suggest that elevated urban temperatures may be beneficial to oak tree growth provided they do not experience drought conditions.

Urban forest soils are often composed of a matrix of remnant natural soils and anthropogenic materials, and exhibit a high degree of spatial variability (Pouyat et al.

2010). Studies indicate that urban soils can have sufficient nutrient levels to support plant growth, but can also have high concentrations of heavy metals and the impact of these contaminants on tree growth and physiology is uncertain (Pouyat et al. 2007, 2010). Urban air contains increased concentrations of atmospheric pollutants, as well as dust arising from roadways and construction activities, resulting in higher concentrations of nitrogen (N) and calcium (Ca) in urban forest soils (Lovett et al. 2000; Pouyat et al. 1995). As a result, urban forest patches may experience greater N availability, leading to higher foliar N concentration than in nearby rural forest sites (Nikula et al. 2010; Falxa-Raymond et al. 2014). However, variation in urban forest patch soil quality may lead to differences in native tree health and growth within an urban area, making it important to assess forest patch heterogeneity within a city (Pregitzer et al. 2016). Furthermore, forest soil chemistry and nutrient cycling may respond differently to urbanization gradients depending on spatial patterns of development, parent material, and pollution sources of a particular city (Pouyat et al. 2008, 2009). As a result, the status of the urban soil biogeochemistry and resulting impacts on tree growth are dynamic, complex, and difficult to predict both within and between cities.

Despite the attention paid to urban climate and soil biogeochemistry, the physiological response of plants to these conditions remains relatively unstudied (Cadenasso et al. 2007; Calfapietra et al. 2015). Furthermore, there is an assumption that the urban environment is inherently stressful to plants (Calfapietra et al. 2015), despite some research to the contrary (George et al. 2009; Zhao et al. 2016). While increases in N availability and temperature found in urban environments will

generally enhance photosynthesis, warmer temperatures may also increase rates of respiration and of soil water evaporation, which can lower net photosynthetic carbon gain. Increased leaf N and respiration rates in New York City red oak trees compared to rural trees suggest a need to further explore the interactive effects of urban climate and nutrient availability on tree physiology (Searle et al. 2011).

Oak trees (*Quercus* spp.) provide critical ecosystem services in forests of the eastern United States, including forests in urban natural areas (Groffman et al. 2006; Schuster et al. 2008; Pregitzer et al. 2019). However, oaks are failing to regenerate through much of their range, primarily due to over-browsing by white-tailed deer and competition from invasive understory plants (Meekins and McCarthy 1999; Huebner 2003; Rooney and Waller 2003; Côté et al. 2004; Doroski et al. 2018). In particular, analysis of long-term changes in regional forest composition reveal that the slow-growing white oak (*Quercus alba* L.) has declined since European settlement of the eastern United States relative to thin-barked shade tolerant species as well as more disturbance adapted oak species (Abrams 1998, 2003). However, eastern populations of white oak are not expected to experience additional 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). White oak is also able to acclimate to conditions of low nutrient availability in infertile soils by increasing nutrient use efficiency (Boerner 1984; Norby et al. 1986). These results suggest that white oaks may be well adapted to the elevated air temperatures and altered soil properties found in the urban forest patch environment.

Baltimore, Maryland is among a growing list of cities advocating for the preservation and restoration of urban woodlands or forest patches (Avins 2013; City of Seattle 2013; Natural Areas Conservancy 2016). These are sites with relatively unmanaged understories comprised of leaf litter and naturally occurring vegetation. Successful seedling establishment and growth in urban forest patch conditions is important for both processes of natural regeneration as well as the success of costly reforestation activities, which often focus on restoration of oak-dominated ecosystems. A preliminary study of Baltimore's forest patches reveals that white oak is the most dominant canopy tree species (Templeton et al. 2016), and it is common in urban woodlands of other cities in the eastern U.S. (Nowak et al. 2016; Pregitzer et al. 2019). In this study, I used a controlled growth chamber experiment to examine the effects of field collected urban soils and air temperatures (compared to reference forest soils and rural air temperatures) on white oak germination, photosynthesis, respiration, chlorophyll fluorescence, and biomass allocation over one growing season.

4.3 Methods

4.3.1 Growth Chamber Study Design

In October 2015, acorns were collected from a stand of white oak trees at a rural forest site used by the Baltimore County nursery for acorn collection used in oak propagation. After collection, acorns that floated in a bucket of water were removed

from the sample, and 600 viable acorns were weighed and planted in Baltimore soils collected from three urban and three reference sites (Figure 4.1).

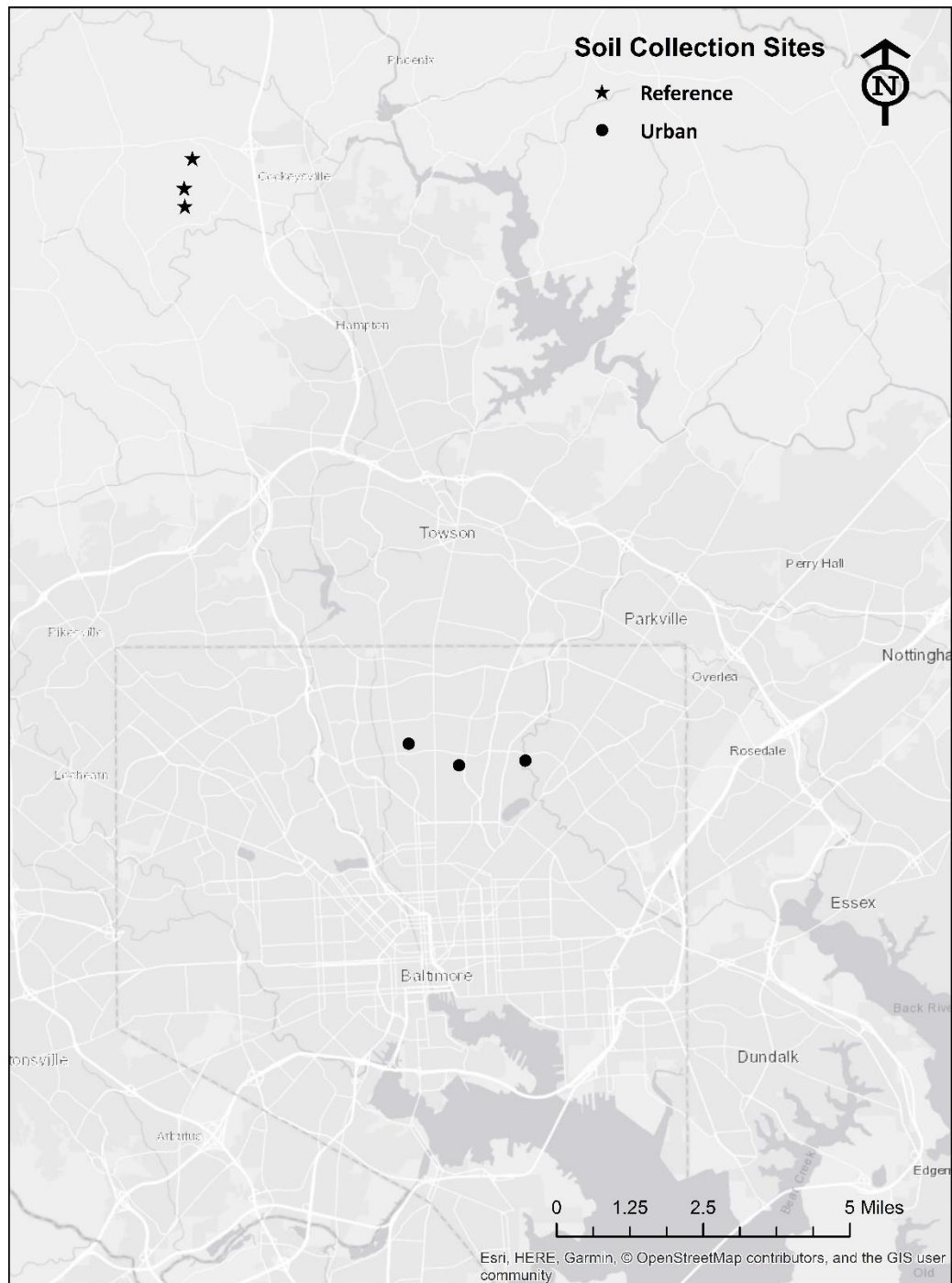


Figure 4.1 Map of reference and urban forest patch soil collection sites in Baltimore, Maryland.

To capture variation, urban soil collection sites were located within forest patches on a variety of land uses approximately 5 km from the city center. The three urban sites included city parkland, museum grounds, and undeveloped vacant land within Baltimore City limits. The reference sites were all located within the 1000-acre Oregon Ridge Park in Baltimore County, approximately 30 km from the Baltimore City center. This location was chosen as a representative reference ecosystem for the region, dominated by mature native trees with minimal management by humans (Reisinger et al. 2016). All soil collection sites were on similar soils of Typic Hapludults with slopes less than 25% and were located in mature white oak stands. At each site, leaf litter and surface vegetation were removed and then approximately 38 L of soil was collected and homogenized.

Acorns were initially planted in flats and placed in a growth chamber at 4°C for three months. After this stratification period, the acorns were divided into two growth chambers, one set at Baltimore City air temperatures ('warm grown') and one set at rural Maryland temperatures ('cool grown'). Long term temperature records were not available for the forest patch soil collection sites at the time of the experiment, so NOAA climate records from urban and rural sites in the Baltimore region were used to set growth chamber temperatures. Chamber temperature regimes were adjusted each week based on 2010-2014 NOAA climate records from the Maryland Science Center station in Baltimore City's Inner Harbor and the Millers, MD station in northern Carroll County, about 50 km north of Baltimore City (Figure 4.2). Minimum, maximum, and mean air temperatures were averaged for seven day periods for the duration of the 26 week experiment, which included early April to

early September temperatures. Daily temperatures fluctuations were calculated for six time steps using maximum, minimum, and mean temperatures. The warm temperature treatment had an average maximum daily temperature 2.2 °C warmer than the cool treatment, while average minimum daily temperature was 5.0 °C warmer.

The acorns remained in the initial two growth chambers for 40 days until germination was complete. At this point, successfully germinated acorns were replanted into larger pots of the same soil, and randomly placed in four growth chambers: two at warm temperature regimes and two at cool temperature regimes. This setup created a split-plot design where each growth chamber contained 48 seedlings: 24 in urban soils and 24 in reference soils (8 replicates from each soil collection site). Pots were randomly moved around each chamber every two weeks to reduce the influence of variation in growing conditions within the chamber. The experiment continued for a total of 23 weeks, which corresponded to early April - early September temperature records.

During cold storage and germination, the seedlings were kept in moist soil to improve chances of germination success. During the remainder of the experiment, the seedlings received uniform watering twice weekly. Relative humidity in the growth chambers was maintained at 50% at all temperatures. Light levels in the chambers were 200-300 $\mu\text{mol m}^{-2} \text{s}^{-1}$ of photosynthetically active radiation (PAR), measured at the top of the canopy.

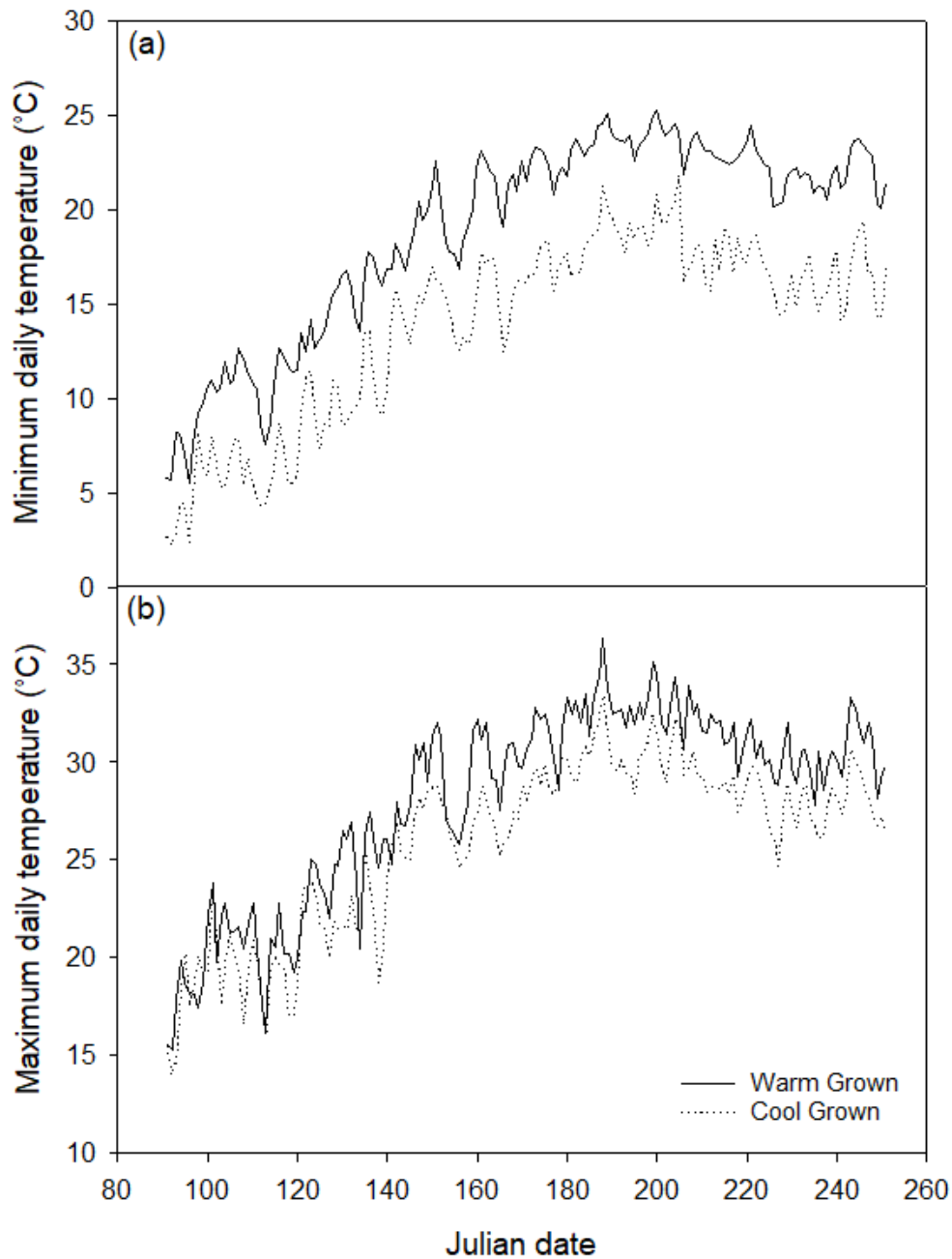


Figure 4.2 Five-year (April-September, 2010-2014) mean daily (a) minimum and (b) maximum air temperatures from urban (Maryland Science Center Station in Baltimore City) and reference (Millers, MD Station in Carroll County) locations in Maryland, used for warm and cool growth chamber treatments, respectively. Data from NOAA Climate Data Online (<https://www.ncdc.noaa.gov/cdo-web/>).

4.3.2 Data Collection

Nine samples of homogenized soil per site were analyzed for physical and chemical properties. Each sample was air-dried and screened to remove particles >2 mm. Samples were analyzed for pH in 0.01 M CaCl₂, organic content was estimated by loss-on-ignition (LOI), and soil texture was determined using the hydrometer method (Day 1965). Plant available elements, including Al, As, Ca, Cd, Cr, Cu, K, Mg, N, Na, P, Pb, and Zn were determined by extracting soil using a modified Morgan's solution (NH₄OAC, pH 4.8, McIntosh 1969). These extracts were analyzed for plant available elements with inductively coupled plasma (ICP) spectroscopy.

Chlorophyll fluorescence was measured four times throughout growing season on the same leaf using a Hansatech Pocket PEA continuous excitation chlorophyll fluorometer. Leaves were dark adapted for at least 30 minutes before measurements were taken. Preliminary trials demonstrated that chlorophyll fluorescence parameters did not change after additional dark adaptation beyond 30 minutes. Measurements of the integrative Performance Index (PI_{abs}) parameter were taken during week 11 (mid-June for seedlings), week 15 (mid-July), week 19 (early August), and week 23 (early September). PI_{abs} reflects the functionality of both photosystems I and II and can indicate stress in plants even before visible symptoms appear on the leaves (Strasser et al. 2000, 2004; Christen et al. 2007).

During weeks 18-20 (early August), net photosynthesis (A_{net}), dark respiration (R_d), and electron transport rates (ETR) were measured on 24 seedlings from each growth chamber (four replicates of each soil), for a total of 96 seedlings. Replicate seedlings were chosen randomly, excluding seedlings whose leaves were senescing.

A_{net} , R_d , and ETR measurements were made on the healthiest fully expanded leaf of the first flush of each seedling, or on a fully expanded leaf from the second flush if necessary. A_{net} and R_d measurements were made using a LI-6400 (Li-Cor, Nebraska USA). Measurements of net photosynthesis rates were made at 25°C over ten CO₂ levels (400, 300, 200, 100, 75, 50, 400, 400, 500, 600, 750, 1000 $\mu\text{mol CO}_2 \text{ mol}^{-1}$). Relative humidity was controlled at 50-60% and light intensity was set to 600 $\mu\text{mol m}^{-2} \text{ s}^{-1}$ PAR after photosynthetic response to light was found to be saturated at that level of intensity. The response of A_{net} to c_i (intercellular CO₂ concentration) was fit with the model and software provided by Sharkey et al. (2007) to predict the maximum light-saturated rate of electron transport for RuBP regeneration (J_{max}) and the maximum rate of Rubisco carboxylation (V_{cmax}). Reported values of A_{net} and stomatal conductance (g_s) are measured at 400 $\mu\text{mol CO}_2 \text{ mol}^{-1}$. R_d rates were averaged over 20 seconds at 25°C and 400 $\mu\text{mol CO}_2 \text{ mol}^{-1}$ after leaves acclimated to the cuvette conditions for at least 3 minutes.

A portable Walz PAM-2000 pulse-amplitude-modulated chlorophyll fluorometer was used to measure the in situ ETR during rapid light curves (RLCs). RLCs were obtained by exposing leaves, not pretreated with a dark treatment, to ten increasing steps of incident PAR ranging from 7 to 2,000 $\mu\text{mol m}^{-2} \text{ s}^{-1}$. The maximum rate of electron transport (ETR_{max}) of light reactions of photosynthesis was determined from the point of light saturation of a plot of ETR against the ambient PAR using a nonlinear curve fitting function in the JMP software (JMP, Version 11. SAS Institute Inc., Cary, NC, 1989-2007). Once RLCs and gas exchange

measurements were completed, each leaf was removed and analyzed on a leaf area meter (LI-3100, Li-Cor, Lincoln, NE, USA).

At the end of the growth chamber experiment, stems of all remaining seedlings were clipped at the base. Coarse and fine roots were removed from the pots and carefully washed to remove soil particles. Leaf, stem, and root tissues were then bagged separately and dried in an oven at 50 °C for one week. All dry tissues were then weighed. Root, stem and leaf mass were summed for total plant biomass. Individual leaves used for RLCs and gas exchange measurements were dried, weighed, and analyzed separately for carbon and N content in order to determine photosynthetic N use efficiency (PNUE). PNUE is the photosynthetic capacity per unit N, and is calculated as the ratio between A_{net} and leaf N content.

Remaining oven-dried foliage was ground to create a composite sample for each seedling, and was subsequently digested using a microwave-assisted acid digestion procedure (USEPA Method 3052) and analyzed for Ca, K, Mg, Mn, and P by ICP spectroscopy. Foliar N was determined by combustion with a PerkinElmer 2400 series II CHNS/O analyzer (PerkinElmer, Waltham, MA).

4.3.3 Data Analysis

Linear mixed effects models in the nlme R package were used to analyze the effects of soil type (urban vs. reference) and temperature regime (warm grown vs. cool grown) and their interactions on seedling physiology and biomass allocation (R Development Core Team 2008; Pinheiro et al. 2010). In order to account for the split-plot experimental design, temperature treatment was analyzed as a plot effect and soil type as a subplot effect. Acorn weight was included as a covariate in the biomass

models. The lme() function allowed the use of soil collection site as a random variable in each model, as well as seedling as a random variable to account for repeated chlorophyll fluorescence measurements on the same seedling over time. In addition, different variance structures were incorporated into some of the models using the varIdent() function in order to meet assumptions of homoscedasticity (Table 4.1). Differences between means were considered significant at $\alpha = 0.05$.

Foliar chemistry and soil nutrients and heavy metals were submitted to principal component analysis (PCA) using the R function prcomp(). Here, 'nutrients' indicate soil 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, 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 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 PCA. A varimax rotation was used to aid in interpretation of the principal components using the function varimax(). The first and second rotated principal components (RC1 and RC2) from each PCA were used as response variables in linear mixed effects models as described above. Finally, biomass and leaf-level physiology variables were correlated with foliar nutrients using Pearson's correlation in the rcorr() function of the Hmisc R package.

4.4 Results

Although acorns in the warm temperature regime germinated earlier, germination rate was 87% across all temperature regimes and soil types. As expected, acorn weight had a significant effect on total seedling biomass (Table 4.1). Soil type also had a significant effect on total seedling biomass, with seedlings grown in urban soils having 20% greater total biomass than seedlings grown in reference soils (Figure 4.3a). However, root:shoot ratio was not significantly impacted by soil type (Table 4.1; Figure 4.3b). Soil type also had a significant effect on all leaf-level physiological variables except for R_d , and there was a significant soil type * temperature interaction effect on V_{cmax} , ETR_{max} , and g_s (Table 4.1; Figure 4.4). Repeated measures analysis revealed a significant three-way interaction effect of soil type * temperature treatment * time on PI_{abs} (Table 4.2; Figure 4.5).

Response Variable	Soil Type	Temperature	Soil Type * Temperature	Acorn Weight	Variance Structure
Total Biomass	11.12 0.004	2.49 0.26	2.72 0.12	49.45 <0.0001	
Root:Shoot	0.10 0.75	1.41 0.36	1.79 0.12	3.00 0.09	Temp Treat
A_{net}	48.33 <0.0001	0.001 0.98	1.41 0.25		
J_{max}	24.45 0.0001	1.45 0.35	2.45 0.13		
V_{cmax}	18.17 0.0005	0.45 0.57	4.07 0.058		
ETR_{max}	24.54 0.0001	0.08 0.80	4.07 0.058		Temp Treat & Soil Type
PNUE	42.37 <0.0001	0.003 0.86	1.37 0.26		Soil Type
R_d	0.03 0.8629	0.65 0.50	1.86 0.19		Chamber
g_s	13.85 0.0016	0.59 0.52	7.93 0.01		Chamber
Foliar Chem RC1	18.18 0.0005	0.04 0.85	1.93 0.18		
Foliar Chem RC2	30.99 <0.0001	5.85 0.14	0.05 0.83		

Table 4.1 Results from linear mixed effects models analyzing the effects of urban vs. reference forest soils and warm vs. cool grown temperature treatment and their interactions on seedling growth and leaf level physiology. Acorn weight was not included in leaf level physiology models. F-values and p-values are listed for each fixed effect, and bold p-values indicate significant differences ($\alpha = 0.05$).

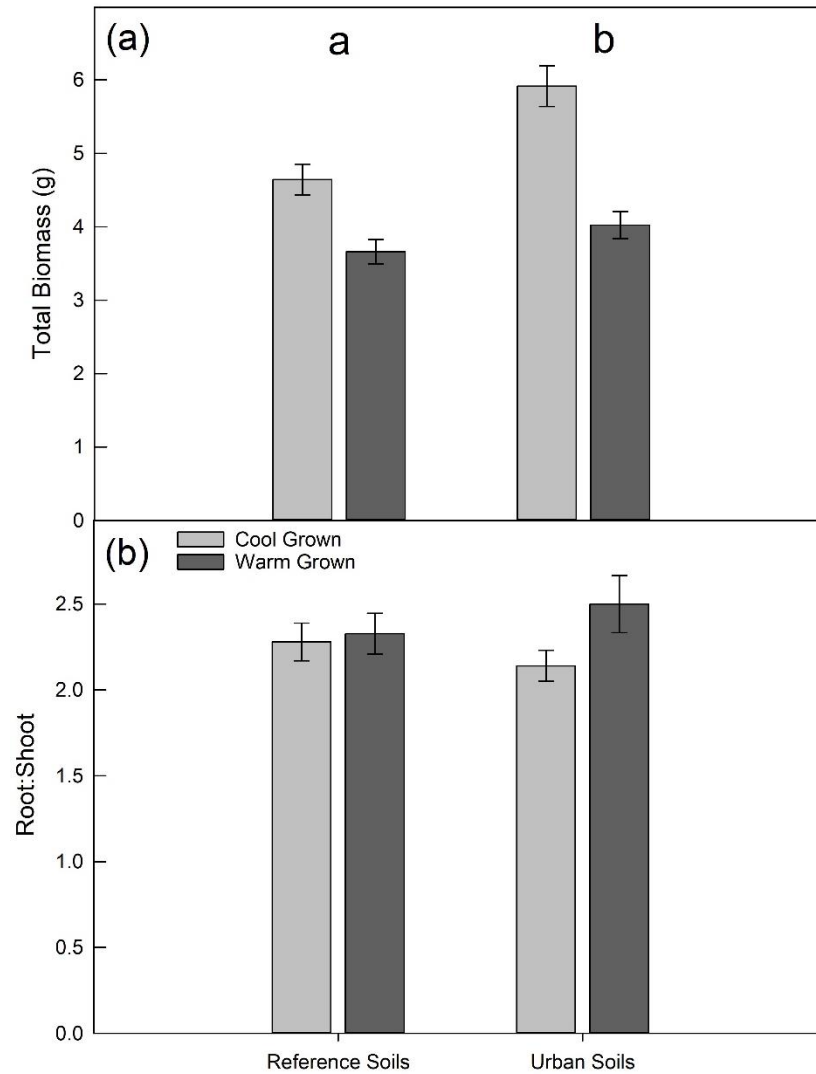


Figure 4.3 White oak seedling biomass allocation by soil type and temperature treatment: (a) total biomass and (b) root:shoot ratio. Values shown are mean \pm standard error (n = 192). Letters show a significant difference in total biomass between soil types (p = 0.004).

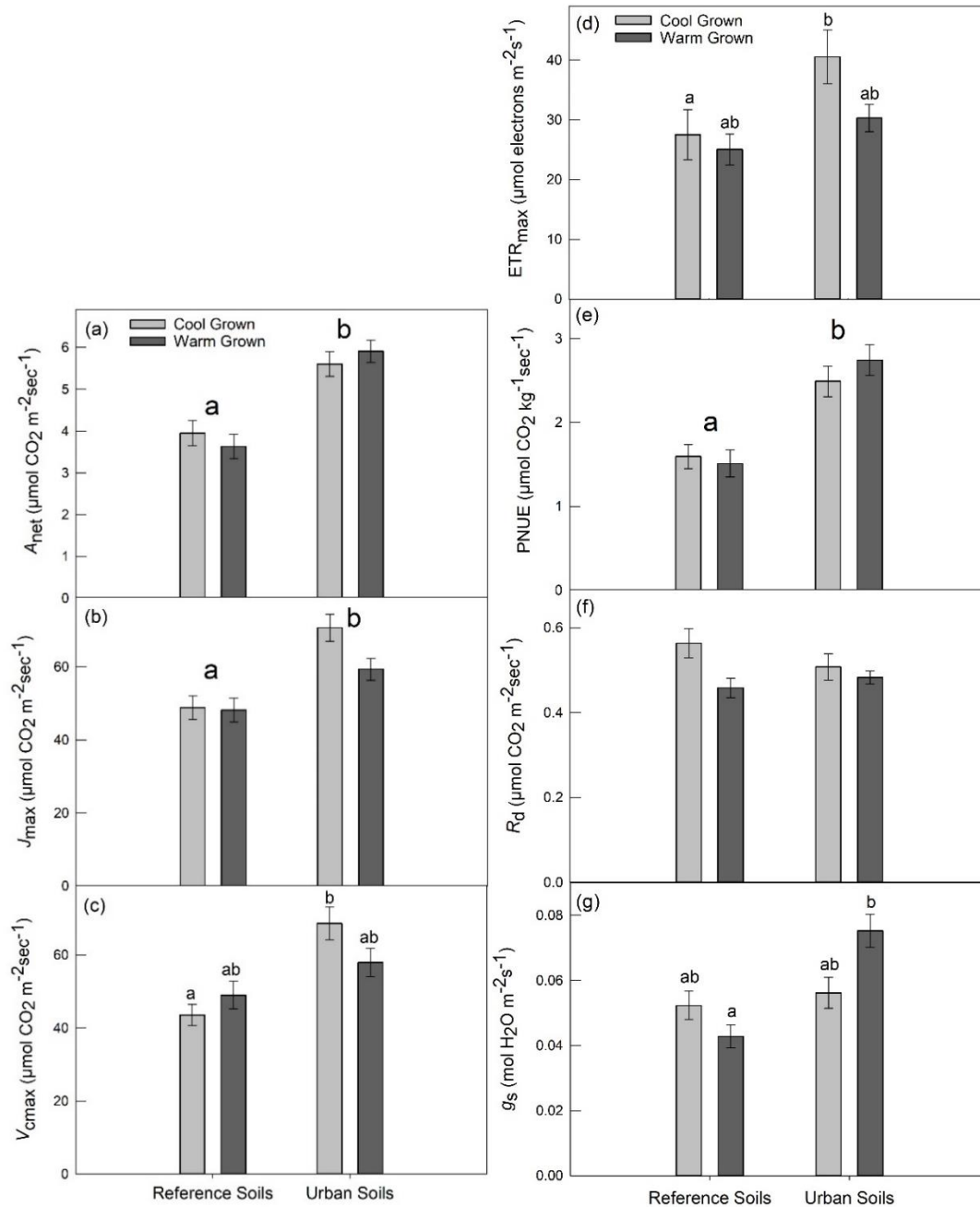


Figure 4.4 Leaf level physiological parameters of white oak seedlings by soil type and temperature treatment: **(a)** net photosynthesis at 25°C, $[\text{CO}_2]$ of 400 $\mu\text{mol mol}^{-1}$, and 600 $\mu\text{mol m}^{-2} \text{ s}^{-1}$ PAR; **(b)** maximum light-saturated rate of electron transport for RuBP regeneration at 25°C and 600 $\mu\text{mol m}^{-2} \text{ s}^{-1}$ PAR; **(c)** maximum rate of Rubisco carboxylation at 25°C and 600 $\mu\text{mol m}^{-2} \text{ s}^{-1}$ PAR; **(d)** maximum rate of electron transport of light reactions of photosynthesis assessed via chlorophyll fluorescence; **(e)** photosynthetic nitrogen use efficiency measured at 25°C and 600 $\mu\text{mol m}^{-2} \text{ s}^{-1}$ PAR; **(f)** dark respiration at 25°C and $[\text{CO}_2]$ of 400 $\mu\text{mol mol}^{-1}$; **(g)** stomatal conductance 25°C, $[\text{CO}_2]$ of 400 $\mu\text{mol mol}^{-1}$, and 600 $\mu\text{mol m}^{-2} \text{ s}^{-1}$ PAR. Values

shown are mean \pm standard error (n = 96). Letters show significant differences between soil types, or soil * temperature combinations (p < 0.05).

Response Variable	Soil Type	Temperature	Time	Soil Type * Temperature	Temperature * Time	Soil Type * Time	Soil Type * Temperature * Time	Variance Structure
PI _{abs}	113.91 <0.0001	0.01 0.92	16.81 <0.0001	1.59 0.22	25.29 <0.0001	15.03 <0.0001	5.16 0.0016	Soil Type & Chamber

Table 4.2 Results from linear mixed effects model analyzing the effects of urban vs. reference forest soils and warm vs. cool grown temperature treatment over time and their interactions on Performance Index (PI_{abs}). F-values and p-values are listed for each fixed effect, and bold p-values indicate significant differences ($\alpha = 0.05$).

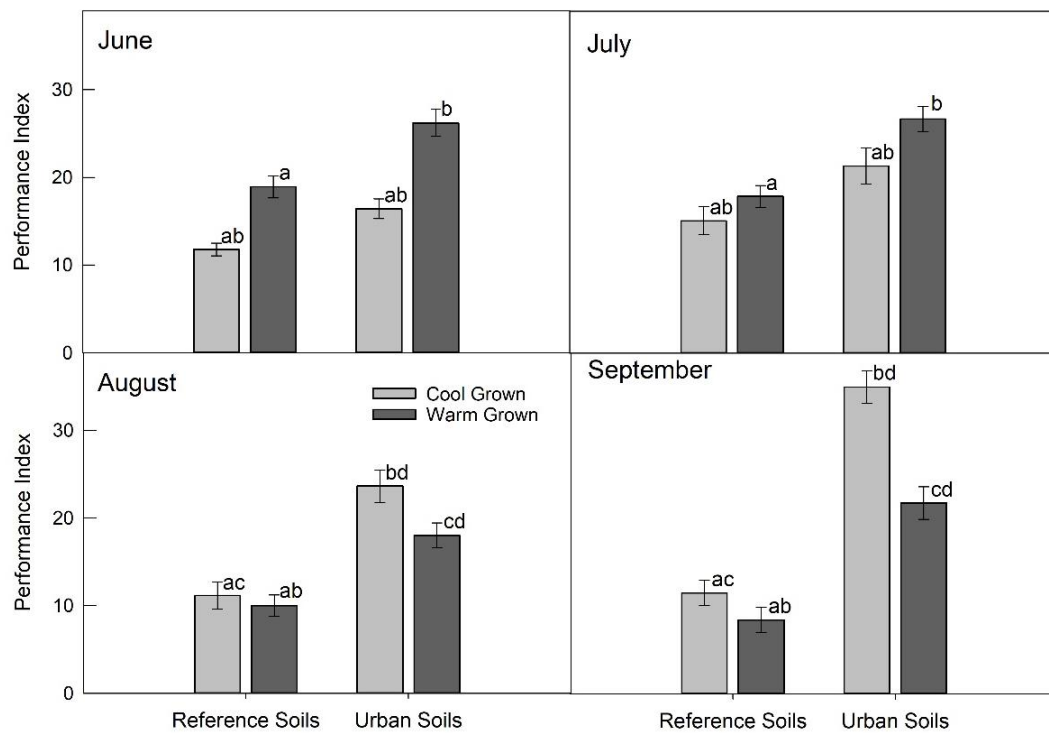


Figure 4.5 Performance Index (PI_{abs}) of white oak seedling foliage by soil type and temperature treatment throughout the growing season. Values shown are mean \pm standard error (n = 192). Letters show significant differences between soil * temperature treatment combinations within each time period (p < 0.05).

In the foliar chemistry PCA, 50% of the variation was explained by the first two varimax rotated components, with RC1 accounting for 27% of the variation and RC2 23% (Figure 4.6). Positive loadings of RC1 corresponded to high values of foliar

Ca and Mg, whereas positive loadings of RC2 corresponded to high foliar P and negative loadings to foliar Mn (Table 4.3). Linear mixed effects models revealed a significant effect of soil type on foliar chemistry RC1 and RC2 (Table 4.1). Inspection of the scatterplot for RC1 and RC2 shows a clustering of foliar chemistry grown in urban soils to the top right and foliage grown in reference soils to the bottom left, meaning that foliage from seedlings grown in urban soils is likely to be higher in Ca, Mg, and P, and lower in Mn than foliage from seedlings grown in reference soils. There is no clear separation of foliar chemistry by temperature treatment.

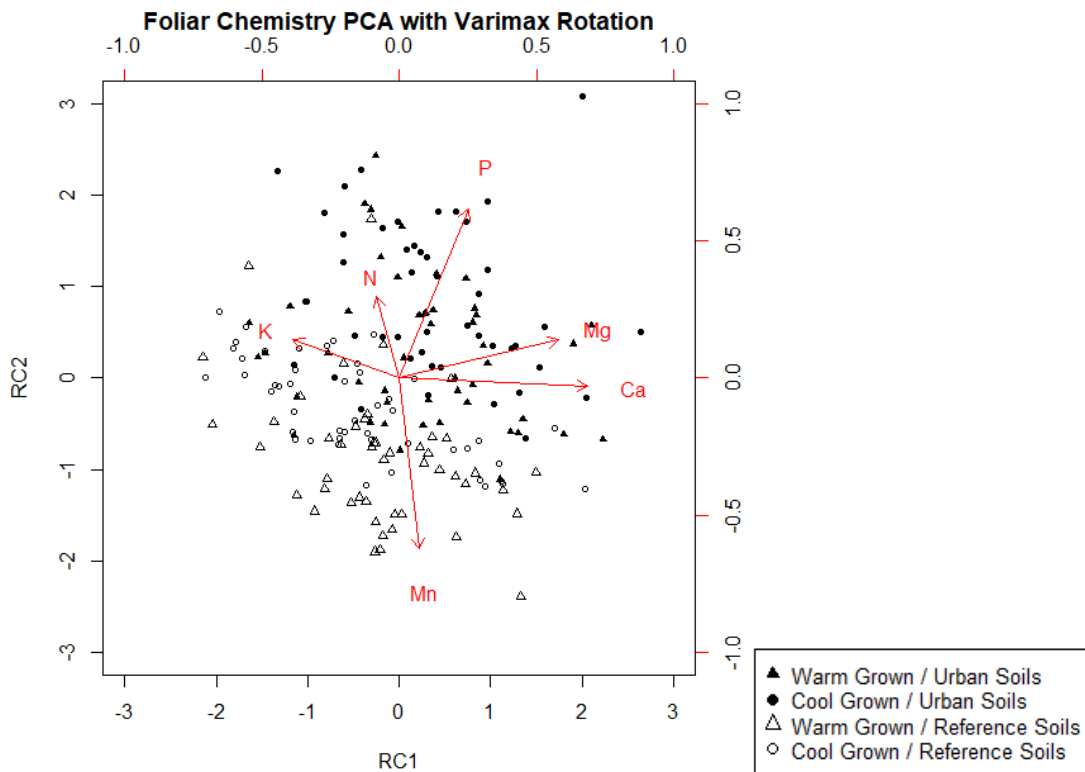


Figure 4.6 Biplot of white oak seedling foliar chemistry principal components 1 and 2 after varimax rotation (27% and 23% of variance). Seedlings grown in urban soils are represented by solid symbols and seedlings grown in reference soils are represented by clear symbols; warm grown seedlings are represented by triangles and cool grown seedlings are represented by circles.

Element	RC1 Loading	RC2 Loading
N	-0.104	0.371
K	-0.484	0.175
Ca	0.858	
Mg	0.724	0.174
Mn	-0.633	-0.778
P	0.316	0.769

Table 4.3 Loadings for foliar chemistry varimax rotated principal components (RCs).

Total biomass was significantly correlated with foliar P and Ca and negatively correlated with foliar Mn (Table 4.4). Root:shoot ratio was significantly correlated with foliar N, P, and Mn. Leaf-level parameters related to photosynthetic performance (A_{net} , J_{max} , V_{cmax} , and ETR_{max}) were most strongly associated with foliar P concentration, but also positively correlated with foliar N and negatively correlated with foliar Mn. R_d was not significantly correlated with any foliar nutrients, and g_s was only significantly correlated with foliar P. PNUE was positively correlated with foliar N and P, and negatively correlated with foliar Mn.

Parameters	Foliar N (leaf)	Foliar N	Foliar P	Foliar K	Foliar Ca	Foliar Mg	Foliar Mn
Total biomass		-0.03	0.28	-0.07	0.18	0.09	-0.30
Root:shoot		0.17	0.19	-0.05	0.04	-0.05	0.20
A_{net}	0.28	0.25	0.52	-0.03	0.13	0.12	-0.29
J_{max}	0.23	0.12	0.55	-0.08	0.10	0.11	-0.27
V_{cmax}	0.23	0.05	0.52	-0.10	0.06	0.02	-0.18
ETR_{max}	0.19	0.24	0.39	0.12	0.00	0.18	-0.25
R_d	-0.11	-0.18	-0.03	0.05	-0.01	-0.19	0.01
g_s	0.12	0.17	0.22	-0.04	0.11	0.00	-0.11
PNUE	0.53	0.38	0.51	-0.04	0.13	0.13	-0.23

Table 4.4 Correlation matrix of physiological parameters and foliar chemistry (Pearson correlation coefficients). Total biomass and root:shoot correlations are based on 192 replicate seedlings; leaf-level parameter correlations are based on 96 replicate seedlings. All values in bold print indicate significant differences ($\alpha = 0.05$).

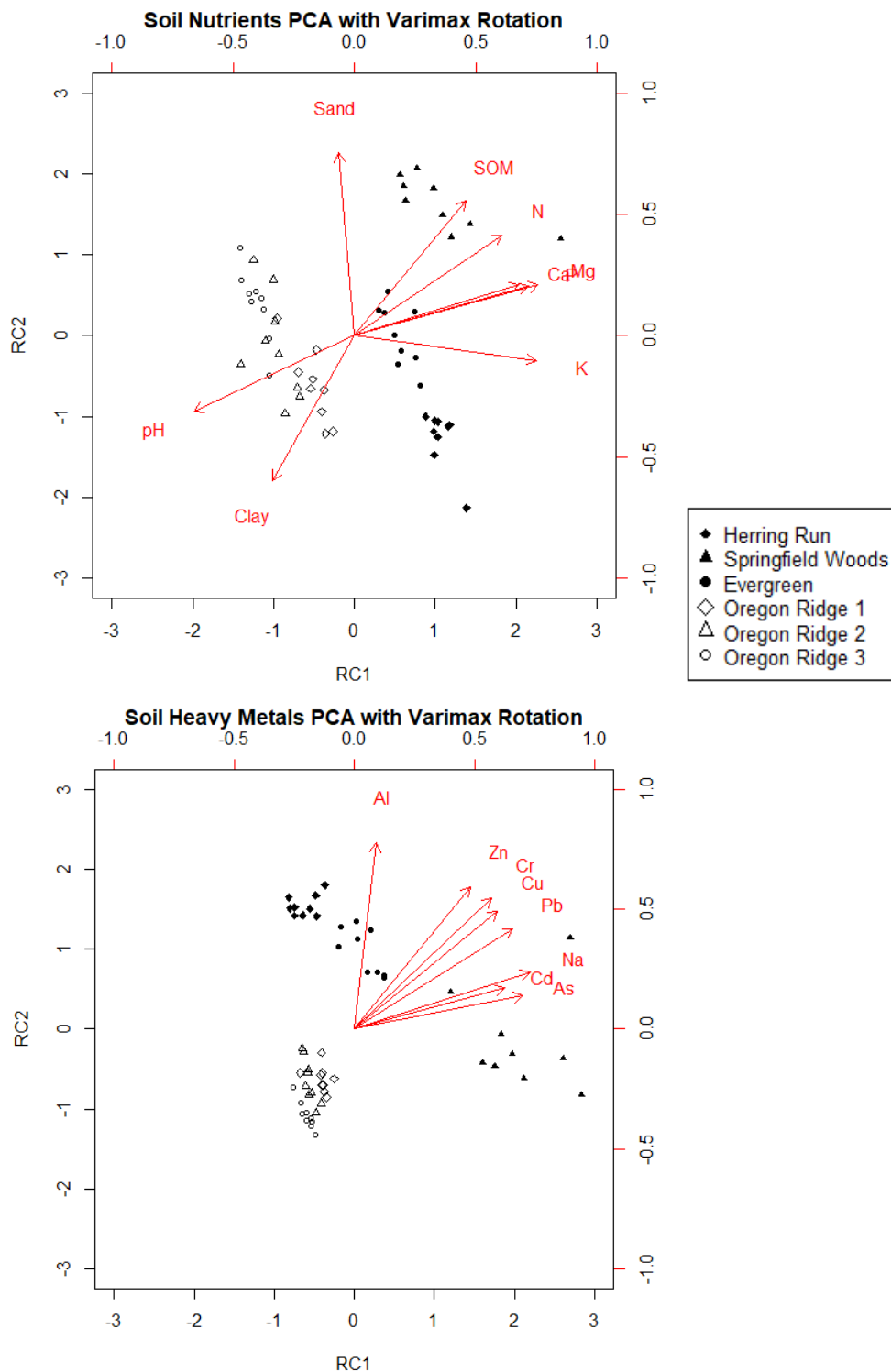


Figure 4.7 (a) Biplot of soil nutrients principal components 1 and 2 after varimax rotation (57% and 29% of variance); **(b)** Biplot of soil heavy metals principal components 1 and 2 after varimax rotation (54% and 35% of variance). Seedlings grown in urban soils are represented by solid symbols and seedlings grown in reference soils are represented by clear symbols; shapes represent soil collection sites.

In the soil nutrients PCA, 86% of the variation was explained by the first two varimax rotated components, with RC1 accounting for 57% of the variation and RC2 29% (Figure 4.7a). Positive loadings of soil nutrients RC1 corresponded most strongly to SOM, N, P, K, Ca, and Mg, and negative loadings to pH. Positive loadings of soil nutrients RC2 corresponded to SOM and % sand and negative loadings to % clay (Table 4.5). Inspection of the scatterplot for soil nutrients RC1 and RC2 showed a greater clustering of reference sites to the left (negative loadings of RC1 or lower pH and lower concentrations of SOM, N, P, K, Ca, and Mg) compared to urban sites. The scatterplot also suggests that there is more variation in urban soil nutrients compared to reference soils, and that the greater clustering of urban soil collection sites reveals that they are less similar to one another than the reference soil collection sites. Linear mixed effects models revealed a significant effect of soil type on soil nutrients RC1, but not RC2 (Table 4.6).

Soil Nutrients	RC1 Loading	RC2 Loading
pH	-0.827	-0.391
SOM	0.574	0.697
Sand		0.943
Clay	-0.421	-0.745
N	0.762	0.518
P	0.899	0.255
K	0.938	-0.129
Ca	0.847	0.260
Mg	0.947	0.264
Soil Heavy Metals	RC1 Loading	RC2 Loading
Cu	0.743	0.615
Zn	0.603	0.743
Pb	0.825	0.522
Na	0.915	0.295
Cr	0.714	0.686
Cd	0.783	0.215
As	0.876	0.174
Al	0.115	0.970

Table 4.5 Loadings for soil nutrients and heavy metals varimax rotated principal components (RC).

Response Variable	Soil Type	Variance Structure
Soil Nutrients RC1	15.18 0.02	Soil Collection Site
Soil Nutrients RC2	0.65 0.46	Soil Collection Site
Soil Heavy Metals RC1	25.42 0.007	Soil Collection Site
Soil Heavy Metals RC2	0.57 0.49	Soil Collection Site

Table 4.6 Linear mixed effects model summaries for soil nutrients and heavy metals varimax rotated principal components (RC) response variables. F-values and p-values are listed for each fixed effect, and bold p-values indicate significant differences ($\alpha = 0.05$).

In the soil heavy metals PCA, 89% of the variation was explained by the first two varimax rotated components, with RC1 accounting for 54% of the variation and RC2 35% (Figure 4.7b). Positive loadings of soil heavy metals RC1 corresponded most strongly to As, Cd, Cr, Cu, Na, Pb, and Zn, while positive loadings of soil heavy metals RC2 corresponded to Al, Cr, Cu, Pb, and Zn (Table 4.5). Inspection of the scatterplot for soil heavy metals RC1 and RC2 showed a greater clustering of reference sites to the bottom left (negative loadings of RC1 and RC2 or lower concentrations of all heavy metals) compared to urban sites. The scatterplot also suggests that there is more variation in concentrations of heavy metals in urban soils compared to reference soils, and that the greater clustering of urban soil collection sites reveals that they are less similar to one another than the reference soil collection sites. Linear mixed effects models revealed a significant effect of soil type on soil heavy metals RC1, but not RC2 (Table 4.6).

	pH	SOM	Sand	Clay	N	P	K	Ca	Mg
				(%)				(ppm)	
Urban									
Springfield	2.63	11.52	57.09	17.09	0.37	14.42	90.46	229.22	66.90
Woods	(2.56-2.75)	(10.91-12.15)	(53.83-62.38)	(14.96-19.61)	(0.34-0.39)	(8.43-27.87)	(74.90-152.75)	(190.09-329.49)	(56.48-99.55)
Evergreen	3.27	8.16	47.55	21.20	0.27	8.74	92.68	267.37	45.62
	(3.18-3.52)	(7.56-8.81)	(38.93-55.38)	(17.94-23.64)	(0.22-0.35)	(6.37-13.01)	(83.17-100.75)	(258.72-282.05)	(43.56-48.07)
Herring Run	3.05	7.78	32.92	22.40	0.23	9.86	116.34	144.50	52.14
	(3.00-3.09)	(7.00-8.89)	(21.55-39.35)	(18.54-24.13)	(0.20-0.25)	(8.31-13.53)	(113.77-122.10)	(134.20-154.82)	(48.99-56.39)
Reference									
Oregon Ridge 1	3.43	7.30	40.16	26.20	0.20	3.76	49.61	63.53	30.83
	(3.41-3.45)	(6.70-7.96)	(32.34-48.98)	(22.49-31.52)	(0.18-0.22)	(2.99-5.25)	(41.17-54.06)	(50.77-74.74)	(25.15-33.31)
Oregon Ridge 2	3.49	7.45	43.92	24.21	0.17	3.46	40.51	17.29	20.87
	(3.33-4.08)	(6.18-8.05)	(37.16-52.74)	(18.01-28.46)	(0.14-0.19)	(2.69-4.07)	(35.61-44.65)	(10.83-22.54)	(17.56-24.32)
Oregon Ridge 3	3.61	7.35	50.56	21.80	0.15	1.85	44.77	26.99	22.76
	(3.58-3.66)	(5.63-8.53)	(43.57-55.82)	(19.00-24.54)	(0.14-0.17)	(1.17-2.84)	(42.90-47.33)	(19.75-36.10)	(21.18-25.37)

	Cu	Zn	Pb	Na	Cr	Cd	As	Al
					(ppm)			
Urban								
Springfield	2.50	6.42	58.41	16.10	0.98	0.05	0.07	317.41
Woods	(2.15-2.83)	(5.09-10.67)	(49.36-70.58)	(13.33-23.78)	(0.78-1.40)	(0.04-0.10)	(0.01-0.12)	(280.61-437.68)
Evergreen	1.44	7.01	37.94	5.35	0.78	0.03	0.03	364.16
	(1.29-1.67)	(5.95-8.86)	(36.57-40.49)	(5.04-5.82)	(0.68-0.94)	(0.03-0.04)	(0-0.05)	(338.12-392.67)
Herring Run	1.75	5.12	28.29	5.34	0.71	0.02	0.01	409.49
	(1.61-1.92)	(4.20-6.64)	(25.46-29.41)	(5.15-5.67)	(0.62-0.85)	(0.01-0.04)	(0-0.06)	(386.60-434.23)
Reference								
Oregon Ridge 1	0.22	1.87	4.10	1.52	0.21	0.03	0.00	285.17
	(0.17-0.35)	(0.79-2.62)	(3.62-4.64)	(0.78-2.85)	(0.15-0.27)	(0.01-0.04)	(0-0)	(266.45-315.65)
Oregon Ridge 2	0.33	0.82	4.00	2.57	0.21	0.01	0.00	277.30
	(0.26-0.40)	(0.25-1.81)	(3.30-5.03)	(1.62-3.42)	(0.16-0.27)	(0.01-0.02)	(0-0)	(244.65-314.30)
Oregon Ridge 3	0.09	0.63	3.99	0.03	0.16	0.01	0.00	229.97
	(0-0.14)	(0.12-1.15)	(3.04-4.94)	(0-0.28)	(0.14-0.20)	(0-0.01)	(0-0)	(207.74-255.16)

Table 4.7 Mean and range of soil characteristics from field soils collected at urban and reference forest sites in Baltimore, MD. Soil nutrients and heavy metals are shown in parts per million (mg kg⁻¹) unless otherwise indicated.

	N -----%-----	P -----	K -----	Ca ----- (ppm)-----	Mg -----	Mn -----
Urban						
Springfield Woods	2.34 (1.57-3.52)	976 (677-1463)	4876 (3314-7219)	8285 (4965-14279)	2946 (2035-4527)	1342 (819-2799)
Evergreen	2.27 (1.46-3.14)	809 (521-1096)	4988 (3577-7071)	10267 (5637-14336)	2224 (1367-3120)	5389 (2722-11314)
Herring Run	2.01 (1.41-2.67)	906 (511-1928)	4673 (2922-8731)	9120 (5773-16070)	2902 (1805-5237)	3705 (1688-6385)
Reference						
Oregon Ridge 1	2.34 (1.32-3.79)	570 (442-715)	5498 (3431-8331)	8419 (5115-12590)	2413 (1394-3303)	6281 (2663-12111)
Oregon Ridge 2	2.08 (1.39-5.43)	676 (463-948)	5764 (3400-8309)	7442 (5233-10143)	2282 (1286-3661)	7414 (3821-11800)
Oregon Ridge 3	1.84 (1.25-2.42)	599 (372-1099)	5760 (2788-9899)	8071 (5747-12061)	3007 (1701-4212)	5150 (1942-11971)

Table 4.8 Mean and range of foliar chemistry values from white oak seedlings grown in field soils collected from urban and reference forest sites in Baltimore, MD. Values are reported in parts per million (mg kg⁻¹) unless otherwise indicated.

4.5 Discussion and Conclusions

In this growth chamber experiment, white oak seedling biomass was greater after one growing season in field collected urban forest soils than in reference forest soils. In addition, leaf level physiological parameters were higher in seedlings grown in urban soils, including A_{net} , J_{max} , V_{cmax} , ETR_{max} , PNUE, and g_s . These results suggest that the nutrients provided by urban soils may be supporting greater photosynthetic capacity resulting in increased biomass accumulation. Indeed, foliage from seedlings grown in urban soils was higher in nutrients than foliage from seedlings grown in reference soils. In addition, high values of foliar Mn suggest that seedlings grown in reference soils may be experiencing Mn toxicity, potentially disrupting photosynthesis (St. Clair and Lynch 2005). Acorn germination was high across all soil and temperature treatments, indicating that urban soils and elevated air temperatures do not significantly impact germination.

Soil physical and chemical analysis revealed that although there is some variation in urban soils, they are clearly separated from reference soils by many parameters. On average, urban soils were higher in nutrients and SOM than reference soils, consistent with previous soil analyses from the same field sites (Sonti 2019). The higher SOM in the urban soils appears to be driven by one site (Springfield Woods), and may increase water holding capacity and nutrient supply to those seedlings (Table 4.7). Foliar chemistry analysis shows that all of the seedlings are somewhat P deficient (below 1400 mg kg⁻¹), but the seedlings grown reference soils are much more so (Table 4.8; Mellert and Gottlein 2012). Foliar P was significantly correlated with total biomass, g_s , and all leaf-level photosynthetic parameters,

suggesting that the greater availability of P in urban soils is benefitting white oak seedling growth. In addition, seedlings grown in one of the reference soils (Oregon Ridge 3) have foliage that is N deficient (below 1.4%; Mellert and Gottlein 2012). None of the average foliar N concentrations in this study are in the surplus range (over 2.7%; Mitchell and Chandler 1939; Mellert & Gottlein 2012), and the higher PNUE of seedlings grown in urban soils suggest that they are still experiencing N limitation. Furthermore, foliar N was significantly correlated with A_{net} , J_{max} , and V_{cmax} , suggesting that additional N is enhancing photosynthesis rates at the leaf level. Although foliar N concentration was not significantly correlated with total biomass, the larger seedlings are likely to have greater overall N content.

Other foliar element concentrations are at similarly deficient (K), normal (Ca), or surplus (Mg) levels across urban and reference soils (Table 4.8; Mellert and Gottlein 2012). However, foliar Ca concentration was significantly correlated with total biomass, suggesting that the elevated Ca availability in urban soils may benefit white oak seedling growth. Calcium additions to rural white oak stands have been found to improve diameter growth (Ward and Bowersox 1970), and urban soils are known to experience elevated Ca availability as a result of cement dust (Lovett et al. 2000). Seedlings grown in reference soils may be experiencing Mn toxicity, as the values of foliar Mn found in these seedlings is high compared to naturally occurring levels (Table 4.8; Davis et al. 1995; St. Clair and Lynch 2005). Although manganese toxicity can cause tree decline (Horsely 2000; Hallett et al. 2006; Watmough 2010), white oak seedlings are considered tolerant to excess Mn (St. Clair and Lynch 2005). However, I found foliar Mn to be significantly negatively correlated with total

seedling biomass, A_{net} , J_{max} , ETR_{max} , and PNUE. Surprisingly, foliar Mn was significantly positively correlated with root:shoot ratio, although excess Mn has been found to inhibit sugar maple (*Acer saccharum* Marsh.) root growth (McQuattie et al. 2000).

Although urban soils from all sites were higher in heavy metals than reference soils, the levels were likely not high enough to negatively impact plant growth (Table 4.7; Sonti 2019). There is little research relating tree growth and physiological response to concentrations of plant available heavy metals (Amacher et al. 2007), but photosynthetic capacity of gray birch (*Betula populifolia* Marsh.) has been shown to be robust to exposure to heavy metal contaminated soils, although tolerance to climatic stress may be reduced (Salisbury et al. 2018). Therefore, the differences found here in growth and photosynthetic performance of white oak seedlings grown in urban and reference soils is likely to due to a combination of increased N and P availability in urban soils, and possible Mn toxicity in reference soils.

There was no significant temperature treatment effect on seedling biomass, although there was a trend of greater average total biomass in cool grown seedlings than warm grown seedlings. It is possible that greater replication of the temperature treatment (more growth chambers) would lead to statistically significant differences between warm and cool grown seedling biomass and leaf level physiology. There was also no difference in root:shoot ratio between soil types or temperature treatments. These results contradict those of Searle et al. (2012) who found greater total leaf area and lower root:shoot ratio in northern red oak (*Quercus rubra* L.) seedlings grown at urban vs. rural temperatures from the New York City region. However, Wertin et al.

(2011) observed a negative growth response of northern red oak to elevated temperatures near the southern limit of the species range. Deciduous tree species growing near their warm range limits exhibit reductions in net photosynthesis and growth, whereas species near their cold range limit respond positively to warming (Reich et al. 2015). Although Baltimore, Maryland is not currently near the southern range limit of white oak, it is possible that the elevated downtown air temperatures used in this experiment are high enough to negatively impact seedling growth. Indeed, white oak is projected to be less prevalent in the Baltimore region under future climate scenarios as the species' range shifts north (Prasad et al. 2014).

Soil type * temperature interaction effects reveal that only cool grown seedlings have significant differences in V_{cmax} and ETR_{max} between soil types. Although the interactions are not significant, the same pattern exists in measurements of total biomass, J_{max} , and PNUE. These results suggest that warm grown seedlings may be experiencing physiological stress that prevents them from taking advantage of the enhanced urban soil conditions (greater nutrient concentrations and lack of Mn toxicity). Cool grown seedlings in urban soils have the greatest total biomass and photosynthetic capacity, demonstrating that this treatment combination provides a better growing environment compared to warm temperatures and relatively nutrient-poor reference soils which may be causing physiological stress to the seedlings. Warm grown seedlings in urban soils had the highest rates of g_s , which were significantly different from that of warm grown seedlings in reference soils. This finding suggests that warm grown seedlings in this experiment were not drought stressed or CO_2 limited, at least when grown in urban soils. There were no significant

soil type or temperature treatment effects on dark R_d . However, the lack of differences in total respiration may conceal greater allocation of respiratory function to growth over maintenance as suggested by the findings of Searle et al. (2011, 2012).

Measurements of the chlorophyll fluorescence parameter PI_{abs} taken throughout the growing season show consistently higher performance of seedlings grown in urban soils compared to reference soils. In addition, warm grown seedlings start out with higher values of PI_{abs} at the beginning of the growing season, but drop over time to become lower than cool grown seedlings by the end of the growing season, indicated by a significant time * temperature treatment interaction effect. Early in the growing season (June and July), only warm grown seedlings show significant soil type differences in PI_{abs} . However, in August and September there are significant differences by soil type within both temperature treatments, indicating that urban soils are better able to support continued photosynthetic activity late in the growing season when urban and rural temperature differences are greatest (Figure 4.2). Although not statistically significant, the difference in PI_{abs} between warm and cool grown seedlings in urban soils at the end of the growing season suggests a decrease in photochemical efficiency in warm grown seedlings compared to cool grown seedlings. PI_{abs} of seedlings in reference soils are much lower, and they may be too nutrient stressed to show any difference by temperature treatment.

The physiological parameters measured in this experiment provide evidence that enhanced leaf level photosynthetic performance of seedlings grown in urban soils may contribute to greater total biomass. However, elevated urban temperatures may not enhance seedling growth relative to rural temperatures. Repeated chlorophyll

fluorescence measurements show that soil and temperature treatment effects change throughout the growing season. The relative values of August PI_{abs} across treatment combinations are generally consistent with J_{max} and V_{cmax} measurements also taken in August, suggesting that the rapid chlorophyll fluorescence parameter may be a good proxy for more time intensive gas exchange measurements.

Mature white oak trees growing at the soil collection sites used in this study were found to have greater basal area increment, higher values of PI_{abs} and greater thermal tolerance of photosynthesis at urban forest patches relative to reference forest sites, attributed to elevated air temperatures and higher availability of soil nutrients (Sonti 2019). Recent measurements from these sites indicate that daily maximum and minimum air temperature differences between the urban and reference forest patches are smaller than those used in this growth chamber experiment, based on NOAA climate records from downtown Baltimore (Sonti 2019). Therefore, the results of this study may indicate future impacts on native tree establishment and growth as the climate continues to warm. In addition, white oak foliage collected from mature trees growing at the reference forest site had lower levels of foliar Mn than the seedlings in this study (Sonti 2019). Mature trees are able to access much deeper soil and may not be as heavily impacted by surface soil conditions of P and N deficiency and excess Mn at reference forest sites. Furthermore, tolerance of mature trees to environmental stress may be greater than that of seedlings (Niinemets 2010). However, seedling establishment and growth in surface soils is critical to forest regeneration and replacement of aging canopy trees.

One advantage of a controlled growth chamber experiment is the ability to isolate the effects of air temperature and soil on native seedling performance. However, as a result there are other aspects of the urban environment that were not taken into account in this study of seed germination, and seedling physiology and growth. For example, urban forest patches may experience flashier hydrology and increased drought conditions due to the urban heat island effect and surrounding impervious surfaces (Groffman et al. 2003; Jacobson 2011; Calfapietra et al. 2015; Lahr et al. 2018). In this experiment, regular watering and relative humidity controls kept seedling water status relatively consistent. In addition, urban areas may experience different levels of atmospheric CO₂ and ozone than surrounding rural areas, with important implications for plant growth (Gregg et al. 2003; Ziska et al. 2004; Lahr et al. 2015). Finally, many important biotic factors excluded from this experiment affect seedling growth in urban and reference forest patches, including deer, invasive plants, insect pests, soil invertebrates and mycorrhizal fungi (e.g., Cregg & Dix 2001; Szlavecz et al. 2006; Karpati et al. 2011; Trammell et al. 2012; Oldfield et al. 2013; Templeton 2016; Schmidt et al. 2017).

In this growth chamber experiment, I found that urban soils and elevated temperatures do not impact germination rates of white oak acorns. However, urban soils collected from Baltimore City forest patches support enhanced growth and photosynthesis, particularly in cool grown seedlings. Although temperature treatment did not have a significant effect on response variables, warm grown seedlings often performed worse than cool grown seedlings, particularly when grown in reference forest soils. Elevated air temperatures resulting from regional climate change may

interact with relatively nutrient poor soils to impact future tree seedling performance in rural forests of the eastern U.S. Nutrient deficiency in eastern deciduous forests is not uncommon, given decades of harvesting and atmospheric pollution (Adams et al. 2000). In addition, forest reserves are often located on nutrient poor sites, as more productive areas have long since been developed for farmland and human settlement (Huston 2005). Therefore, the results found here may not be unique to the Baltimore region. Infertile rural forest sites have been more susceptible to further nutrient loss and soil cation imbalances due to acid deposition (Horsley et al. 2000; Bailey et al. 2005; Sullivan et al. 2013). Seedlings of some native tree species may have difficulty establishing in these conditions when they are also under heat stress likely in future climate scenarios. Furthermore, the ability of urban forest patches to support robust tree seedling growth may be limited in the future as air temperatures continue to warm due to climate change and the urban heat island effect. Successful germination and growth of oak seedlings will be critical to the work of natural resource managers in establishing and enhancing native forest canopy in urban and rural areas.

Chapter 5: Ambivalence in the Woods: Baltimore Resident Perceptions of Local Forest Patches

5.1 Abstract

Urban natural areas have the potential to provide restorative benefits to local residents and may foster feelings of environmental identity and attachment to place. However, wild urban forests can elicit positive and negative emotions, both at a community level and within an individual. These ambivalent feelings may influence the types of benefits derived from these urban green spaces, as well as local residents' desires for their future structure and function as social-ecological spaces. Using data collected from semi-structured interviews, this paper examines resident perceptions and use of local forest patches in Baltimore, Maryland across four case study neighborhoods selected for differences in homeownership and forest patch management. I found that Baltimore residents living adjacent to forest patches experience some of the restorative benefits associated with immersion in wild nature, even when they do not actually enter the woods. However, these positive perceptions and experiences are balanced by negative emotions resulting from the perception that urban wilderness is chaotic and unpredictable. Although homeownership and property management regime may lead to nuanced differences in the benefits and concerns related to forest patches across neighborhood contexts, the qualitative data reveal local residents' strongly ambivalent attitudes towards urban wilderness across all case study sites.

5.2 Introduction

The social benefits of wild urban forest patches are increasingly recognized by major cities throughout the United States (U.S.). Municipal governments and organizations such as Trees for Seattle and New York City's Natural Areas Conservancy strive to increase urban residents' awareness of and access to natural areas. These groups believe that such access will improve natural area visitors' quality of life, including positive impacts on their health and well-being (City of Seattle 2013; Natural Areas Conservancy 2016). However, there is evidence that urban wilderness elicits both positive and negative emotions, both at a community level and within an individual. These ambivalent feelings may influence the types of benefits derived from these urban green spaces, as well as local residents' desires for their future structure and function as social-ecological spaces.

While ecologists often consider wilderness to be the absence of human impacts, social scientists may view it as a cultural construct (Kowarik 2018). For many urban residents, the experience of "wilderness" is more likely to come from a local patch of woods than from an excursion to a national park. As described by Kowarik (2005), "wild urban woodlands" have emerged on vacant or abandoned land in recent decades throughout North American and European post-industrial cities. These sites have the potential to provide critical ecological functions and also important social functions for local communities. However, the characterization of these urban green spaces as "forests" or as sites of social-ecological value is not straightforward for scientists or local community members (Kowarik 2005; Ogden et al. 2018). The claim that these sites provide valuable socio-cultural benefits may be

particularly problematic in cities where a large amount of green space on vacant land has resulted from decades of economic decline and depopulation (Schwarz et al. 2018).

Ambivalence towards natural landscapes is characterized as simultaneous and contradictory perceptions that evoke a mixture of positive and negative feelings and thoughts (Jorgensen and Tylecote 2007; Bonnes et al. 2011; van den Berg and Konijnendijk 2012). Empirical evidence suggests that an ambivalence towards wild nature does not stem from a lack of knowledge and experience with wilderness landscapes, but rather from fundamental human motivations and existential anxiety about one's own vulnerability (van den Berg and Konijnendijk 2012). For example, quantitative research from the field of social psychology finds that ambivalent attitudes towards local urban green space are more closely related to specific values and environmental worldviews than to everyday experiences with such urban green spaces (Bonnes et al. 2011). Ambivalent feelings towards wilderness have been shown to arise from the lack of human control in these spaces, which can lead people to experience both inspiration and terror when confronted with wild nature (van den Berg and Konijnendijk 2012). In a study of Dutch students, Koole and van den Berg (2005) find that wild nature is more likely to evoke thoughts about both death and freedom than either managed nature or a city environment. Quantitative analysis of emotional responses to different types of natural threats reveals that most individuals report a mixture of positive and negative emotions after a fearful experience with nature (van den Berg and ter Heijne 2005).

Urban wild spaces, including forest patches, may evoke similarly strong positive and negative feelings in those who encounter them (Skar 2010; Sonti et al. in press). Jorgensen & Tylecote (2007) suggest that urban woodlands provide contemporary “interstitial” wilderness experiences, existing as a distinct type of representational space where “natural rather than human agencies are in control of shaping the land” (p. 453). Despite the obvious lack of human management in comparison to the surrounding streetscape, urban woodlands arise in the midst of densely crowded cities and therefore may not conform to well-known aesthetic ideals of nature as either cultivated gardens or remote wilderness (Jorgensen and Tylecote 2007). In fact, the local community may interpret a lack of intentional human management as a sign of neglect (Nassauer 1995a), particularly in cities struggling with high vacancy rates and disinvestment.

Despite their potential for provoking deep ambivalence, urban natural areas are considered to provide unique aesthetic and restorative benefits compared to other more intensively managed urban green spaces (Campbell et al. 2016; Threlfall and Kendal 2017). Forested areas inside public parks may provide opportunities for engagement with nature, as well as opportunities for reflection while sitting, reading, walking, or gazing (Feldman 2007; Sonti et al. in press). These sites of wild nature allow urban residents the educational and inspirational experiences of ecosystem processes such as phenology of leaf burst, flowering, color changes, and migratory bird sightings (Konijnendijk 2005). Environmental psychology research has found that urban woodlands have a greater ability to provide restorative experiences than landscaped park settings (Korpela et al. 2010; Tyrväinen et al. 2014). Outside of

formal parkland, the ecological and social value of urban “wildscape” fragments and large vacant areas have been described in case studies from countries including Germany (Keil 2005; Hofmann 2012). Local residents appreciate the unsuppressed natural elements within these unmanaged green spaces, which provide space for active recreational activities, as well as passive “lingering” (Keil 2005). The opportunity for reflection experienced while in wild urban nature may lead to the development of environmental identity: “a sense of connection to some part of the nonhuman natural environment that affects the way we perceive and act toward the world; a belief that the environment is important to us and an important part of who we are” (Clayton and Opatow 2003, pp. 45-46). Urban forest patches may also foster place identity and/or place attachment, concepts that refer to components of identity associated with feelings about a particular place and the emotional ties to place (Clayton and Myers 2015; Stedman 2003). Place attachment in urban natural areas is influenced by the physical characteristics of the place itself, the type and intensity of people’s experience with a place, and their knowledge about nature in general (Ryan 2005).

The aesthetic qualities of urban natural areas can influence the perceived social and psychological benefits to visitors. Urban woodland vegetation may be perceived as either valuable or threatening, depending on social context, individual preference, and vegetation characteristics (Jansson et al. 2013). Urban forests can have a high social value, given that they are easily accessible, are large enough to provide an escape from urban life, and have an open vegetation structure (Coles and Bussey 2000). In addition, perceived restorativeness may increase significantly as a

function of the “naturalness” of urban green space, with the most restorative sites having minimal man-made elements (Carrus et al. 2013, Sang et al. 2016). However, Martens and colleagues (2011) found that “tended” urban forests have a more positive effect on visitor wellbeing after a solitary walk than do “wild” urban forests, as park visitors may have responded to visual cues that the area lacked signs of decay or neglect. Urban forest visitation and perceptions of safety may also be improved by specific management interventions such as removal of trash and signs of vandalism, improved signage and trails, and increased community engagement (Thompson et al. 2013).

Individual social values and experiences can also shape perceptions and use of urban forest patches. For example, Maruthaveeran and Konijnendijk van den Bosch (2014) found in a recent review that in a majority of studies, personal factors (such as gender and past experience) were more influential than social and physical environmental factors in evoking fear of crime in urban green spaces. Although women derive equal or greater benefits from both urban and remote wilderness compared to men, they are also more likely to feel physically threatened in these spaces (Virden and Walker 1999; Pohl et al. 2000; Thompson et al. 2004; Jorgensen et al. 2012). In addition, van den Berg and ter Heijne (2005) found that low sensation seekers and women are more likely to respond to natural threats with fear and avoidance tendencies than are high sensation seekers and men, who are more likely to respond with positive emotion. Sensation seeking is an individual’s genetically and socially determined tendency to seek varied, novel, and intense sensations and experiences, and high sensation seekers typically view risky activities as less

threatening than low sensation seekers (van den Berg and ter Heijne 2005). Because gender and sensation-seeking are stable personal characteristics, this finding suggests that individual differences in emotional responses to natural threats may be difficult to influence or change. Furthermore, individuals with ecocentric views or a greater orientation towards nature are more likely to seek out landscapes that resemble wild nature (De Groot and Van den Born 2003; Shanahan et al. 2015). However, past experiences may also determine attitudes towards urban woodlands; research from the United Kingdom has found that childhood nature experiences are important in determining adult perceptions and use of urban woodlands (Milligan and Bingley 2007; Thompson et al. 2008).

Most research on the social meaning of urban woodlands has focused on park visitors. However, there is a need to understand perceptions of nearby residents who observe these wild spaces every day, but who may or may not enter or engage in physical stewardship of the woods. Similarly, it is important to include forest patches outside of protected parkland, as these may be the most accessible sites of wild nature for many urban residents (Gobster 2011; Rupprecht et al. 2016; Kowarik 2018). As municipal agencies and community organizations work to expand access to and use of all types of urban natural areas, it is important to understand whether they are used or appreciated by those who live nearby. Similar to park visitors, residents who live near urban forest patches may develop ambivalent feelings towards them. For example, urban residents can simultaneously identify local woodland spaces as their favorite places but also feel unsafe when they were alone in them (Jorgensen et al. 2007). Neighborhood context may also affect the perceptions and use of wild urban green

spaces. Focus group research from Christchurch, New Zealand found that residents from different neighborhoods had different perceptions of urban natural areas: “The levels of tolerance to what was perceived as a wild area also appeared to vary according to the location. Those in areas with greater security risks had more reservations about adding a biophysical ‘wilderness’ to a place already seen as a ‘social wilderness’” (Kilvington and Allen 2005, p. 33). Therefore, the restorative benefits provided by urban forest patches may vary according to local community characteristics.

Building on research about the restorative benefits of urban natural areas and about ambivalence towards wilderness, I use qualitative empirical research to examine positive and negative perceptions of wild urban forest patches in Baltimore, Maryland. Case study sites were selected for differences in homeownership and forest patch management in order to investigate variation in resident perceptions and use of forest patches across different social contexts. Data collected through semi-structured interviews reveal ambivalent attitudes held by local residents toward these urban green spaces. In the sections that follow, positive, negative, and ambivalent perceptions of Baltimore’s forest patches are illustrated through four emergent themes, including discussion of variation in responses by homeownership and forest patch management regime.

5.3 Methods

To elicit resident perceptions regarding positive and negative social meanings of local forest patches, this study employed a qualitative case study design. Cases

included four forest patches located in residential neighborhoods of Baltimore, Maryland. In each neighborhood, in-depth semi-structured interviews were solicited with residents living across the street from the forest patch.

5.3.1 Study Setting

Baltimore, Maryland is a post-industrial city that has suffered from depopulation and economic disinvestment, leading to a large number of vacant properties (Boone et al. 2009; Grove et al. 2015). Within Baltimore, thirty-four percent of the tree canopy cover is made up of forest patches, defined as areas of tree canopy greater than 10,000 ft² (Avins 2013). More than half of this forested area exists outside of municipal park boundaries (Avins 2013), creating a varied mosaic of neighborhood characteristics, ownership, and management regimes. Baltimore's forest patches are found on a variety of public and private land uses including municipal parkland and institutional grounds such as universities or churches (Ogden et al. 2018). These forests may be managed as public green space or may not receive any management at all, depending on ownership, institutional budgets and goals, the location of the forest patch, or its visibility within the community. In addition, many Baltimore forest patches exist on undeveloped parcels of land that appear vacant or unused, without clear ownership or a formal management regime. Some of these forest fragments exist across vacant lots and "paper streets" that appear on maps but do not exist in reality (Avins 2013). These sites were never developed and are not formally recognized or protected as green space. In such cases, forest patch stewardship depends on the actions of nearby residents or other community members who may volunteer their time to care for the land. Some local residents do actively

care for forest patches, both individually and as part of community-led stewardship activities (Avins 2013; Ralston 2017). According to a survey of environmental stewardship groups in 2009-2010, there are at least 30 nonprofit organizations working in Baltimore's urban forest patches or woodlands (Romolini and Grove 2011).

5.3.2 Selection of Forest Patches

Four forest patch sites were selected as case studies for this research. All four sites lie in middle income residential neighborhoods of similar housing density that are outside of the inner urban core of Baltimore, where forest patches are rare. Baltimore City median household income from 2010-2014 is \$41,819, and the census blocks surrounding the four patches included in this study range in income from \$34,000 to \$58,000 (US Census Bureau 2013). Although these forest patches lack formal or paved trails to their interior, they also lack fences or any other type of barrier to entry. As a result, all four sites are easily approachable from the public right of way and are easily visible by the residents that live across the street from them. However, each also consists of at least four acres of tree canopy, which is large enough that it is possible to walk inside them and feel a sense of enclosure and separation from the urban streetscape.

After meeting the common criteria of location, accessibility, size, and neighborhood income described above, forest patch sites were selected to contrast varying management regimes and homeownership of adjacent residents (Table 5.1). Two forest patches (Glen Oaks-Chinquapin Run and Perring Loch-Chinquapin Run) are owned by the City of Baltimore and managed by the Baltimore City Department

of Recreation & Parks (BCRP), while the other two forest patches (HEPP Park and Springfield Woods) lie outside of city parkland and are managed by the local community. Then, each management regime was varied by homeownership rates of the adjacent residential blocks. Glen Oaks-Chinquapin Run and Springfield Woods are adjacent to apartment buildings occupied by renters, while Perring Loch-Chinquapin Run and HEPP Park are adjacent to blocks of single family homes occupied primarily by homeowners. This approach allowed for an investigation of the influence of local forest patch and neighborhood context on resident perceptions.

The local non-profit organization Baltimore Green Space has been supporting community-led stewardship activities at HEPP Park and Springfield Woods for several years. “HEPP Park” is not actually a city park, but a patch of forest that was acquired by the city for building a school in 1951 and remains undeveloped. Springfield Woods is comprised of 23 undeveloped lots owned by the Alameda Development Limited Partnership since 1986. Both of these sites have informal signs that designate them as community forests and have had nature walks and environmental stewardship events organized by the community in recent years. The other two case study sites are forest patches in Chinquapin Run Park, which is a 76-acre stream corridor buffered by varying widths of forest patch and fragmented by several major streets. Historical aerial imagery shows that these sites consisted of only a few trees lining Chinquapin Run in 1926 when the land was included as “proposed park extension” in the Olmsted Brothers plan for Baltimore City (Baltimore DPW 1926; Lagrosa et al. 2017). Although the forest patches have grown since then, the city has not created any formal trails or nature programming at these

sites and there appears to be little human management or intervention into natural succession.

Forest Patch	Neighborhood	Homeownership	Management Regime	Number of Interviews
Springfield Woods	Pen Lucy	Primarily renter occupied	Community stewardship	11
HEPP Park	Hamilton Hills	Primarily homeowner occupied	Community stewardship	11
Glen Oaks - Chinquapin Run	Glen Oaks	Primarily renter occupied	Baltimore City Recreation & Parks	10
Perring Loch - Chinquapin Run	Perring Loch	Primarily homeowner occupied	Baltimore City Recreation & Parks	10

Table 5.1 Description of study neighborhoods.

5.3.3 Data Collection and Analysis

In the spring of 2017, semi-structured interviews with Baltimore residents were solicited via door-to-door canvassing by the author and a research assistant. As a result, interviews targeted a cross-section of community members who live near forest patches, including those who may not be actively engaged with or may have negative feelings about the forest. This approach is in contrast to other qualitative research that targets active users of urban green space (but see Jorgensen et al. 2007; Skår 2010). Interviews took place during weekday mornings, afternoons, and early evenings. Contact was attempted up to three times at every door on the blocks adjacent to the forest patches, unless there were clear signs of vacancy. The author explained that the purpose of the study was to understand how residents feel about the forest across the street and asked whether the individual was willing to participate. If the resident agreed, the interview began. All respondents agreed to audio recording of

the interview using a digital recorder. Interviews lasted 15-45 minutes. \$30 gift cards to Target stores were provided to interviewees as an incentive to participate in the study. Data were collected in accordance with University of Maryland IRB protocol (Project 924948-1), and are interpreted as examples of attitudes that exist within the case study sites rather than proportionally representing perceptions of the broader population.

A semi-structured interview is a qualitative research tool that combines a set of open-ended interview questions with the opportunity for the interviewer to use probes or prompts to explore particular themes or responses further (Warren and Karner 2010). In the interviews, residents were asked questions about: (1) their perceptions and interactions with local urban nature in general, including any environmental stewardship activities; (2) their perceptions and interactions with the forest patch specifically, including their awareness of any management activities; (3) demographic information, including age, ethnicity, employment status, level of education completed, and home ownership (see Appendix A. for Interview Protocol). After each interview and at the end of the day, the author made additional field notes and debriefed with the research assistant by discussing patterns in the main themes of the interviews, noting similarities or differences between respondents, and new ideas learned with respect to residents' perceptions of forest patches. These field notes and debriefs provided additional, qualitative context and insights that informed the analysis of the patterns observed in the interview data.

Audio recordings of the interviews were transcribed. Interview transcripts were coded and analyzed using the qualitative data analysis software package NVivo

11. A combination of deductive and inductive coding strategies was applied to the data (Lofland et al. 2006). First, the text was coded for deductive themes of interest to this research, including positive and negative perceptions of the adjacent forest patch, as well as motivations for residents to enter the space or to keep their distance. Emergent inductive themes relating to residents' ambivalence towards the forest patch included personal identity, perceptions of wildlife, forest patch aesthetics, and social function. These cross-cutting themes were identified and coded based on content analysis of the interview text and are elaborated upon in the Results and Discussion below. Building off of previous work on ambivalence towards natural landscapes, I define ambivalence as simultaneous and contradictory positive and negative perceptions, either within a community or within an individual (Jorgensen and Tylecote 2007; van den Berg and Konijnendijk 2012). This paper focuses on ambivalence towards Baltimore's forest patches, and variations in resident use and perceptions of these urban green spaces across the case study neighborhoods.

5.4 Results

Forty-two interviews were conducted across the four sites and the overall response rate was 45% (sampled homes that participated), with a 10% refusal rate (those who answered the door but declined the interview). Homes and apartments that were clearly vacant were not approached and are not included in the response rate. However, there may have been more vacant apartments that were not immediately obvious, leading to a lower response rate. Variation in demographics, length of

residence, and forest patch use across the sites are discussed below, followed by an examination of resident ambivalence towards Baltimore's forest patches.

Respondents ranged in age from 20 to 77 years old with the average age being 50. As might be expected, renters were generally younger than homeowners; average age of respondents in the homeowner neighborhoods was 59, while the average age in the rental neighborhoods was 41. Also as expected, homeowners had lived in their current location for longer than renters; on average homeowner respondents had lived in their house for 12.7 years, while renters had lived in their apartment for 1.2 years. None of the respondents in the rental neighborhoods had lived in their home for more than 10 years, and only 21% had lived there more than two years. In the homeowner neighborhoods, 76% had lived there longer than 10 years and only one respondent had lived in the home less than two years (the individual's family had owned the house for decades, but she had recently moved in). This pattern suggests that the respondents in the homeowner neighborhoods had lived in their home long enough to establish roots in the community, while the majority of the renters were still relative newcomers (Bolan 1997). Table 5.2 contains additional demographic information about the interviewees.

	<i>n</i>	<i>%</i>		<i>n</i>	<i>%</i>
<i>Gender</i>			<i>Employment</i>		
Female	29	69	Employed	19	45
Male	13	31	Full Time Student	3	7
			Employed and in School	2	5
<i>Race/Ethnicity</i>			Disability	3	7
Black	31	74	Unemployed	2	5
White	5	12	Retired	13	31
Other	6	14			
<i>Education Completed</i>					
Some High School	4	10			
High School	12	28			
Some College*	17	40			
4-year College	5	12			
Graduate School	4	10			

Table 5.2 Demographic descriptors of residents interviewed.

N = 42. *Includes vocational or associates degrees.

The majority of respondents said that they never go into the woods (62%), and this response was more pronounced in rental neighborhoods (76%) than in homeowner neighborhoods (48%). Of those who did interact with the woods, more than half reported very minimal contact with the space, including walking along the edge or just having visited once. Only six respondents (14%) reported regularly going into the forest patch now or in the past, and these included residents from all four case study sites.

Thirty-one percent of respondents said that presence of the forest patch affected their decision to live in that location, and these individuals were largely motivated by the aesthetic qualities of the forest patch. However, there was no difference in responses between renter and homeowner neighborhoods. In addition,

residents who chose to live next to the woods were not more likely to regularly visit than those who did not choose their home based on the presence of the forest. In fact, there were many respondents who had never gone inside the forest patch but did seek out their home because of its proximity to the green space.

Strong evidence for resident ambivalence towards urban forest patches was found throughout the four case study neighborhoods, both at the community level and within individual respondents. The four themes discussed below illustrate different aspects of these ambivalent attitudes held by local residents: perceptions of wildlife, personal identity, forest patch aesthetics, and social function. These themes were identified in interviews across all four case study neighborhoods. Contrary to expectations, there were limited differences in the emphasis of each theme by homeownership and forest patch management regime.

5.4.1 Perceptions of Wildlife

A majority of residents at all four sites talked about their perceptions of wildlife in and around the neighborhood forest patch, and it was the most prevalent theme discussed in the interviews. Individual residents expressed positive, neutral, or ambivalent feelings toward the wildlife associated with the forest patches, but were never completely negative. When residents expressed fear of animals, it was almost always mixed with awe and fascination. It is clear that many residents derive inspirational, restorative, recreational, and educational benefits from the presence of wildlife. Although the provision of wildlife habitat was considered an intrinsic feature or benefit of the forest patch, appreciation of wildlife provided further psycho-social benefits to many residents.

Deer were the most commonly discussed animals throughout the interviews, and elicited very strong emotions from respondents. One woman homeowner living next to the BCRP-managed Perring Loch-Chinquapin Run forest patch talked about the deer as if they were members of the community, remarking that “especially around fall when I’m coming home at night, sometimes they have a reunion, they have the deer reunion, there’s so many of them.” Another woman from the same block said, “We take pictures when we see a lot of deer. And then I allow my little babies to come out and see and say ‘Oh look at the deer’.” Several women respondents from Perring Loch-Chinquapin Run also used the word “love” in describing the deer. One woman said, “on some mornings you can see the deer come out, I love that” and another exclaims: “I miss the deer...I sure ain’t seen none lately! I loved seeing them.” Although these large, charismatic animals elicited feelings of joy for many residents, these respondents did not include any of those who reported going into the woods. Rather, these residents loved the experience of watching deer from their porch, and felt a strong emotional connection to the forest patch while keeping a safe distance.

Besides deer, respondents mentioned seeing or hearing birds, fish, raccoons, possums, squirrels, foxes, rabbits, and tadpoles. Fishing, catching tadpoles, and chasing foxes were described as beneficial activities for children who use the woods recreationally, either with their parents or on their own. The sight of foxes, possums, and raccoons seemed to surprise many residents who thought that these wild animals would not live within Baltimore City. Respondents often anthropomorphized these animals, describing them as if they were neighborhood characters. One woman renter

living near Springfield Woods said she loves watching the wildlife and doesn't mind "seeing Miss Foxy walk up and down the sidewalk." A mother and daughter had a vivid recollection of an encounter with a possum from the Perring Loch-Chinquapin Run forest patch who "was going down the steps holding a cucumber" from their backyard garden and had "nerve, like a person going down the steps." As with viewing deer, observing other forest patch wildlife can provide emotional, spiritual, and creative inspiration for city residents. Several respondents talked about taking pictures of the wildlife, including songbirds, foxes, and even a vulture on a light post by Springfield Woods who "was up there like he was posing." Watching or listening to birds or deer was described as "relaxing" and something that makes a person "feel good." Although not exactly considered "wildlife," some residents at community-managed Springfield Woods expressed a fondness for seeing the feral cats that are fed by local residents. One woman remarked, "I've seen this one black cat every time, it's always like that's its natural habitat...it comes over here sometimes but it always goes back to the forest. I think it's superstition with black cats, I guess that's its magical ground." The urban wildlife supported by forest patches can provide inspiration and even spiritual fulfillment for Baltimore's residents.

In contrast, some respondents conveyed feelings of fear or concern about deer, bugs, snakes, raccoons, and foxes. A few residents expressed the belief that the animals were unnaturally comfortable around humans. One young man renting an apartment near the BCRP-managed Glen Oaks-Chinquapin Run forest patch was unnerved by the fact that both deer and raccoons were "too close for real...they're not scared of humans." Others were more afraid of the unknown creatures that might be

hiding in the forest patch. A woman living near Glen Oaks-Chinquapin Run explained: “I am a little nervous about what’s going to hop out at me,” clarifying that she was more afraid of unknown animals than people who might be lurking in the forest.

Some individual residents expressed mixed feelings about different kinds of forest patch wildlife. For example, a woman homeowner who was fond of the deer in the Perring Loch-Chinquapin Run forest was afraid that other wildlife might harm her: “Basically I’m afraid. Because I’ve never been in there before. Usually I’m by myself so there’s no one to go in there with. And I’m just afraid, I don’t know. Might be snakes over there, who knows. It’s the deer I know, but I don’t know what else is there.” This fear of the unknown prevented many residents from getting too close to the forest patch. Other residents were similarly comfortable with only some elements of forest patch wildlife. One woman fondly listed all the animals she has seen from her home at the Perring Loch-Chinquapin Run forest patch, but then explains that she never goes in the woods because she is “allergic to trees and grass and [doesn’t] like the bugs and insects.” The ubiquitous front stoop in this neighborhood allows residents to observe charismatic wildlife from a distance without coming into contact with undesirable parts of the forest ecosystem.

Overall, perceptions of wildlife were discussed more often by residents living adjacent to city parkland, likely because the physical characteristics of these forest patches influence the amount and type of wildlife that residents are likely to see compared to those on vacant land. Both BCRP-managed Chinquapin Run forest patches are part of larger riparian corridors (although not entirely forested), and are

inhabited by deer that move along the corridor. While the two forest patches on vacant land do not contain a smaller area of tree canopy, they are not directly connected to a larger network of green space, and so they do not appear to support deer populations. In fact, one woman living next to HEPP Park was particularly shocked and scared to see one deer near the woods in thirteen years, and the experience clearly stayed with her. In contrast, many respondents at both Chinquapin Run sites gave detailed descriptions of the enjoyment they derive from regularly watching deer.

5.4.2 Forest Patch Aesthetics

Aesthetic values of the neighborhood forest patches were another important theme, particularly among homeowners. Whether they were focused on the trees, deer, birds, water, or the snow in winter, many respondents valued the visual qualities of their forest patch. One man found inspiration in the beauty of community-managed Springfield Woods, saying “I don’t want to change it or rearrange it...’cause it’s a beautiful thing as it is, you know?” These and other accounts reveal strong feelings of place attachment related to the aesthetics of the forest patches. One woman homeowner feels a sense of ownership looking across at HEPP Park, saying “I just call it my greenery.” Another woman who intentionally chose her rental apartment for the view of Glen Oaks-Chinquapin Run forest patch enjoys watching the birds and squirrels from her window and says she “can just look up and see green right away.” The woods provides an important aesthetic backdrop to these residents’ daily lives, providing comfort, beauty, and inspiration.

Residents may be strongly attached to the aesthetics of local forest patches while still having no desire to physically interact with the space. A woman homeowner who explains that she has no reason to visit or explore the BCRP-managed Perring Loch-Chinquapin Run forest patch is still very much attached to its beauty, stating: “I enjoy the peacefulness of having that nice, wooded area over there. And when it snows, it’s nice. It’s very picturesque when it snows.” Another woman from the same neighborhood who doesn’t go into the woods because she is afraid of the animals that might be in there still enjoys sitting out on her porch in the evening “just watching the leaves go, the wind go through the leaves and stuff, it’s really nice.” The aesthetic qualities of the woods provide restorative benefits, even from a distance.

Some respondents chose to live in their home or apartment because of its proximity to the woods. These individuals were largely motivated by the aesthetic qualities of the forest patch. A man living near HEPP Park explained his motivation for buying a home across from the forest patch: “I grew up in a Baltimore row house, so this was like moving to the country.” Similarly, a woman explained that she bought her house across from HEPP Park because the setting was “picturesque.” Other respondents appreciated their view of the forest patch because of the privacy provided by the lack of neighbors across the street. One woman renter near the Glen Oaks-Chinquapin Run forest explained, “That’s one reason that I chose this apartment, because I wanted to be able to see out and not see out into somebody’s brick wall or their bathroom.” The sense of enclosure and seclusion provided by

urban wilderness is an aesthetic benefit that helps mitigate the crowded, public nature of urban life.

Despite the aesthetic benefits derived by many residents, a majority of respondents were dissatisfied with some aspect of the physical characteristics of their local forest patch vegetation. In particular, homeowners were more likely than renters to have concerns about the appearance or structure of forest patch vegetation. Often, the same residents who valued the beauty or other aesthetic benefits of the forest patch also had concerns with specific physical features of the woods and offered suggestions about how they could be improved, possibly because they spent so much time observing and appreciating the appearance of the forest.

Although many residents had concerns about trash being dumped in the woods, an even more prevalent aesthetic concern was that the forest patch vegetation itself needed some “clearing out.” One woman near Glen Oaks-Chinquapin Run felt that “some of the trees look a little scary...some of it looks weird...I’m talking about cutting the branches and making it look a little more presentable.” There was a perception that dead branches or trees should be removed, although dead and decomposing wood is important for forest ecosystem function, providing habitat and nutrients to many organisms. However, some residents perceived this natural cycle of death and decay as signs of neglect. One man observed of community-managed HEPP Park: “Those dead trees just falling apart there, nobody cares about it, nobody’s cleaning it.” Aside from detracting from the forest’s aesthetic qualities, other residents had safety concerns about the “clutter” they perceived in the forest understory. Two men felt that HEPP Park vegetation used to be in better condition,

but that the overgrown understory prevented anyone from entering the forest anymore. Respondents at both Chinquapin Run forest patches felt that additional “landscaping” would create more visibility, reducing the potential for crime and making parents feel more comfortable allowing their children to play in the woods. One woman renter in Glen Oaks remarked, “Kids can fall down in the back and nobody will know, because of the way the bushes is right there.” Another felt that if tree branches were removed, the forest patch would be “a little brighter for at least a parent or just anybody else to just see, that might be more appealing.” Though many residents see beauty in a patch of wild urban nature, some residents feel that more human intervention would improve the aesthetics of the forest patch and its associated value to the community.

Respondents were often able to identify ecological problems that led to changes to forest patch aesthetics, even if they did not use the scientific language to articulate their concerns. For example, residents from three different sites noticed that invasive vines were overtaking some of the canopy trees in their forest patch. They explained that the trees were dying as a result of the vines “strangling,” “choking,” or “sucking the life” out of the trees. A woman who is a long-time homeowner near the Perring Loch-Chinquapin Run forest remarked that “It’s been a lot of change since I’ve been here because there was more trees over there, and they were more beautiful...it was much thicker.” Similarly, a woman homeowner living near HEPP Park noticed that her forest patch had “thinned out over the years.” These women are likely noticing regeneration failure in the forest patches, caused by a combination of invasive plants and deer browse. As mature trees die, they are not being replaced with

a new generation of native trees, leading to a forest patch with a less dense canopy that is dominated by invasive vines, shrubs, and herbaceous plants.

5.4.3 Personal Identity

Many residents spoke about their identity when asked how they feel about the forest patch or whether or not they ever visit the space. The few residents who do venture into the woods had a particularly strong sense of identity related to their perceptions and use of the local forest patch. These respondents made statements like “I’m an outdoorsman,” “I am connected to the earth,” “I’m a strong advocate for the forest,” or “it’s something that I gravitate towards.” For these individuals, their connection to the forest patch is a manifestation of their environmental identity. Sometimes this place attachment relates to important landscapes from their past. A man from rural Maryland living in an apartment across from community-managed Springfield Woods recounts his childhood: “I’m from the woods, coming up, I used to play in the woods...it’s like second nature to me...I used to cut through the woods to go to school. I got my first kiss from a girl in the woods. We’d bring some food and hang out in the woods all day! That’s how I was brought up.” Another resident appreciates that the BCRP-managed Perring Loch-Chinquapin Run forest patch gives her access to “just a little bit of nature in the city because you don’t get that much because everything is so urban now. So that’s my thing. I love like woods and camping and stuff like that, so it kind of brings that into the city for me.” She appreciates living across from the woods because they connect her to previous outdoor experiences and pastimes that are part of her identity.

Residents who do not enter the woods but enjoy it from a distance, either by sitting nearby or taking a walk on the edge, still had strong emotions regarding the forest patch. A woman renter who likes to sit in the shade at the edge of Springfield Woods describes herself by saying “I’ve been a woods person for a lot of years.” Another woman renter who likes to sit at the edge of Glen Oaks-Chinquapin Run explains, “I’m a nature lover, I’m from the Caribbean so we always go hiking and stuff. It reminds me of home, having the trees, it reminds me of home. So, it’s the closest I get to home.” A woman who says she “love[s] nature and woodlands” feels that her family was “blessed” to be able to own a home across the street from Perring Loch-Chinquapin Run, where nature provides a background that improves their lives. For both those who regularly visit the woods and those who admire it from a distance, the local forest patch brings joy to their lives and helps them connect to their sense of self.

Residents do not appreciate living near the forest patch also invoked identity when explaining their distaste for nature. When asked why they haven’t ventured near the woods across the street, some residents explain that they just have no interest in experiencing the forest patch. One woman homeowner says of HEPP Park: “I never been over there, I know nothing about those woods. You know what y’all could put Santa Claus up in there...no I’m not going up in there. I have no reason to go up in there, no need.” Other respondents reveal that they don’t enjoy interacting with nature in general, or don’t view it as part of their identity. These residents say: “it’s not my thing” or “I’m not a woodsy person.” Some view the woods only as a place for kids. Several longtime residents describe their kids going into the forest patch even though

they would not enter themselves; or perhaps they did visit the woods as a child but have not been in decades. One man has “no need” to go into the HEPP Park woods because he “is not a kid anymore.” However, these respondents were much fewer in number than those who described a positive association between their identity and the forest patch.

Renters and homeowners were equally likely to talk about their identity being positively related to the local forest patch. This suggests that for homeowners, neither long-term investment in the local community nor observing the woods for many years will necessarily influence a resident to identify with these wild urban green spaces. Conversely, renters who may be more transient and do not necessarily have the same long-term investment in their surroundings are still able to form a deep connection to their local forest patch. The few residents who spoke about their identity when explaining their aversion to the forest patch were also comprised of renters and homeowners, suggesting that it is an innate social value that was brought to their current living situation.

5.4.3 Social Function

Varying perspectives on the social function of the forest patch comprise the final theme that emerged from the interviews. Residents had conflicting perspectives about the purpose of these green spaces and their use by the community. As they currently exist, the woods are quiet spaces that provide a unique opportunity for reflection and escape from the Baltimore City streetscape. The lack of intentional human design and structure allows children the opportunity for “free play” and lets visitors of all ages interact with wild nature. However, there is also a pervasive feeling that the woods

could be improved upon, and that additional development could help create more active sites of social recreation.

A common theme described by residents was the sense of refuge provided by their local forest patch, whether for a private social interaction, or for solitary reflection. One woman renter near the Glen Oaks-Chinquapin Run forest patch observes, “there’s often a young couple sitting there and talking. You know that’s a good private place for talking...the woods don’t tell.” Similarly, a woman homeowner near Perring Loch-Chinquapin Run woods explains “It’s nice because there’s nothing on the other side but the woods. And you can sit here, and you can relax, just watch and listen, and talk.” Others, such as this woman renter near Glen Oaks-Chinquapin Run, prefer to enjoy the woods alone: “In the summertime last year I used to always go over there and sit and...it’s peaceful. I love the trees. And I can meditate in there.” The woods can provide refuge from daily life, in a place that “doesn’t make you feel so much like you’re in the city.” Whether they are enjoying the forest patch alone or with another person, these respondents appreciate the fact that there are not a lot of other people nearby.

Many residents also mentioned the value of the forest patch to local kids, either for informal nature exploration and the opportunity for free play, or for educational activities like school projects. This theme was more prevalent in homeowner neighborhoods, possibly because those residents were more likely to live in multi-generational households where they had raised children or had grown up visiting the woods themselves. Residents described kids taking nature walks, skipping rocks, “exploring,” building treehouses, and catching tadpoles. One woman living in

an apartment near Springfield Woods explained, “We need more outdoors and something for kids to learn from. Not just in the classroom but somewhere to go and have field trips, and study nature.” Another woman who grew up next to the Perring Loch-Chinquapin Run forest patch recalled her childhood experiences fondly: “Not just me, but my sisters and the kids in the neighborhood, we would call ourselves camping, day camping, and we would take food down there and...we’d walk along in the creek.” The seclusion of the forest patch allows children to escape city life, fostering creative play and also interaction with the plants, animals, and other components of the forest ecosystem.

It is the wild qualities of these forest patches that lead to the recreational and inspirational benefits that both adults and children derive from these urban green spaces. However, some residents also criticized the lack of physical amenities in the space, discussing their preference for more benches, picnic areas, paths, playgrounds, and gardens. One woman said she might consider going into HEPP Park “maybe if it was more of a park feel. To me it’s just woods.” A few women explained that they might bring their children to the forest if there were more amenities, but right now there’s “nothing there.” If more formal structures were added to the forest patches, they might make additional residents feel welcome in the woods, but could also detract from the benefits they currently provide as sites of wild urban nature. One woman thought about entering HEPP Park but turned back “because it was kind of desolate in there.” Others did worry about the potential for crime in the woods and felt that lights or other modifications might improve visibility, making them feel more

comfortable. However, this “desolation” is exactly the quality that allows others to experience complete solitude and escape.

Forest patches on vacant land were less likely to provide residents with a sense of refuge, and were also less likely to provoke fear of nature. Although Springfield Woods and HEPP Park exist in the interstices of the urban landscape, periodic community stewardship activities and the lack of large wild animals (deer) may contribute to the perception of these sites as being less wild than the Chinquapin Run sites managed by BCRP. As a result, the forest patches on community stewarded vacant land did not evoke the strongest positive or negative feelings associated with wilderness.

5.5 Discussion and Conclusions

The data gathered in this study reveal that Baltimore residents may feel strong place attachment to their local forest patches and that many derive restorative benefits from these urban wild green spaces. However, the interviews also revealed deep ambivalence within each community and sometimes within individuals. These feelings of ambivalence are illustrated by residents’ discussion of personal identity, perceptions of wildlife, forest patch aesthetics, and desired social function. Although there were some differences in prevalence of the themes across homeownership and forest patch management categories, these relatively universal themes were expressed throughout the four case study sites. For many Baltimore residents, urban forest patches provide a type of wilderness experience, conveying the same lack of human influence as other sites of wild nature. In addition, these perceptions and experiences

of urban wilderness were described by many residents who did not actually enter the forest patch, but rather viewed it from their front stoop or passing along the edge.

Despite the fact that homeowner respondents had lived in their homes for much longer than renters, both groups of respondents were equally likely to relate their identity to the local forest patch. Renters are generally perceived as more transient, while homeowners may develop deep place attachment and place identity after many years (Taylor 1996; Brown et al. 2003; Rollwagen 2015). However, the interviews in this study suggest that innate social values are important in determining an individual's nature orientation, regardless of whether they have had long term regular exposure to a particular local green space. In fact, some renters suggested that they intentionally seek out wild nature in every location where they live, demonstrating a strong sense of environmental identity. Despite living near the woods for a shorter amount of time, these individuals describe a profound connection to the natural world and a resulting sense of personal fulfillment provided by the existence of their local forest patch (Clayton and Myers 2015). Conversely, a similar number of homeowners who had lived across from a forest patch for decades described their identity in opposition to nature and had no desire to interact with the green space. Illustrations of environmental identity and anti-environmental identity were found at all four sites, transcending neighborhood categories of homeownership and forest patch management.

Although situated in an urban landscape, Baltimore's forest patches are clearly providing opportunities to observe and interact with wildlife. This study showed that the physical characteristics of the forest patch can influence the amount

and type of wildlife encounters experienced by residents. Residents near forest patches of city parkland were more likely to discuss wildlife experiences, likely because the ecological connectivity of these sites provided habitat for deer populations. These close encounters were at times awe-inspiring and frightening to residents interviewed in this study. Consideration of plant and animal species can be a transformative and transcendent experience, providing an individual with the opportunity to consider what it means to be human (Clayton and Opatow 2003; Clayton and Myers 2015). In addition to contributing to an individual's sense of self, attitudes towards neighborhood wildlife experiences impact broader wildlife value orientations. Fulton et al. (1996) identified "residential wildlife experience" as one of the basic belief dimensions contributing to an individual's wildlife value orientation. Underlying these beliefs are fundamental values towards residential wildlife, which were demonstrated by many Baltimore residents in this study who considered the experience of seeing birds and other wildlife near their home to be an important part of their community. However, at times there was inherent conflict between the desirable and undesirable elements of wildlife perceived by residents. For example, ambivalent attitudes were demonstrated by residents who valued forest patch songbirds but not the insects they feed on.

Whether inside park boundaries or not, Baltimore's forests have a reputation for providing cover for criminals, leading some residents to view them as dangerous spaces. However, fear of nature itself was a much more prevalent theme in interviews at all four sites, driven by the "creepy" appearance of forest patch vegetation, the boldness of some urban wildlife, and the mysterious nature of urban wilderness.

These resident attitudes support Jorgensen and Tylecote's (2007) theory of urban wilderness as a unique type of space that exists in the gaps where human agency gives way to formative natural processes. In the highly controlled and ordered space that is Baltimore City, the appearance of wild nature is unexpected and unfamiliar to local residents. Yet, urban forest patches are hardly pristine nature, and the impact of pollution and other human disturbance is physically apparent to those who observe the space. Therefore, they exist as "interstitial spaces within capitalism's imperfectly formed spatial fabric" (Jorgensen and Tylecote 2007, p. 452). Both too wild and also too spoiled by human activity, the forest patches are not easily categorized as entirely urban or wilderness and provoke feelings of insecurity in some residents who perceive their unmanaged vegetation or wildlife as out of place or (paradoxically) unnatural. As described by Jones and Cloke (2002), forest landscapes are "other to civilization and modernity" and are constructed as contradictory places: "as savage and noble, as well as savage, backward and uncivilized" (p. 26). The contrast between forest and civilization is even starker in an urban environment; the forest patch offers an escape that is both a refuge from the mundane urban life, but also frighteningly lawless compared to the known stresses of the city. Indeed, wild nature leads people to reflect upon death and their own mortality (Koole and Van den Berg 2005), which may be why decaying vegetation is particularly unsettling to some Baltimore residents. However, it is precisely the tension between pleasure and terror caused by a sense of mystery and loss of control that leads to the sublime experience of nature (Herzog and Miller 1998; Van den Berg and Ter Heijne 2005).

Despite the ambivalence provoked by wild urban nature, many Baltimore residents demonstrated place attachment to their local forest patch, indicated by the strength of their emotional bond to the space. However, the place meaning that they ascribed to the space is an independent concept related to beliefs about what kind of space it is, and how it should be used and managed accordingly (Enqvist et al. 2017). As a result, there were differences in perception of forest patch aesthetics and desired social functions. For some, the woods in their current wild state fulfill a need for privacy and sense of refuge from city life. A similar number of respondents appreciate the forest patch from a distance but would like the forest interior to be more discoverable before they are willing to venture inside. These residents would prefer a forest patch that is less wild and supports increased visitation—one that is more strongly shaped by humans for human use, rather than the interstitial wilderness that results when nature is left to its own devices. Research has shown that the presence of company can enhance the restorative effects of nature by providing a feeling of safety, but that solitude enhances restoration when safety is not an issue (Staats and Hartig 2004). Furthermore, the absence of social feedback or judgment may contribute to relaxation in natural settings (Wohlwill 1983). Urban residents living near forest patches may experience enhanced autonomy due to the relative lack of social regulations, oversight, and consequences (Clayton and Myers 2015). Therefore, there are competing desires for the space to provide increased social functions for the community or to maintain their role in providing privacy and seclusion. Because individuals experience vulnerability in urban natural areas differently based on gender, age, and nature orientation, management interventions

that allow some people to feel more comfortable may limit others' ability to derive restorative benefits.

In addition to fears that messy vegetation might conceal danger, some residents were concerned about making the woods look "presentable." This language implies that the aesthetic qualities of the forest patch are a reflection of the local community. In contrast with other themes identified in this study, a majority of respondents described some dissatisfaction with the aesthetics of their local forest patch. Lacking visual cues that indicate active care, neighborhood residents may not feel that untended wilderness communicates positive social values to the broader public (Nassauer 1995a). Regardless of forest patch management regime, homeowners were more likely than renters to talk about both positive and negative aesthetic values of their local forest patch. With a long-term financial investment, homeowners may be more concerned with neighborhood reputation and its effect on their property values.

As a significant portion of Baltimore's tree canopy, forest patches are a critical component of the city's TreeBaltimore campaign, which works to increase urban tree canopy cover citywide. As municipal agencies and local nonprofits work to preserve and augment Baltimore's forest patches, it is important to understand how local residents perceive these wild urban green spaces, whether or not they actively use or steward the forest. Although homeownership and property management regime may lead to nuanced differences in the benefits and concerns related to forest patches across Baltimore neighborhoods, it is clear that homeowners and renters have strong and ambivalent feelings about these patches of urban wilderness on BCRP and vacant

land. Furthermore, this study demonstrates that Baltimore residents living adjacent to forest patches may experience some of the restorative benefits associated with immersion in wild nature, even when they do not actually enter the woods. Urban forest patches foster the development of place attachment and environmental identity in many individuals, providing unique and restorative aesthetic experiences and encounters with wildlife. However, these positive perceptions and experiences are balanced by negative emotions resulting from the perception that urban wilderness is chaotic and unpredictable. More human management of messy undergrowth and dead tree limbs would render the impenetrable woods more discernible but may also detract from their restorative wilderness qualities. Natural resource managers from city agencies and community non-profits must consider the variety of social values and experiences that lead to different preferences for urban forest patch aesthetics and social function.

Chapter 6: Conclusion

Urban forest patches and other natural areas are gaining recognition from policymakers as important sites of accessible nature worthy of conservation and restoration efforts (Forgione et al. 2016; Salbitano et al. 2016). As a significant proportion of urban tree canopy in cities of the eastern U.S., forest patches should not be overlooked in either scientific research or policy and planning efforts. More information about the social and ecological functioning of these green spaces will help establish their value to urban communities as sources of ecosystem services, laboratories in which to test scientific theory, and sites of nearby nature for environmental education benefitting growing urban populations.

Cities are thought to provide us a glimpse into future environmental conditions resulting from global change patterns. Like other aspects of global change, urbanization impacts tree growth and productivity, resulting in changes to forest tree species composition, nutrient cycling, hydrology, and biodiversity. An increased understanding of the effects of urbanization on the growth and physiology of important native tree species in forest patches will help ensure that urban forests continue to provide critical ecosystem services and help us understand the future structure and function of rural forests under future environmental conditions. The social functions of these wild urban green spaces provide similar insight into the future of eastern forests. Because forests in American suburban and rural landscapes are becoming increasingly fragmented, and because an increasing proportion of the

population lives in urban areas, both urban and rural Americans are more likely to interact with smaller forest patches than large protected areas.

In this dissertation, I set out to investigate the ecophysiological and social functions of urban forest patches of the eastern U.S. using a multidisciplinary approach combining diverse empirical methods. After carefully selecting urban and reference forest patch sites in New York City, Philadelphia, and Baltimore, I characterized the air temperature and soil environments of each site. Three years of air temperature monitoring revealed consistently warmer air temperatures at urban forest sites compared to reference forests, although the nighttime warming effect was not as strong as may be found in a downtown city center. Urban forest patch soils generally had more nutrients and more heavy metals than reference forest sites, although these differences were city and site dependent, which is not surprising given the variation in site history across different metropolitan areas.

Although I found variation by city and species and over time, red maple and white oak trees in urban forest patches generally had larger basal area increment (BAI) than reference forest trees, particularly in recent decades. This finding suggests that both native species are well-suited to the current environmental conditions of urban forest patches, which do not appear to be particularly stressful to healthy, established mature trees. Changes in BAI as well as the relative production of white oak earlywood and latewood may relate to differences in precipitation patterns across site types and over time within a given urban area. Higher white oak BAI despite lower latewood ratio (LWR) prior to 1970 suggests that the enhanced urban tree growth in NYC was temperature-driven and occurred despite the drier urban

conditions. Warmer springs at the NYC urban sites may have driven higher BAI and lower LWR. However, once annual precipitation began to increase, urban white oaks were able to increase their BAI even more relative to reference forest trees.

The differences in leaf-level physiological parameters between urban and reference trees were not as pronounced. However, it appears that red maple may be able to acclimate more readily than white oak to the urban forest patch environment. The Performance Index (PI_{abs}) of both tree species was higher in urban forest patches compared to reference sites, but red maple stomatal pore index (SPI) was also higher in urban trees, while white oak SPI was significantly lower in urban trees. Together, these physiological results suggest that the urban environment may allow for greater photosynthetic capacity in red maples, but not in white oaks. Furthermore, red maple thermal tolerance of photosynthesis (T_{crit}) was higher than that of white oaks in this study, which may mean that red maples will be better able to withstand temperature stress from the urban heat island effect and climate change.

The high values of T_{crit} found in the Baltimore urban white oaks compared to the other sites (which are all located further north) suggest that species suitability and response to the urban environment vary across a latitudinal gradient. Indeed, when Baltimore urban and rural air temperatures were used in the growth chamber experiment, the warm urban temperatures appeared to inhibit rather than promote overall growth and photosynthesis rates. The higher downtown Baltimore air temperatures used in the growth chamber may have been too stressful for the white oak seedlings compared to current urban forest patch temperatures. However, Baltimore urban forest patch soils did support significantly higher growth and

photosynthesis rates, likely due to the increased amounts of phosphorus and nitrogen in urban soils. There was also elevated manganese in the foliage of seedlings grown in reference forest soils, which may have toxic effects. Mature white oak trees from the field study similarly showed higher foliar P concentrations at urban sites and higher foliar Mn concentrations at reference sites, although the differences were more pronounced in the seedlings. Seedling physiological responses may be different from those of mature trees, but robust seedling growth is important to ensure the regeneration of native forests, particularly in the face of competitive invasive species.

Although both species are currently able to thrive in a variety of sites, results from the growth chamber study suggest that poor reference soils combined with warming air temperatures may cause physiological stress and reduced seedling growth rates. Previous research from Massachusetts has found soil warming to enhance growth rates of red maple seedlings (Wheeler et al. 2017), but these results may not apply consistently across the entire range of the species. Because white oak and red maple are both widespread throughout the eastern U.S., it is important to examine ecophysiological responses to urbanization across multiple urban areas with varying environmental conditions. In the face of continued climate change, it may also be beneficial to consider introducing tree species with more southern distributions that can withstand the elevated temperatures of urban forest patches.

The continued existence of urban forest patches is influenced by social, political, and economic forces just as strongly as biophysical conditions (Ogden et al. 2018). Community engagement with the forest patches may lead to advocacy for their protection and sustainable management. However, the results of my interviews with

Baltimore residents reveal that individuals do not necessarily need to enter or use these green spaces directly in order to appreciate them and derive restorative benefits associated with immersion in wild nature. I also found that these positive perceptions and experiences are balanced by negative emotions resulting from the perception that urban wilderness is chaotic and unpredictable. These feelings of ambivalence within communities and individuals are illustrated by residents' discussions of personal identity, perceptions of wildlife, forest patch aesthetics, and desired social function. Although there were some differences in prevalence of the themes across homeownership and forest patch management categories, these universal themes were expressed throughout the four case study sites.

In order to address negative perceptions and enhance benefits derived from these urban green spaces, natural resource managers must consider the variety of social values and experiences that lead to different preferences for urban forest patch aesthetics and social function. In addition, the social functions of urban forest patches will be context dependent, leading to variation across urban areas, just like the ecophysiology of the forest patch trees. Continued research on the social and ecological functions of urban forest patches will allow community groups, city agencies, and other environmental stewards to maximize social and biophysical benefits to urban communities.

Appendix A: Interview Protocol

- How long have you lived here (in this house)?
- What's it like living in this neighborhood? (Do you like living here?)
 - Do you feel safe in this neighborhood?
 - Can you tell me a bit about how the neighborhood has changed while you've lived here?
- Tell me about the woods over there.
 - What do you call that area?
 - How much did living near the woods affect your choice to live here?
- Do you ever go into the woods? Why or why not?
 - What do you (like to) do in the woods?
 - What determines when you go in there?
- Do you see anyone else go in there? If so who? (Gender, age, ethnicity)
 - Why do other people go in there? (or why not?)
 - Who takes care of that area? (are you a member of the organization?)
 - Who should take care of that area?
- Have you noticed any changes in the woods during the time you've lived here?
 - Any specific stories / memories (good or bad) about the woods?
- Is there anywhere else that you go in the outdoors?
- Do you ever do any volunteer work in the outdoors?
 - Where?
 - How often?
 - What activities do you participate in?
- Demographic Questions:
 - Are you: Employed? In school? Retired?
 - What level of school did you complete? HS / College / Graduate School
 - What is your age?
 - What do you consider your race or ethnicity?
 - Do you own the house or are you renting?

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