Influence of Climate, Construction Disturbance, and Site Factors on Tree Resistance, Resilience, and Performance in Urban Forests

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TABLE OF CONTENTS

ACKNOWLEDGEMENTS i

LIST OF TABLES v

LIST OF FIGURES vi

INTRODUCTION 1

CHAPTER 1 5

URBAN TREE GROWTH RESPONSE TO CLIMATIC FACTORS, AND RESISTANCE AND RESILIENCE TO DROUGHT 5

Introduction 6

Methods 8

Results 13

Discussion 19

Conclusion 26

Works Cited 26

CHAPTER 2 33

THE INFLUENCE OF SIDEWALK REPLACEMENT ON URBAN STREET TREE GROWTH 33

Introduction 34

Methods 36

Results 44

Discussion 50

Conclusion 55

Works Cited 56

CHAPTER 3 64

SITE CHARACTERISTICS AND TREE ARCHITECTURE AS INDICATORS OF URBAN TREE PERFORMANCE 64

Introduction 65

Methods 68

Results 73

Discussion 84

Conclusion 89

Works Cited 90

CONCLUSION 97

Works cited 99
Bibliography 104

LIST OF TABLES

Table 1.1. Mean age and DBH by species and site. Standard deviation is in parenthesis. 13

Table 1.2 Monthly climate variables significantly correlated with BAI for a 3-month growing season by species and site based on analysis using SEASCORR (α = 0.05). 15

Table 1.3. Best approximating models describing the impacts of climate variables and age on basal area increment by species within a site (α = 0.05)...16

Table 1.4. Mean tree age during two drought periods. Standard deviation is given in parenthesis...17

Table 2.1 Sample size and descriptive statistics of readable increment cores by construction activity (WC=with construction and NC=no construction) and species. 41

Table 2.2 Model coefficients, statistical significance, and AIC for fixed effects. Average PDSI during the growing season is denoted by the variable PDSI, con are the years of growth after sidewalk construction, the interaction between construction and PDSI, and population-construction which represents the population of trees in the WC group. 46

Table 3.1 Reference guides to variable abbreviations used in analysis. 73

Table 3.2 Descriptive statistics by species. Age in years, DBH in cm, height in meters, CPA in meters, BRATIO, and TPI. The standard deviation is presented in parenthesis...74

Table 3.3. Comparison of model coefficients, standard errors, and R^2 values for models examining the influence of site characteristics and tree defects on TPI, DBH, CPA, and BRATIO. 82

LIST OF FIGURES

Figure 1.1 Climate variables for Saint Paul and Minneapolis, Minnesota during the years 1982 through 2013. PDSI moderate drought is any point at or below the dotted horizontal line and severe drought is any point at or below the dashed horizontal line (National Centers for Environmental Information, 2016; Prism Climate Group, 2017). 10

Figure 1.2. Coefficient of variation of basal area increment by site (a) and species within a site (b) and comparison of mean BAI by site (c) and species within a site (d). Means with the same letter did not differ statistically based on the least-squares mean comparisons with a Tukey adjustment. Bars indicate a 95% confidence interval. ACPL = *A. platanoides,* CEOC = *C. occidentalis,* GLTR = *G. triacanthos,* TI = *Tilia* spp. N = 292..14

Figure 1.3. Comparison of overall drought resistance between the 1988 drought and the 2007 drought. Comparisons between sites and between species within a site for the two drought periods. Different letters indicate statistical difference using Tukey comparison of means. Bars indicate a 95% confidence interval. N=219. 18

Figure 1.4. Comparison of overall drought resilience between the 1988 drought and the 2007 drought. Comparisons between sites and between species within a site for the two drought periods. Different letters indicate statistical difference. Bars indicate a 95% confidence interval. N=219..19

Figure 3.1 The average growing season PDSI over the measurement period (National Centers for Environmental Information, 2016). 40

Figure 3.2 Growth trends in basal area increment (BAI) by species and disturbance group over the study period. Error bars indicate one standard error. (N=292) 42

Figure 3.3 Comparison of modeled basal area increment (BAI) by species for pre-construction and post- construction. Error bars indicate one standard error. N = 292, trees. N = 6766, years of growth. 47

Figure 3.4 Indices of resistance and resilience to sidewalk construction by species. Bars indicate one standard error. Trees, N=142. 48

Figure 3.5 Mean growth response for 5 years after sidewalk construction by species; bars indicate one standard error. Tree, N=142. Year of growth, N=710. 49

Figure 3.6 Recovery index by years after sidewalk construction, species, and width of planting space. Bars indicate one standard error. Trees, N=142. Years of growth, N=710 50

Figure 4.1. Age by site, species, and site:species interactions. Different letters indicate statistically different means. *Acer platanoides* (ACPL), *C. occidentalis* (CEOC), *G. triacanthos* (GLTR), and *Tilia* spp. (TI). Park trees (PT) and street trees (ST). (N = 292) 75

Figure 4.2. DBH by site, species, and site:species interactions. Different letters indicate statistically different means. *Acer platanoides* (ACPL), *C. occidentalis* (CEOC), *G. triacanthos* (GLTR), and *Tilia* spp. (TI). Park trees (PT) and street trees (ST). (N = 292) 76

Figure 4.3. CPA by site, species, and site:species interactions. Different letters indicate statistically different means. *Acer platanoides* (ACPL), *C. occidentalis* (CEOC), *G. triacanthos* (GLTR), and *Tilia* spp. (TI). Park trees (PT) and street trees (ST). (N = 292) 77

Figure 4.4. Height by site, species, and site:species interactions. Different letters indicate statistically different means. *Acer platanoides* (ACPL), *C. occidentalis* (CEOC), *G. triacanthos* (GLTR), and *Tilia* spp. (TI). Park trees (PT) and street trees (ST). (N = 292) 78

Figure 4.5. Basal area increment (BAI) ratio by site, species, and site:species interactions. Different letters indicate statistically different means. *Acer platanoides* (ACPL), *C. occidentalis* (CEOC), *G. triacanthos* (GLTR), and *Tilia* spp. (TI). Park trees (PT) and street trees (ST). (N = 292) 79

Figure 4.6. Tree performance index (TPI) by site, species, and site:species interactions. Different letters indicate statistically different means. *Acer platanoides* (ACPL), *C. occidentalis* (CEOC), *G. triacanthos* (GLTR), and *Tilia* spp. (TI). Park trees (PT) and street trees (ST). (N = 292) 80

Figure 4.7 Model output for mean TPI, DBH, CPA, and BAI ratio by species on pervious surface area, based on model results presented in Table 4. Grey bans represent a 95% confidence interval. *C. occidentalis* (CEOC), *G. triacanthos* (GLTR), and *Tilia* spp. (TI). N=292 83

INTRODUCTION

Urban forests are composed of trees growing in city parks, in natural areas within city boundaries, along city streets, and trees on private land. Research investigating the links between human health (Jiang, et al., 2014; Lee, et al., 2009), energy reduction, pollution mitigation (McPherson, et al., 1997; Nowak D. J., 1993; Brack, 2002), water runoff attenuation (Bartens, et al., 2008), and economic benefits to landowners (Sanders, et al., 2010) has increased as the focus of urban forestry has shifted from management concerned primarily with tree aesthetics to management designed to maximize the benefits trees provide (McPherson E. G., 2006). Trees in poor health and condition do not grow as large (Koeser, et al., 2013; Lee, et al., 2014) and smaller, shorter lived trees provide fewer benefits compared to larger, longer lived trees (McPherson E. G., 2003; Scott & Betters, 2000).

As urban areas increase in physical size, distribution, and population, increased importance will be placed on the benefits provided by trees in the urban forest (Dwyer, et al., 2000). Appropriate tree species selection based on environmental criteria is critical in order to achieve the larger and longer lived trees (Chacalo, et al., 1994) needed in urban landscapes. An analysis of urban planting sites can be difficult and costly when dealing with the highly variable nature of the urban environment. Site analysis approaches that model tree growth using easily obtainable site variables, such as soil surface, planting space width, and regional climatic factors are crucial for developing species specific site selection criteria designed to maximize tree size and longevity.

Tree longevity is a function of species, adjacent land use, and tree health (Nowak, et al., 2004). The long-term performance of urban trees is a complex issue involving many

factors from species to site conditions to climate factors and construction activities. Yet our current understanding of appropriate tree selection to maximize benefits over the long-term is limited.

There is a gap in the literature on the long-term influence on tree growth and performance following infrastructure construction activities, and the effect of urban site characteristics. While climate in the urban environment is well documented over the life-span of the majority of living urban street and park trees (~28 years on average (Roman & Scatena, 2011)), the impacts to tree growth related to climate in combination with urban environmental factors is less well understood. Larger trees have been shown to provide greater benefits (McPherson E. G., 2003; Scott & Betters, 2000), yet our understanding of what constitutes a "larger" urban tree in terms of specific benefits has not been well defined.

The research in this dissertation examined the influence of climate, construction, and physical environment on the growth of urban trees primarily through the use of tree ring analysis for trees in the cities of Minneapolis and Saint Paul, Minnesota. Chapter 1 examines the influence climate over the past 30 years on the growth of municipally-managed street and park trees. Chapter 2 investigates the effect of sidewalk construction on growth of street trees that survived the initial construction. Finally, chapter 3 evaluates four different approaches to quantifying tree performance in urban environments.

Minneapolis and Saint Paul municipal forestry departments provided inventory data for their managed park and street trees to allow for selection of genera, species, and individual trees planted in both cities. *Acer platanoides, Celtis occidentalis, Gleditsia triacanthos* and *Tilia* spp. were selected as tree genera and species commonly planted in

Minnesota and elsewhere in the Midwest of the United States. The results presented in Chapter 1 compare growth of trees in parks to trees growing along city streets using annual basal area increment determined from tree cores. Additionally, the chapter examined the influence of precipitation and mean monthly temperature growth over the period of 1982 to 2013 and identified the monthly climate variables that significantly influenced growth. Differences in growth among species and between sites (i.e. parks and along streets) were found as well. The period between 1982 and 2013 also contained two drought events and the resistance and resilience of species was quantified with significant differences occurring in response to the two droughts that were linked to tree age and tree size.

In chapter 2, the effect of sidewalk construction on tree growth was quantified in terms of mean basal area increment (BAI), resistance, resilience, and recovery. As in Chapter 1, differences in growth response were detected among the four species investigated. Resistance and resilience showed statistically significant differences among species. To further quantify the growth impact of construction, time to recovery was also examined as the number of years post-construction needed for a species to regain previous levels of basal area growth. The interaction of boulevard or planting space-width on tree recovery was analyzed and found to have significant impact on growth recovery for all species.

Finally, to investigate the influence of urban site characteristics on tree growth and performance, four separate measures of tree growth were analyzed: canopy projection area (CPA), diameter at breast height (DBH), growth rate as a ratio, and a tree performance index (TPI). The TPI was created in an attempt to include DBH, CPA,

height, and growth rate into a single metric defining tree performance in urban environments. Species performance differed based the metric analyzed. The amount of pervious surface area under the canopy had a consistent positive influence on all species regardless of the performance metric investigated. Other factors influencing growth included: damage to tree trunk, the presence of stem girdling roots, nearness of neighboring trees, and tree age (all of which varied in significance, direction, and magnitude of effect on based on the performance metric analyzed). This chapter discusses the complexity in assessing performance and highlights the importance of identifying the primary objective as part of the tree selection process.

CHAPTER 1

URBAN TREE GROWTH RESPONSE TO CLIMATIC FACTORS, AND RESISTANCE AND RESILIENCE TO DROUGHT

Introduction

The urban forest will likely play an increasingly important role as urban populations continue to grow worldwide. For example, 82 percent of the population live in urban areas in North America (United Nations, 2015) with projected increases in global climate variability expected to result in greater frequency of drought events and increased temperatures in these and other regions of the globe (Schar, et al., 1998). The ability of urban trees to mitigate some of the deleterious effects of increased climate variability (McPherson, et al., 1997) will depend on maintaining large, mature, and healthy trees (McPherson E. G., 2003). Managers of urban forests will need to understand the influence of climate on tree growth in order to appropriately manage trees for future benefits provided to increasing urban populations.

While many studies have investigated the influence of temperature, precipitation, and drought on forest trees, research has only recently started to examine the response of urban trees to climate. The urban forest is comprised of trees growing under highly variable environmental conditions that produce a wide range of growth responses based on species and site interactions (Iakovoglou, et al., 2001; Quigley M. E., 2004). Recent research has used tree ring analysis to investigate the growth response of urban trees to climate variables (Chen, et al., 2011; Gillner, et al., 2014; Fahey, et al., 2013). Chen, et al. (2011) investigated tree growth response to climate factors along a gradient of rural to urban areas. Trees growing closer to urban areas were less sensitive to changes in climate (i.e. precipitation and temperature) and more sensitive to anthropogenic environmental changes (Chen, et al., 2011). Gillner, et al. (2014) detected differences in climatic growth response based on species and site conditions. Trees growing in highly-sealed (i.e. roots

systems covered by non-porous material) were more sensitive to drought than trees growing in less sealed environments (Gillner, et al., 2014).

Precipitation and temperature during the growing season have been widely demonstrated as the primary climate variables influencing tree growth for most tree species in forests around the globe (Kipfmueller, et al., 2010; Dymond, et al., 2016). This relationship has also been demonstrated in the limited studies conducted in urban conditions with precipitation during the growing season exhibiting a strong positive relationship with urban tree growth (Cedro & Nowak, 2006; Monteiro, et al., 2017). High temperatures during the growing season can reduce urban tree growth, particularly in interaction with precipitation resulting in low levels of growth during warm, dry periods (Zweifel, et al., 2006; Cedro & Nowak, 2006; McLaughlin, et al., 2003; Monteiro, et al., 2017). Work in forested settings has demonstrated the influence that local site conditions have on climate sensitivity within a tree species (Gewehr et al. 2014); however, few similar investigations have been conducted in urban forests. Given the range of growing environments in urban areas, an understanding of the impacts of local growing conditions on climate response of urban trees is critical for evaluating the vulnerability of urban forests to future global change.

This study was designed to address information gaps on how the growth of publicly-managed urban park and street trees varies across site conditions and in response to climatic factors. To assess the influence of site and climate factors on tree growth and to determine growth resistance and resilience to drought events we posed three research questions: 1) how does the variation in tree growth differ between sites and species? 2)

what is the influence of temperature and precipitation on tree growth? 3) how do resistance and resilience to two distinct drought events differ by species and site?

Methods

Living trees were sampled in publicly-managed parks and in the right-of-way in the cities of Saint Paul and Minneapolis, Minnesota, USA. Using the combined inventory data of both cities, the species *Acer platanoides, Celtis occidentalis,* and *Gleditsia triacanthos* were selected as publicly managed trees common throughout both cities. The genus *Tilia* was also selected, but due to inconsistencies in identification of hybrid and cultivated varieties trees were not identified to species.

Park trees (PT) included in the study were considered to have unrestricted growing space beneath the canopy with no impervious surfaces at a minimum of 1m from the canopy dripline. All PT were growing in managed municipal parks where there was evidence of tree and landscape maintenance (i.e. pruning and lawn mowing). Street trees (ST) in the right-of-way were growing between sidewalk and curb on residential non-arterial streets. Street trees were considered to have restricted growing space beneath the canopy. A minimum diameter at breast height (DBH) of 25cm was set in an attempt to sample only trees that were at least at least 20 years old (Frelich, 1992) and established in their environments (Sherman, Kane, Autio, Harris, & Ryan, 2016). There was no upper limit imposed on DBH. Genus and species were field-verified by researchers with no attempt to identify individual trees to variety or cultivar.

One increment core was obtained from each tree at 0.5m from the ground using a Haglöf, 4.30mm core, increment borer. Cores were mounted and sanded to produce a flat surface. Each core was aged and dated by two researchers by counting the rings from

bark to pith. Where pith was not visible, tree age was estimated using a series of concentric circles based on ring curvature to approximate the number of non-visible rings (Applequist, 1958). Tree cores lacking a visible pith and with insufficient ring curvature to provide a reasonable approximation of age were not used in the analysis. Ring-width was measured using a Velmex Tree-Ring measurement system and the software J2X to the nearest 0.001 mm. Diameter at breast height was measured at 1.3m to the nearest 0.1cm using a diameter tape. All data were collected between the months of June and August in 2014 and 2015.

Climate variables used in analysis included mean monthly temperature in degrees Celsius (T) and total monthly precipitation (P) in centimeters for the years 1982 – 2013 (Prism Climate Group, 2017). Drought years were identified using the mean Palmer Drought Severity Index (PDSI) between April and October for the years 1982 through 2013 (National Centers for Environmental Information, 2016). The PDSI is an index of relative moisture for a region and is derived through the use of weekly total precipitation, mean temperature, and measures of soil water holding capacity. Levels of PDSI between -1.9 and +1.9 are considered normal with levels -2.0 or less range from moderate to extremely dry, and +2.0 or higher range moderate or extremely wet (National Weather Service, 2005). The PDSI was used to identify drought because it incorporates precipitation, temperature, and soil water holding capacity which was believed to be a more accurate measure of moisture deficits affecting tree growth. There were two periods of moderate to severe drought as identified by PDSI and defined by the National Centers for Environmental Informational occurred in the study area: 1987 to 1989 (D1), and 2007 (D2) (Figure 1.1).

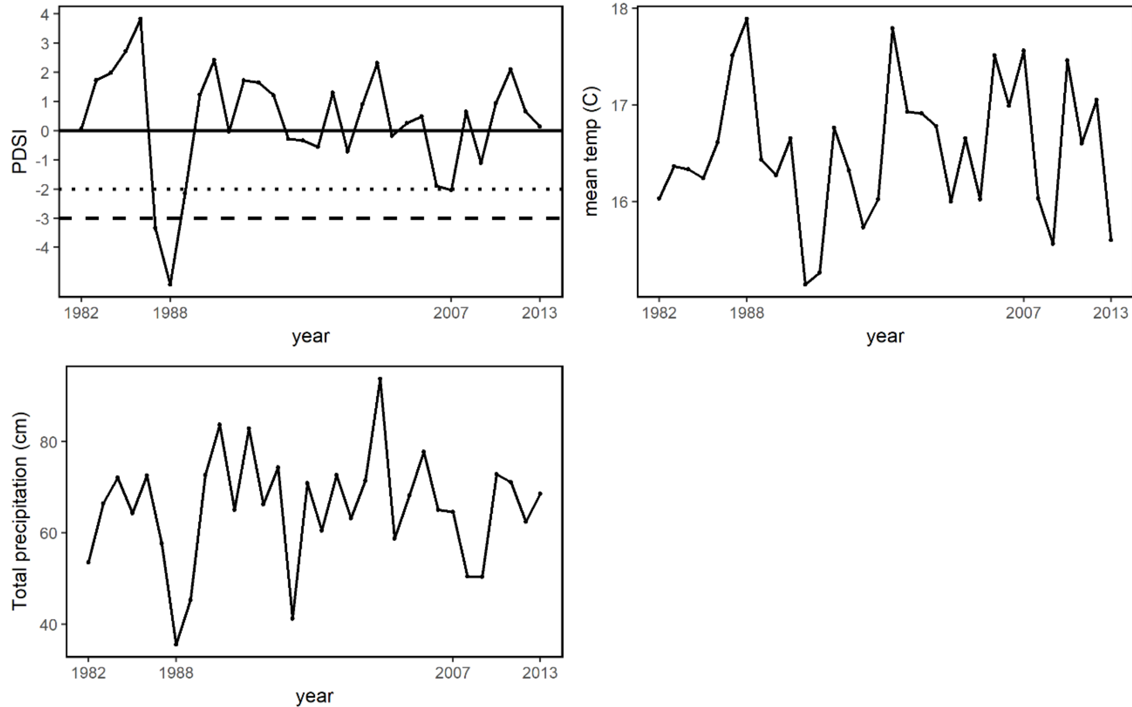

Figure 1.1 Climate variables for Saint Paul and Minneapolis, Minnesota during the years 1982 through 2013. PDSI moderate drought is any point at or below the dotted horizontal line and severe drought is any point at or below the dashed horizontal line (National Centers for Environmental Information, 2016; Prism Climate Group, 2017).

Data analysis

Basal area increment (BAI) was calculated to assess tree growth and was determined from tree-ring width measurements using R (R Core Team, 2016) with the dplR package (Bunn A. G., 2010). Only the most recent 31 years of growth (1982-2013) were used in the analysis.

Variation in growth was assessed using the coefficient of variation (CV) of BAI over the 31-year period for each tree. A one-way ANOVA was conducted to assess the difference in CV or BAI between sites and a two-way factorial ANOVA was conducted to compare differences in mean CV of BAI between species within a site. An ANOVA

was conducted to assess the differences in mean BAI between trees within a site and between sites.

To investigate the relationships between tree growth and monthly local climate, chronologies based on BAI for species by site were created using the dplR package in R (Bunn A. G., 2010). Seasonal correlations between climate and growth were analyzed using the SEASCORR function from the treeclim package in R (Zang & Biondi, 2015). SEASCORR analysis was run on species by site to determine which monthly climate variables significantly influenced growth (Dymond, et al., 2016). October was assumed as the last month of growth with a moving 3-month window (e.g. current August to October, July to September, etc.). Correlations of total month precipitation (cm) and partial correlations of mean monthly temperature (ºC) after accounting for the influence of precipitation on BAI were assessed.

Generalized least squares models were constructed using the gls function in R (Pinherio, Bates, DeRoy, Sarkar, & R Core Team, 2016) to investigate which of the climate variables, indicated as significant in the SEASCORR analysis, most effectively explained variation in BAI while accounting for tree age. To account for the non-random and correlated nature of tree ring data (Fritts, Tree rings and climate, 1976), a first-order autoregressive covariance structure was added to the models. Heteroscedasticity in residuals was addressed with a power function of DBH to stabilize residual variance. Models were compared using the monthly values and composite climate variables of either total precipitation over the significant period or mean temperature over the significant period. The best approximating models for explaining the influence of climate

variables and age on tree growth were selected based on Akaike information criteria (AIC) and minimized residual standard error.

Tree growth pre-and post-drought were compared using two indices: resistance and resilience. Resistance to drought is a species' ability to maintain growth during the drought period, while resilience to drought is a species' ability to regain previous levels of growth post drought (D'Amato, et al., 2013; Fahey, et al., 2013). Species-level drought resistance (D_{rs}) and drought resilience (D_{rl}) indices were created based on the techniques employed by D'Amato, et al. (2013) and Fahey, et al. (2013).

Species-level drought resistance was defined as $D_{rs} = BAI_D / BAI_{pre}$ and where species-level drought resilience was defined as $D_{rl} = BAI_{post} / BAI_{pre}$. BAI_D is the species-level average BAI during the drought period and BAI_{pre} is the species-level average BAI for the five calendars years preceding the drought period and where BAI_{post} is the species-level average BAI for the five-calendar years post-drought period. Values of D_{rs} and D_{rl} greater than 1 indicate resistance or resilience to drought. Two-way factorial ANOVAs were conducted to compare the differences in mean resistance and resilience indices by drought period (D1 and D2) and site (PT and ST). Two-way ANOVAs were also conducted to compare the differences of mean resistance index and mean resilience index on species and site interactions over the defined drought periods. In cases in which significant main effects were detected, Tukey adjustments were applied to the means comparisons and the significance level was set at 0.05. All statistical analyses were conducted in R (R Core Team, 2016).

Results

Of the 360 trees observed, 292 had observable rings that spanned the two drought periods and were used in the analyses. Table 1.1 provides details on the species and site statistics for the trees analyzed.

Table 1.1. Mean age and DBH by species and site. Standard deviation is in parenthesis.

Species	park trees			street trees		
	n	age	DBH (cm)	n	age	DBH (cm)
ACPL	37	38 (11)	42.5 (10.2)	37	38 (5)	44.1 (6.6)
CEOC	35	64 (29)	49.7 (10.7)	37	36 (7)	44.7 (7.6)
GLTR	37	41 (17)	38.3 (11.0)	38	37 (6)	47.5 (9.2)
TI	34	36 (14)	45.0 (10.6)	37	34 (8)	44.9 (10.8)
all species	143	44 (22)	43.8 (11.3)	149	36 (7)	45.3 (8.7)

Note: ACPL = *A. platanoides,* CEOC = *C. occidentalis,* GLTR = *G. triacanthos,* TI = *Tilia* spp.

Park trees showed greater variation in BAI (CV of BAI = 62%) as compared to street trees ($p = 0.0056$; Figure 1.2a). In parks, *C. occidentalis* had the lowest mean CV of BAI of 0.4484 ($p < 0.0001$) compared to other park species (Figure 1.2b). Mean CV of BAI for all other park species did not differ statistically. *Tilia* spp. had the highest mean CV of BAI of 68% compared to other street tree species ($p < 0.05$). Mean CV of BAI between other street tree species did not differ statistically (Figure 1.2b). Park trees had a lower annual basal area increment compared to street trees ($p < 0.0001$; Figure 1.2c). In parks, *Tilia* spp. exhibited greatest growth in terms of mean BAI and differed significantly from both *C. occidentalis* and *G. triacanthos* ($p = 0.0005$, $p < 0.0001$; respectively; Figure 1.2d). *Gleditsia triacanthos* also showed lower mean BAI compared to *A. platanoides* ($p = 0.0027$; Figure 1.2d). For street trees, only *Tilia* spp. had

significantly higher mean BAI compared *A. platanoides* and *G. triacanthos* (p < 0.0001, p = 0.0293; respectively; Figure 1.2d).

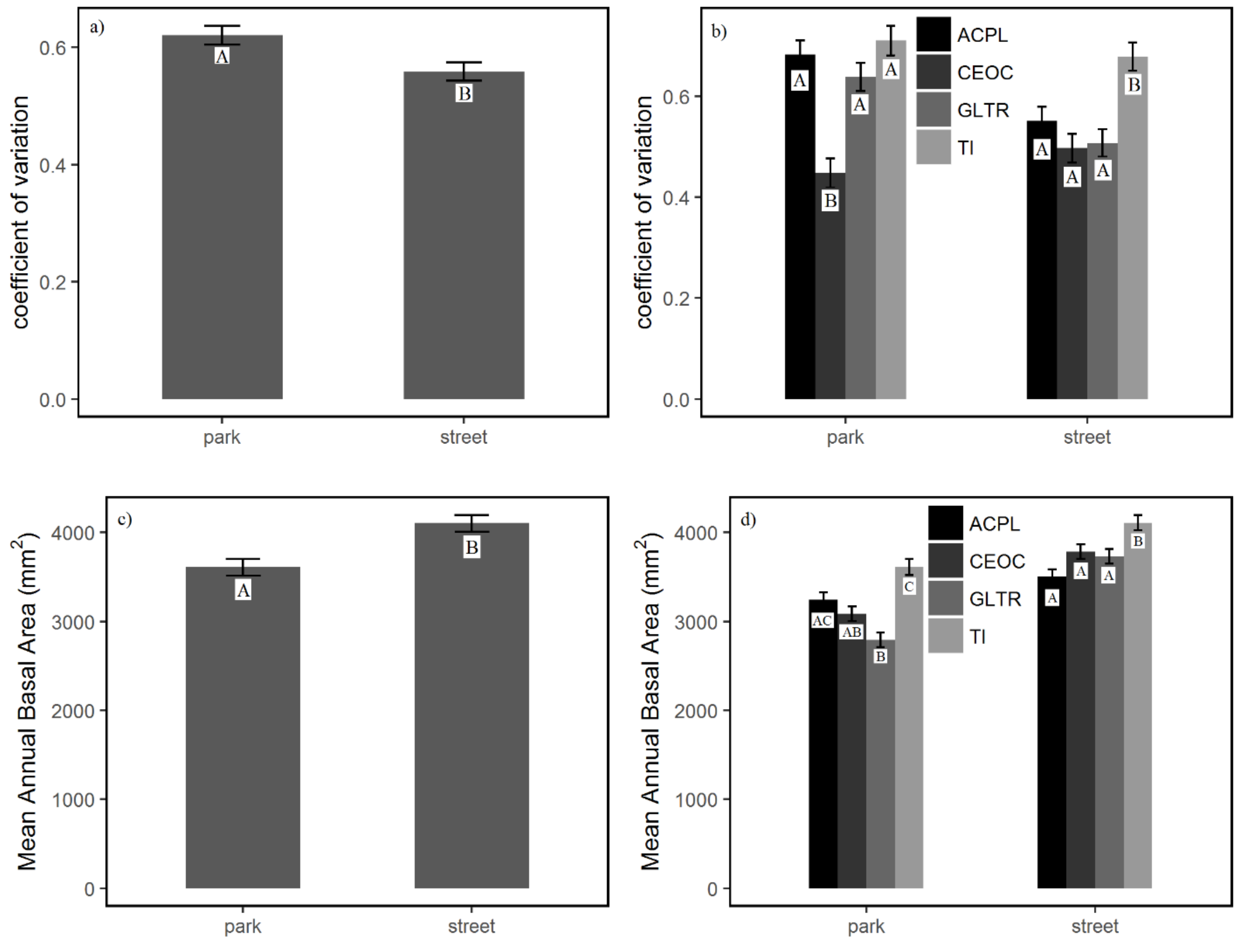

Figure 1.2. Coefficient of variation of basal area increment by site (a) and species within a site (b) and comparison of mean BAI by site (c) and species within a site (d). Means with the same letter did not differ statistically based on the least-squares mean comparisons with a Tukey adjustment. Bars indicate a 95% confidence interval. ACPL = *A. platanoides,* CEOC = *C. occidentalis,* GLTR = *G. triacanthos,* TI = *Tilia* spp. N = 292.

Several climate variables significantly influenced tree growth for the species examined at each site (Table 1.2). Temperature was only a significant factor influencing growth in *A. platanoides* growing as street trees, *C. occidentalis* growing in parks, and *G. triacanthos* growing in parks. For *A. platanoides* the mean previous September

temperature was a significant factor. No climate factors were found to significantly influence *Tilia* spp. growing in parks and only December precipitation had a positive, significant influence on *Tilia* spp. growth along streets. Precipitation significantly influenced growth in a positive fashion in all cases, whereas mean monthly temperature, when significant, had a negative influence on growth in all cases expect during September of the current year for *G. triacanthos* growing in parks.

Table 1.2 Monthly climate variables significantly correlated with BAI for a 3-month growing season by species and site based on analysis using SEASCORR ($\alpha = 0.05$).

Species	Site	Climate Variables
A. platanoides	park	$P_{Jun,}$, P_{Jul}, P_{Aug}
	street	$P_{Jul,}$, P_{Aug}, $P_{Sep,}$ $\mathbf{T_{pSep}}$
C. occidentalis	park	P_{Dec}, $P_{Jun,}$, P_{Jul}, $P_{Aug,}$ $\mathbf{T_{May}}$
	street	P_{Dec}, $P_{Jun,}$, P_{Jul}, $P_{Aug,}$ $P_{Sep,}$
G. triacanthos	park	P_{Dec}, $P_{Jun,}$, P_{Jul}, $P_{Aug,}$ $\mathbf{T_{May}}$, T_{Sep}
	street	P_{Jul}, $P_{Aug,}$
Tilia spp.	park	no variables identified as significant
	street	P_{Dec}

Note: Bold variables indicate negative correlation between the variable and BAI. P = total precipitation (cm), T = mean temperature (ºC). The subscript month preceded by p, indicates the influence of the previous year's climate variable on current year's growth.

A series of candidate models were constructed to predict BAI based on tree age and climate factors identified as influential from the SEASCORR analysis (Table 1.3). Models constructed with total precipitation over contiguous and significant months had lower AIC and residual standard errors compared to models using individual monthly precipitation values (Table 1.3). In all models, tree age had a positive influence on BAI and had the greatest magnitude compared to the climate variables modeled. Although no climate factors were identified by SEASCORR for park *Tilia* spp. and only December

precipitation influenced street *Tilia* spp, mean temperature (April-October) and total precipitation over the same period were tested against individual monthly values and against the composite combinations used in the other species analyses. For all of the species in both site types precipitation in July and August were identified as important climate factors influencing growth. Mean temperature in May had a significant positive impact on growth of *C. occidentalis* in parks and a significant negative impact on growth of *G. triacanthos* in parks. Growth for *A. platanoides* along streets was positively influenced by higher temperatures in the previous year. With the exception of the interface and age for park *C. occidentalis* parameter estimates for the variables in the final models were statistically significant.

Table 1.3. Best approximating models describing the impacts of climate variables and age on basal area increment(mm^2) by species within a site ($\alpha = 0.05$).

Species	Site	Final model	RSE (mm^2)
A. platanoides	park	BAI = **-109.67** + 2.17*$P_{Jun+Jul+Aug}$ + 95.36*age	79.42
	street	BAI = -1282.70 + 2.27*$P_{Jul+Aug+Sep}$ + 103.63*T_{pSep} + 87.22*age	66.62
C. occidentalis	park	BAI = 3864.38 + 1.35*$P_{Jun+Jul+Aug}$ -109.35+T_{May} + **6.26*age**	367.37
	street	BAI = **-213.52** + 1.82*$P_{Jun+Jul+Aug+Sep}$ + 134.54*age	36.46
G. triacanthos	park	BAI = 1412.44 + 1.54*$P_{Jun+Jul+Aug}$ – 51.66*T_{May} + 60.34*age	32.09
	street	BAI = **-144.97** + 1.19*$P_{Jun+Jul+Aug}$ + 124.62*age	34.34
Tilia spp.	park	BAI = 973.20 + 4.95*$P_{Jun+Jul+Aug}$ + 75.87*age	44.52
	street	BAI = -417.07 + 5.60*$P_{Jun+Jul+Aug}$ + 156.15*age	54.76

Note: BAI = basal area increment (mm^2), $P_{n+...+}$ = total precipitation (cm) for the significant months, RSE = residual standard error for the model. Variables in bold parameter estimates in bold were non-significant.

Drought resistance

Average tree age by site during the two drought periods can be seen in Table 1.4. Park trees were older on average than streets trees with greater variability in age (Table 1.4).

Table 1.4. Mean tree age during two drought periods. Standard deviation is given in parenthesis.

site	N	Age	drought
park	104	27 (21)	1988
street	115	14 (4)	1988
park	104	45 (21)	2007
street	115	33 (4)	2007
park & street	219	19 (16)	1988
park & street	219	35 (16)	2007

Resistance to drought differed significantly between the two drought periods ($p < 0.0001$) with trees in the 1988 drought exhibiting mean resistance index values greater than 1, indicating resistance to drought and trees with mean resistance index values less than 1 for the 2007 drought (Figure 1.3a). Trees in parks had a lower mean resistance index compared to street trees during the 1988 drought ($p < 0.0001$; Figure 1.3b); both sites had mean resistance index values greater than 1. Mean resistance index were below 1 for both park and street trees for the 2007 drought ($p = 1.000$; Figure 1.3b). All species had mean resistance index values greater than 1 for both park and street trees in the 1988 drought with no significant difference in resistance values (Figure 1.3c). The mean

resistance index value in parks was the highest for *G. triacanthos* (0.94) and differed significantly ($p < 0.0001$) from *A. platanoides* for park trees in the 2007 drought (Figure 1.3c). For street trees during the 2007 drought mean resistance index of *G. triacanthos* was 0.97 and differed significantly from both *A. platanoides* and *Tilia* spp. ($p = 0.0002$ and $p = 0.0037$, respectively; Figure 1.3c).

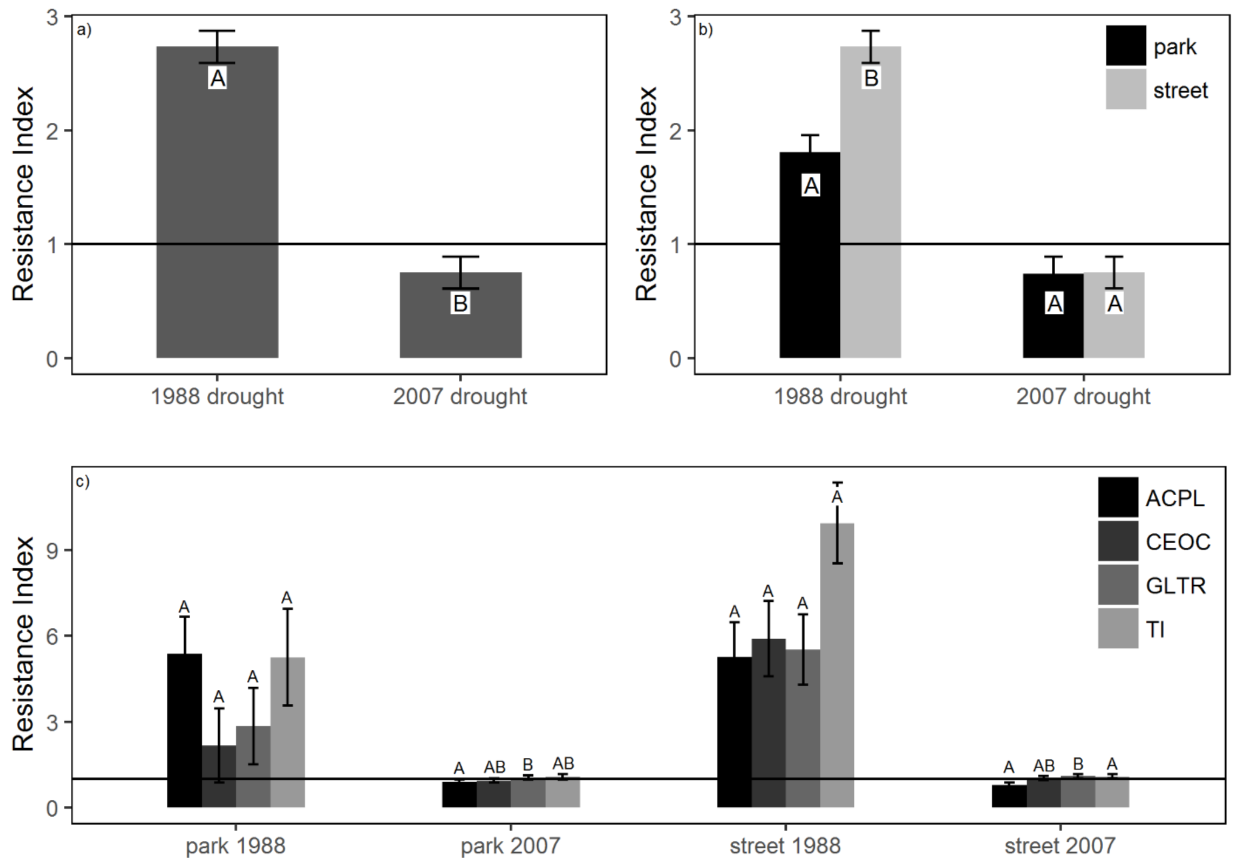

Figure 1.3. Comparison of overall drought resistance between the 1988 drought and the 2007 drought. Comparisons between sites and between species within a site for the two drought periods. Different letters indicate statistical difference using Tukey comparison of means. Bars indicate a 95% confidence interval. N=219.

Drought resilience

Trees showed high resilience to the 1988 drought and moderate to low resilience to the 2007 drought (Figure 1.4a). Mean resilience index differed significantly between

the 1988 drought and the 2007 drought (p = 0.0000; Figure 1.4a). Street trees had a higher mean resilience index than park trees for the 1988 drought (p = 0.0005) and there was no statistical difference in mean resilience index between sites for the 2007 drought (p = 0.9999; Figure 1.4b). There was no statistical difference in mean resilience index between species within sites for either drought period (p > 0.1; Figure 1.4c).

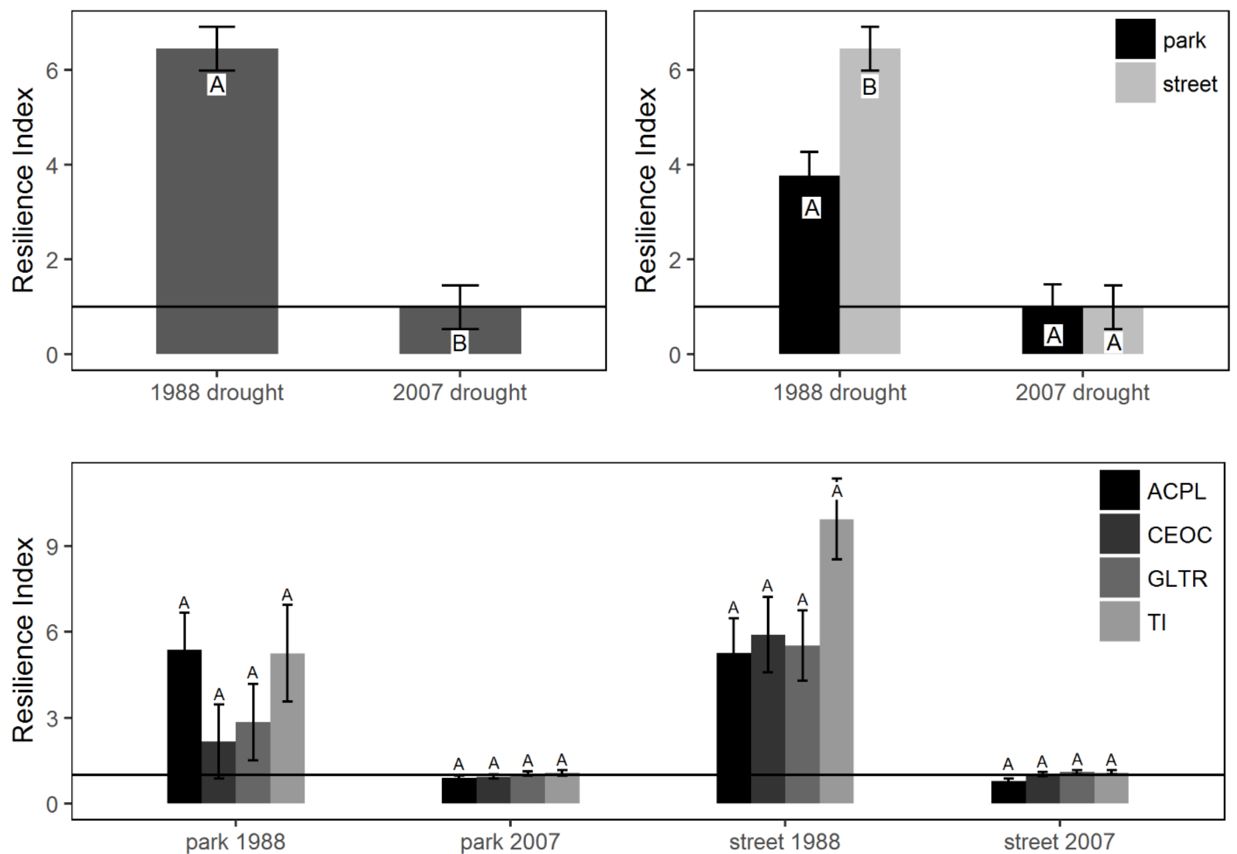

Figure 1.4. Comparison of overall drought resilience between the 1988 drought and the 2007 drought. Comparisons between sites and between species within a site for the two drought periods. Different letters indicate statistical difference. Bars indicate a 95% confidence interval. N=219.

Discussion

As future climate is expected to change, understanding species response to current climate factors will likely prove critical for selection of tree species that have a greater

resilience to climatic changes. On average, park trees showed statistically greater variability in mean BAI (Figure 1.2a). Monterio et al. (2017) showed a similar trend in location and species variable growth response, whereas, Fahey, et al. (2013) showed no statistical difference in variation between sites. Quigley (2001) also reported differences in tree growth by location. In our findings, the higher variability in park sites was unexpected as the park trees were all growing in unrestricted space and growth was expected to have less overall variability compared to street trees. Street trees had a wider range in the amount of available soil surface area under their crowns yet their overall variability was less than that of park trees. One possible explanation is the higher variability in tree age for the park trees compared to street trees. Older trees have been found to continue to increase in BAI, but at a different rate compared to younger trees under similar environmental conditions (Johnson & Abrams, 2009). Species were also observed to have differences in growth when compared within the same site type. Although in both parks and streets only one species differed statistically (*C. occidentalis* and *Tilia* spp.; respectively).

Basal area increment was higher on average for all species growing as street trees compared to park trees (Figure 1.2c). These results were inconsistent with the findings of previous studies (Quigley M. E., 2004; Fahey, et al., 2013; Sanders, et al., 2013). Sanders et al. (2013), found trees growing trees in unrestricted growing spaces were larger on average than those in more restricted spaces. The higher BAI for street trees growing in restricted areas may in part be attributable to runoff of stormwater. The impervious surfaces of sidewalks and buildings can increase the amount of surface runoff directed to a cities' stormwater drainage system (Han & Burian, 2005; Yao, Chen, & Wei, 2016)

from precipitation or city residents watering their lawns. While our data cannot be used to directly determine cause of the increased BAI and tree size for street trees compared to park trees, a possible causal mechanism is the increased runoff that flows over a boulevard planting, some of which likely infiltrates in the surrounding soils. Historically, the majority of rainfall events occurring in Minneapolis and St. Paul are less than 3cm (Fisk, 2017) much of which may be intercepted by a trees canopy and therefore would not infiltrate into the surrounding soils. In a park setting, light rain falling on pervious surfaces creates less runoff and a significant portion of the rain water that reaches the ground is absorbed and transpiration by turf grass (Peters, et al., 2001), resulting in drier soils in the rooting zone of park trees.

Influence of precipitation and temperature on growth

Trees growing in urban environments have rarely been considered for dendroclimatological studies and assumed to be insensitive to climatic conditions. With the exception of *Tilia* spp., our results demonstrate that trees in urban environments do respond to climatic features. Not surprisingly, the most influential climate factor was precipitation, which had a positive influence on growth of all species at all sites with July and August precipitation as significant months in all models. The influence of precipitation during the growing season is consistent with Helama, et al. (2009), where they found that variation in moisture during the growing season was the most prevalent factor influencing growth in oaks regardless of tree vigor. While no interactions between precipitation and temperature were found to be significant, this may be due to the short duration of the tree-ring series examined, as it is expected these two climate variables would interact to influence growth. This interaction may only be significant during

extreme events such as the severe drought in 1988, where precipitation was low and temperatures were high (National Centers for Environmental Information, 2016; Prism Climate Group, 2017). Mean BAI across all species and sites experienced a decrease during the 1988 drought period (data not presented). However, with the exception of the 1988 and 2007 drought periods, Minneapolis and St. Paul experienced normal levels of available soil moisture (National Centers for Environmental Information, 2016), which may have mitigated the influence of temperature.

May temperatures negatively influenced growth of *G. triacanthos* and positively influenced growth of *C. occidentalis*. The differences of temperature influence of growth may relate to phenology of the trees. *Gleditsia triacanthos* produces leaves in late-May or early June in Minnesota (Sullivan, Gleditsia triacanthos, 1994) and warmer temperatures may reduce available soil moisture needed for leaf development. Whereas, *Celtis occidentalis*, produces foliage and flowers in late-April (United States Department of Agriculture, 1990) when temperatures are typically cooler, potentially reducing the evaporative losses of soil moisture. Once the leaves are fully developed, *C. occidentalis* leaves would be better equipped to control temperature-related moisture stress through stomatal closure. Stangler et al. (2016) found that extreme heat during the leaf expansion period resulting in leaf drop had a negative impact on radial growth for *Acer saccharum* due to delay onset of growth, which was not experienced by *Betula alleghaniensis*. While *G. triacanthos* has not experienced annual May leaf-drop over the past 30-years the results presented in Stangler et al., (2016) indicate growth response to early season temperature varies by species, which could coincide with increased leaf predation by insects. For *A. platanoides t*he previous year's mean September temperature had a

positive influence on growth and could be attributed to its noted marginal performance in Minnesota (Nowak & Rowntree, 1990). A warmer fall may result in the reduced potential dieback due to early frosts. The minimal influence of temperature on growth may be the result of reduced sensitivity to temperature for urban trees. Wang et al., (2017) noted a reduction in temperature sensitivity for European trees between 1980 and 2013, although they noted that the long-term changes in temperature sensitivity remain uncertain.

The lack of climate factors significantly influencing growth of *Tilia* spp. was surprising considering all other species exhibited climate-growth responses. The minimal response of *Tilia* spp. growing in urban areas may indicate that *Tilia* spp. is more sensitive to anthropogenic factors (e.g. impervious surfaces, construction, pollution, etc.) than climate, which would reduce the detectable climate signal. This explanation would be consistent with Chen et al. (2011) where they found trees growing in rural areas were more sensitive to climate factors compared to their urban counterparts, which were more sensitive to anthropogenic factors.

Resistance and resilience to drought

Drought years identified using PDSI (National Centers for Environmental Information, 2016) included two drought episodes for the study period: 1987-1989 and 2007. All species, irrespective of site, experienced reduction in mean BAI during drier than average years which is consistent with existing research (Fahey, et al., 2013; Gillner, et al., 2014; Chen, et al., 2011; Cedro & Nowak, 2006). Differences in species response to drought has also been observed in both urban and forest trees (Fahey, Bialecki, & Carter, 2013; Gillner, Brauning, & Roloff, 2014; Dymond, et al., 2016).

Lower resistance and resilience values were expected for the 1988 drought as PDSI indicated a severely dry conditions compared to the moderately dry conditions of 2007. However, during the 1988 drought period all species and sites exhibited a high mean resistance and mean resilience to drought, whereas, no species or site exhibited strong resistance or resilience to the 2007 drought, despite the greater severity of 1988 drought compared to the 2007 drought (National Centers for Environmental Information, 2016). *Acer platanoides, C. occidentalis,* and *G. triacanthos* are all noted as being drought tolerant (Nowak & Rowntree, 1990; United States Department of Agriculture, 1990) and it was expected that these three species would demonstrate resistance and resilience over both drought periods, which was not the case. The influence of tree age and tree size is potentially a contributing factor in the differences in resistance and resilience over time (D'Amato, et al., 2013). When considering the difference in age between the two drought periods, our results are consistent with previous studies that demonstrated younger trees had higher resistance and resilience to drought compared to older trees (Lloret, et al., 2011; Martinez-Vilata, et al., 2012; D'Amato, et al., 2013). However, as trees have not been shown to have a physiologically senescence (Lee & Muzika, 2014), age only (holding size constant) is likely a poor explanation for our observed differences in drought response. Tree size and location may be more relevant for explanation of the temporal differences in drought response. Larger trees may recover more slowly in part due to an increase in water needs as trees grow in both height and leaf to sapwood area (Martinez-Vilata, et al., 2012; D'Amato, et al., 2013) and due to increased maintenance respiration in larger trees (Edwards & Hanson, 1996; Pallardy, 2008), thus reducing resistance to drought for larger (and generally older) trees. In should

be noted that reduced growth in older, larger trees post-drought may not necessarily be a negative trait and may be a strategy for surviving periods less conductive to growth (Johnson & Abrams, 2009).

During the 1988 drought street trees showed an overall higher resistance to drought compared to park trees (Figure 1.4); however, both populations exhibited strong drought resistance and the differences, while statistically significant, have little practical value. Park trees were older than street trees on average, yet their size in DBH, did not statistically differ (data not presented). Tree resistance and resilience to drought may partly be attributable to tree acclimation to site conditions, where trees in drier sites have greater resilience to drought events (Trouve, et al., 2016). Whitlow et al., (1992) observed lower water potentials in unirrigated compared to irrigated street trees, yet the street trees were able to maintain high water conductance. This may indicate that street trees have adapted to periods of greater temporary water deficits as a result of lower soil volumes compared to park trees, allowing for an increased water uptake during periods of moisture stress, resulting in higher resistance to drought events.

There were no statistical differences detected among species growing in either parks or along streets. *Celtis occidentalis* had the lowest resistance and resilience values for the 1988 drought period, but not statistically different from other species investigated likely due to the higher variability in growth exhibited for the younger trees. While resistance and resilience were high during the 1988 drought period, it should be noted that tree mortality over the study period was unknown. Therefore, only trees that survived the 1988 drought were available for investigation. It is possible that the resistance and

resilience metrics for the 1988 drought were exaggerated as trees with low resistance and resilience may have declined and died during or shortly after the drought.

Conclusion

Urban tree-rings can provide substantial insight in the influence of climate and anthropogenic factors for tree growth (Bartens, et al., 2012; Chen, et al., 2011). Our results clearly identified differences in growth between species and sites while demonstrating the influence that climatic factors and a temporal difference in resistance and resilience have on drought. Clark and Kjelgren (1990) identified the need for targeted research on the influence of drought on urban tree growth; this study has contributed to the growing body of literature on the influence of climate and drought on tree growth in the urban environment. As urban areas increase in population (United Nations, 2015) and changes in climate may be difficult to accurately predict, it will be critical for managers of urban forests to understand how climate factors impact the growth of urban trees. Greater understanding of the influence that seasonal precipitation and temperature have on tree growth will help guide selection of tree species that are adapted to future climate change while continuing to provide benefits to growing urban populations.

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CHAPTER 2
THE INFLUENCE OF SIDEWALK REPLACEMENT ON URBAN STREET TREE GROWTH

Introduction

Conflicts between sidewalks and trees are not new. Municipalities across the United States have spent thousands to millions of dollars annually to repair sidewalks, with documented damage resulting from interactions with tree roots (Wagar & Barker, 1983; Sydnor, et al., 2000; Olson, 2012). Tree root growth is multidirectional into the surrounding soil, dying if conditions are unfavorable and proliferating in areas conducive to growth (Harris, et al., 2004). The space between a concrete sidewalk and soil surface provides an ideal combination of moisture and oxygen with root densities often concentrated in sidewalk joints in close proximity (< 2 m) to tree trunks (D'Amato, et al., 2002) and concentrated in upper portions of base material under standard sidewalk installations (Grabosky, et al., 2001). Thus, there is a high probability of root damage anytime sidewalks adjacent to trees are repaired.

The probability and severity of damage (e.g., lifted or cracked panels) to sidewalks adjacent to trees increases with increasing tree diameter, the most severe damage chronically occurring with large-diameter trees in narrow planting spaces (Wagar & Barker, 1983; Francis, et al., 1996). Common factors observed in negative tree and sidewalk interactions include: large, fast growing trees ≥ 15 years old, and trees planted in strips < 3m wide that are adjacent to sidewalks with inadequate base material (Randup, et al., 2001).

Tree roots approximately 8 cm in diameter can lift a sidewalk panel 2.5 cm (Day R. W., 1991), which has public safety implications given an abrupt change of sidewalk continuity of 1.3 cm or greater can destabilize a typical pedestrian (Ayres & Kelkar, 2006). A United States 5th Circuit Court of Appeals ruling concluded that sidewalks are

an essential means of access for persons with disabilities and that sidewalks made or altered since 1992 must be constructed and maintained for accessibility (Ferleger, 2012). Sidewalk repair commonly involves removal and replacement of the damaged section and often results in the severance of tree roots to create a level, even base for the new sidewalk section. Impacts on trees from root severance include reductions in diameter and height growth and reduced twig elongation relative to trees with intact root systems (Harris, et al., 1998; Wajja-Musukwe, et al., 2007). Tree condition decreased, assessed using the Council of Tree and Landscape Appraisers system, and mortality increased for trees exposed to construction activities in Milwaukee, Wisconsin (Hauer, et al., 1994; Koeser, et al., 2013). Survival under stressed conditions (i.e., root severance) also differed by species with the greatest mortality observed in *Acer saccharum* and the least mortality observed in *Gleditsia triacanthos* var. *inermis* (Koeser, et al., 2013).

Construction damage may impact the ability of urban trees to tolerate periods of stress, such as drought. Root severance of established trees during suboptimal growing seasons (e.g., droughts) may redirect carbohydrate resources to storage and not root growth (Hamliton, 1988), further limiting water uptake. Even during wet years root severance has been correlated with reduced trunk diameter growth compared to non-impacted trees (Fini, et al., 2012). Individual species response to sidewalk replacement may vary, as *Tilia* x *europaea* was observed to have a greater tolerance to root severance over *Aesculus hippocastanum* in Vertemate con Minoprio, CO, Italy (Fini, et al., 2012).

Gillner, et al. (2014) found differences in growth response based on site conditions using tree-ring analysis with trees growing in highly-sealed environments (i.e., roots systems covered by non-porous material) exhibiting greater growth sensitivity to

stressed conditions compared to trees in less sealed environments. Chen et al. (2011) found complex relationships between anthropogenic environmental changes and the growth of trees. These and other studies have demonstrated that urban tree-ring analysis can serve as a valuable tool to quantify the influence of the built environment on urban street tree growth (Chen, et al., 2011; Fahey, et al., 2013; Gillner, et al., 2014; Vrecenak, et al., 1989).

Previous research on root severance and construction activities have investigated tree response via field studies or focused on tree survival and condition. In this study, our goal was to quantify in situ tree growth response over time on trees that survived root severance or root disturbance as a result of sidewalk construction in Minneapolis and Saint Paul, Minnesota, USA. Growth response was quantified as the annual basal area increment (BAI) and assessed in terms of resistance, resilience, and recovery to root disturbance from sidewalk construction activities. The specific questions we aimed to address were: What is the influence of sidewalk construction on post-construction tree growth and does this vary by species? How does the interaction of construction and seasonal drought affect post-construction growth? How do species resistance, resilience, and recovery to sidewalk construction differ? How does planting space width influence potential growth recovery post-construction?

Methods

The study consisted of publicly-managed street trees in the cities of Minneapolis (44.9778° N, 93.2650° W) and Saint Paul (44.9537° N, 93.0900° W), Minnesota, USA. Trees were selected from inventories provided by the Forestry Department of the Minneapolis Park and Recreation Board and the Forestry Department of the City of Saint

Paul. Using the combined inventory data, the species *Acer platanoides, Celtis occidentalis,* and *Gleditsia triacanthos* were selected as street trees common throughout both cities. The genus *Tilia* was also selected, but due to the inconsistencies with identification of hybrid and cultivated varieties, these individuals were only identified to genus. No attempt was made to identify any tree to cultivar or variety. Species were field-verified by researchers.

In conjunction with the tree inventory data, sidewalk construction data was obtained from the Public Works Departments of both cities. The construction data was limited to sidewalks replaced (full or partial slab removed and repaved) from 2002 through 2009. Construction year was field-verified via date stamp on the replaced sidewalk section. The sidewalk data and tree inventory data were joined together using a GIS to create a single dataset for selection of trees. To reduce the potential of confounding influence on growth due to the use of deicing salts (Cekstere, et al., 2010), only residential streets where winter deicing salts were not routinely applied were sampled. Trees growing under overhead utilities or other above ground obstructions were not sampled as pruning to accommodate above ground obstructions was thought to be a potential confounding influence on tree growth.

Each city consists of districts or wards (herein referred to as districts) used by the forestry departments to facilitate management of their urban forests. The districts served to stratify the cities to select a random sample of trees within each district. To select trees that were old enough to have been potentially affected by sidewalk construction, only trees with a diameter at breast height (DBH) of 25 cm or greater were included in the dataset. This diameter threshold was assumed to correspond to individuals that were at

least 20 years of age (Frelich, 1992) and thus old enough to have been present during the pre-defined construction period.

Street trees were divided into two populations. Trees directly adjacent to sidewalk construction (< 2m from construction), where the length of the replaced sidewalk panel or panels spanned the entire width of the trunk flare at ground line (WC). Proximity to sidewalk construction was not measured as all construction was a minimum of five-years in the past and the distance from construction at the time of construction was not known. However, present distance from construction was calculated as the trunk flare diameter subtracted from the planting space width and divided by two. The mean distance from sidewalks across all trees was 0.76 m with a maximum distance of 1.74 m.

The second population sampled were trees present during the construction period, but were further than 3m from sidewalk construction or where sidewalk construction did not occur within the canopy dripline, whichever distance was greater (NC). The distance criteria for WC and NC trees was based on Hauer, et al. (1994) where no discernable decline in tree condition attributable to construction was found for trees that were 2m from construction. Sample street trees were selected using a GIS by locating trees at addresses with reported sidewalk replacement. Sidewalk replacement rarely consisted of replacing all panels on a given street, which allowed for the sampling of WC and NC trees growing in similar conditions (e.g. planting space width, traffic patterns, localized weather events, etc.) at nearly the same location.

All data were collected from live trees between the months of June and August in 2014 and 2015. Trees adjacent to sidewalk construction (WC) were only sampled if the replaced sidewalk section measured a minimum of 1m in length and the replaced section

spanned the entire trunk flare diameter. Every sample tree was cored at 0.5m from the ground. The DBH was measured to the nearest 0.1cm using a diameter tape. Planting space width was measured as the linear distance between curb and sidewalk in meters to the nearest centimeter. One increment core per tree was obtained using a Haglöf 16 inch, 4.30mm core, increment borer. All increment core samples were taken on the sidewalk side of the tree. Cores were dried, mounted on wood mounts, and sanded with increasingly finer sandpaper up to 800 grit to produce a smooth, flat surface. Each core was viewed and dated by two researchers. Individual tree rings were measured using a Velmex Tree-Ring measurement system and the software J2X to the nearest 0.001 mm.

Data analysis

Annual tree growth was calculated as BAI from the tree-ring measurement using R (R Core Team, 2016) with the dplR package (Bunn, 2010). A comparison of BAI and raw ring-width increment showed BAI as a more robust measure of long-term growth trends for comparison of species and age classes (Johnson & Abrams, 2009). BAI from the most recent 24 years of growth (1991-2014) was used in the analysis. To account for the potential impact of available soil moisture on tree growth, the Palmer Drought Severity Index (PDSI) for the Minneapolis and Saint Paul region (National Centers for Environmental Information, 2016) was averaged over the growing season (April to October) (Fahey, et al., 2013) for each year of tree growth (Figure 2.1).

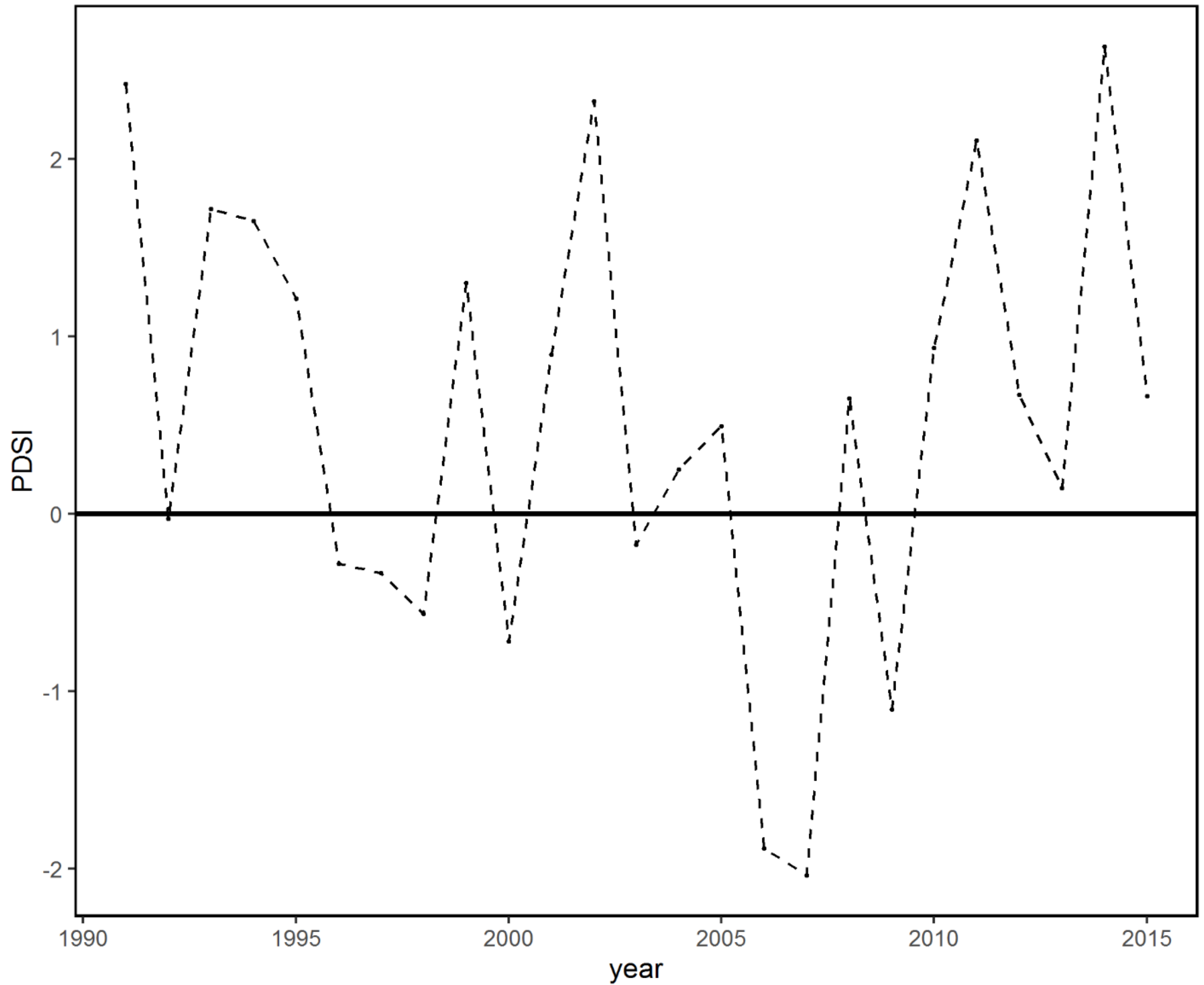

Figure 2.1 The average growing season PDSI over the measurement period (National Centers for Environmental Information, 2016).

Tree age, where pith was not visible in the core sample, was estimated using a series of concentric circles to approximate the number of non-visible rings using ring curvature (Applequist, 1958). Tree cores lacking a visible pith or with insufficient ring curvature (e.g., core too short, decay of internal rings) to provide a reasonable approximation of age were not used in the analysis (Table 1).

Table 2.1 Sample size and descriptive statistics of readable increment cores by construction activity (WC=with construction and NC=no construction) and species.

species	site	DBH (cm)	Age	n
A. platanoides	WC	44.2 (7.1)	38 (4)	34
	NC	44.1 (6.6)	38 (5)	37
C. occidentalis	WC	47.1 (8.5)	37 (5)	38
	NC	44.7 (7.6)	36 (7)	37
G. triacanthos	WC	47.9 (7.7)	37 (5)	36
	NC	47.5 (9.2)	37 (6)	38
Tilia spp.	WC	47.7 (10.8)	36 (7)	35
	NC	44.9 (10.8)	34 (8)	37

Note: Standard deviation shown in parenthesizes

The BAI growth trends showed an increase in growth over the study period and increased variability in growth after the 1990s (Figure 2.2). On average trees in the two study groups had a positive trend in BAI over time.

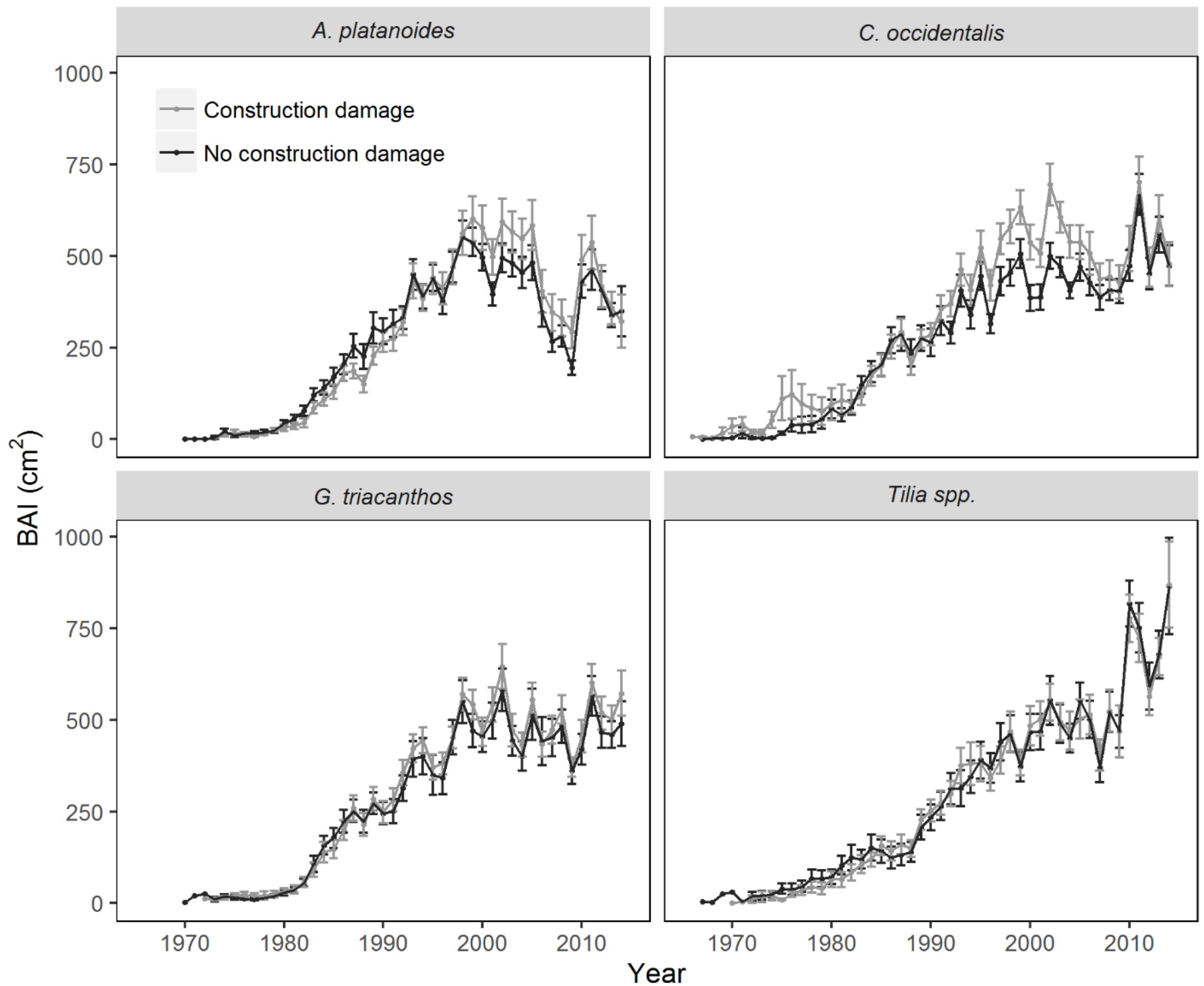

Figure 2.2 Growth trends in basal area increment (BAI) by species and disturbance group over the study period. Error bars indicate one standard error. (N=292)

Multilevel models were used to investigate the influence of sidewalk construction on BAI in trees while accounting for the temporal correlation and the heterogeneity of variation in the data (Uzoh & Oliver, 2008; Pokharel & Dech, 2012). Age (age), species (spp), construction status with 0 indicating pre-construction, and 1 indicating periods of post-construction (con), average PDSI over the growing season (PDSI), and site (WC and NC) were specified as fixed-effects. Year and tree were specified as random effects where the intercept was allowed to vary by year and slope was allowed to vary by tree.

To account for the non-random and correlated nature of tree ring data (Fritts, 1976), a first-order autoregressive covariance structure was added to the multilevel models (Pokharel & Dech, 2012). To address heteroscedasticity in residuals, a power function of DBH was applied to stabilize the residual variance (Pokharel & Dech, 2012). The best approximating model for explaining changes in BAI post sidewalk construction, based on Akiake Information Criteria (AIC), included the variables: age, con, PDSI, species, site, and an interaction term con:PDSI (model 1; Table 2.2). The multilevel models were fit using the lme function from the nlme package in R (R Core Team, 2016; Pinherio, et al., 2016).

Tree growth pre-and post-sidewalk construction were compared using three indices: resistance, resilience, and recovery from construction. Resistance to sidewalk construction is a species ability to maintain growth one year following construction, resilience to construction is a species ability to regain previous levels of growth after construction, and recovery from construction is the length of time required for a species to resume a pre-construction pattern of growth.

To quantify the growth response after sidewalk construction, species-level construction resistance (C_{rs}) and construction resilience (C_{rl}) were adapted from the drought-resistance and drought-resilience approaches used by D'Amato, et al. (2013) and Fahey, et al. (2013). Species-level construction resistance is defined as $C_{rs} = BAI_C / BAI_{pre}$, where BAI_C is the species-level average BAI one year after the construction and BAI_{pre} is the species-level average BAI for the five years preceding the sidewalk construction. Species-level construction resilience was defined as $C_{rl} = BAI_{post} / BAI_{pre}$, where BAI_{post} is the species-level average BAI for the five years after the construction

and BAI_{pre} is the species-level average BAI for the five years preceding the sidewalk construction. Values of C_{rs} and C_{rl} above 1 indicate resistance or resilience to sidewalk construction activities. One-way ANOVAs were conducted to determine the effect of species on the mean resistance index and mean resilience index.

The growth recovery index is defined as $G_{rec} = BAI_{yac} / BAI_{pre}$, where BAI_{yac} is the species-level average BAI for each year after construction and BAI_{pre} is the species-level average of the five-year period before the construction. Values of G_{rec} above 1 indicate growth recovery after sidewalk construction. A two-way ANCOVA was conducted to determine the effect of the interaction between species and years after construction on the mean recovery index. To assess the influence of the interaction between species and years after construction controlling for planting space width on mean recovery index a two-way factorial ANCOVA was conducted. Planting space width classes were defined as less than 1.25m, 1.25m to less than 2m, and greater than or equal to 2m. Tukey-Kramer adjustments were applied to the means comparisons. All statistical analyses were conducted in R (R Core Team, 2016).

Results

Table 2.2 presents the results of the mixed model analysis. Overall mean BAI growth varied by species and age across all investigated models. Tree age had positive influence on mean BAI (p=0.0000) (model 1; Table 2.2). *Celtis occidentalis* (p=0.0522), *G. triacanthos* (p=0.4103), *Tilia* spp. (p=0.0092), all had increased growth compared to *A. platanoides* (model 1; Table 2.2). Parameter estimate for PDSI (p=0.0000) was positive, increasing mean BAI in wetter than average years and decreasing mean BAI in

drier than average years. Trees in the WC population had a higher BAI on average, pre-sidewalk construction, than trees in the NC population (site:WC, p=0.065) (model 1; Table 2.2). The interaction between con:PDSI (p=0.0002) was also positive, indicating further reduction in mean BAI post-construction during drier than average years (model 1; Table 2.2).

Table 2.2 Model coefficients, statistical significance, and AIC for fixed effects. Average PDSI during the growing season is denoted by the variable PDSI, con are the years of growth after sidewalk construction, the interaction between construction and PDSI, and population-construction which represents the population of trees in the WC group.

coefficient	model 1	Model 2	model 3	model 4	model 5	model 6
intercept	8.934 (3.186) *	11.086 (2.972) ***	9.999 (2.961) ***	7.698 (3.172) *	10.890 (3.077) ***	11.073 (3.154) ***
age	1.056 (0.071) ***	1.050 (0.071) ***	1.091 (0.070) ***	1.096 (0.070) ***	0.975 (0.065) ***	1.026 (0.069) ***
con	-5.010 (1.252) ***	- 4.614 (1.234) ***	- 5.103 (1.228) ***	-5.515 (1.245) ***		-5.436 (1.253) ***
PDSI	1.983 (0.155) ***	1.978 (0.155) ***	2.250 (0.137) ***	2.250 (0.137) ***	2.249 (0.137) ***	
C. occidentalis	6.297 (3.230)	6.386 (3.241) *		6.364 (3.236)	6.314 (3.213)	6.208 (3.238)
G. triacanthos	2.684 (3.255)	2.713 (3.267)		2.679 (3.261)	2.619 (3.237)	2.691 (3.262)
Tilia spp.	8.578 (3.272) **	8.564 (3.283) **		8.701 (3.278) **	8.304 (3.252)	8.286 (3.280)
site:WC	4.322 (2.334)		4.339 (2.351)	4.668 (2.336) *	2.916 (2.286)	4.756 (2.337) *
con:PDSI	1.213 (0.330) ***	1.237 (0.330) ***				
AIC	57636	57641	57650	57647	57667	57905

Note: Standard errors are shown in parentheses. *** $p < 0.001$, ** $p < 0.01$, * $p < 0.05$

Post-construction mean BAI was reduced compared to pre-construction mean BAI in all species after accounting for age, PDSI, site, and the interaction between sidewalk construction and PDSI (Figure 2.3). The reduction in mean BAI for *Celtis* occidentalis was reduced, but the standard error range for post-sidewalk construction mean BAI overlapped with the pre-sidewalk construction mean BAI. Compared to the pre-construction mean BAI, post-construction showed increased variability (Figure 2.3).

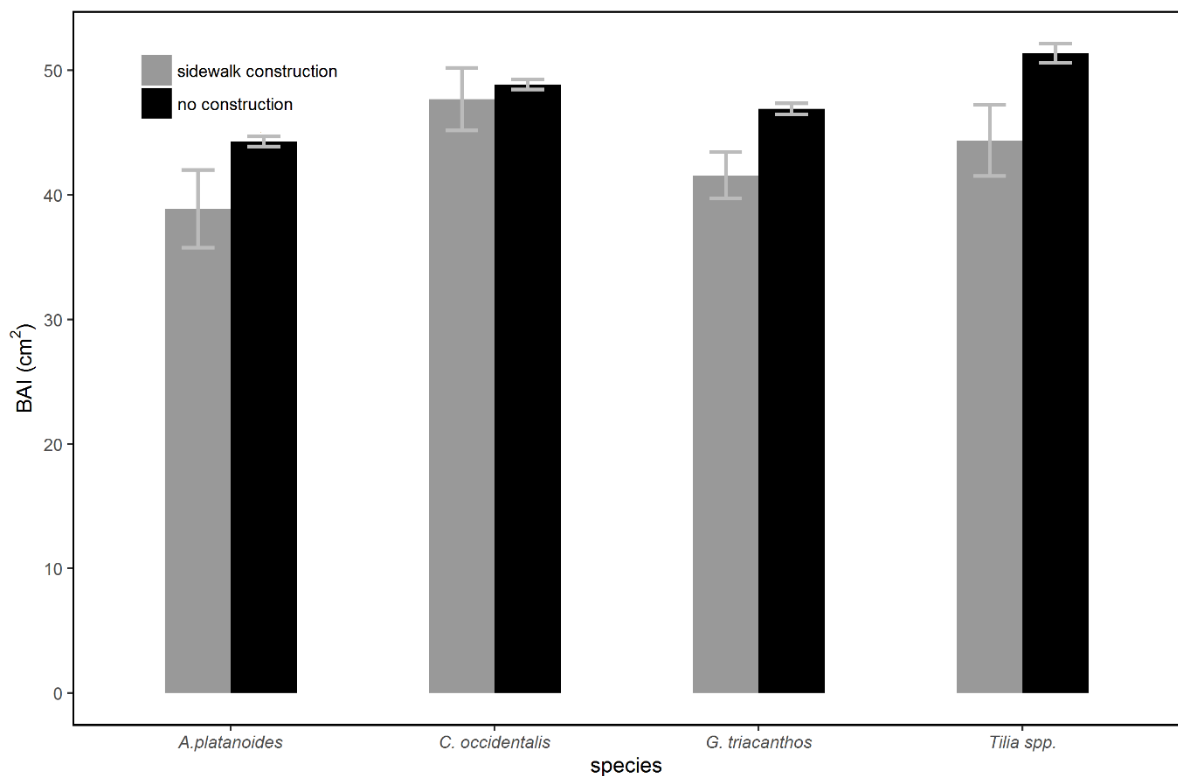

Figure 2.3 Comparison of modeled basal area increment (BAI) by species for pre-construction and post-construction. Error bars indicate one standard error. N = 292, trees. N = 6766, years of growth.

The effect of species on mean resistance index was non-significant, p = 0.4156 (Figure 2.4), whereas the effect of species on mean resilience index was significant, p=0.0116 (Figure 2.4). Mean resilience index comparisons between *Tilia* spp. – *A.*

platanoides (p=0.0087), and *Tilia* spp. – *G. triacanthos* (p=0.0590) were statistically and weakly statistically significant respectively, all other comparison between species were non-significant (p>0.05).

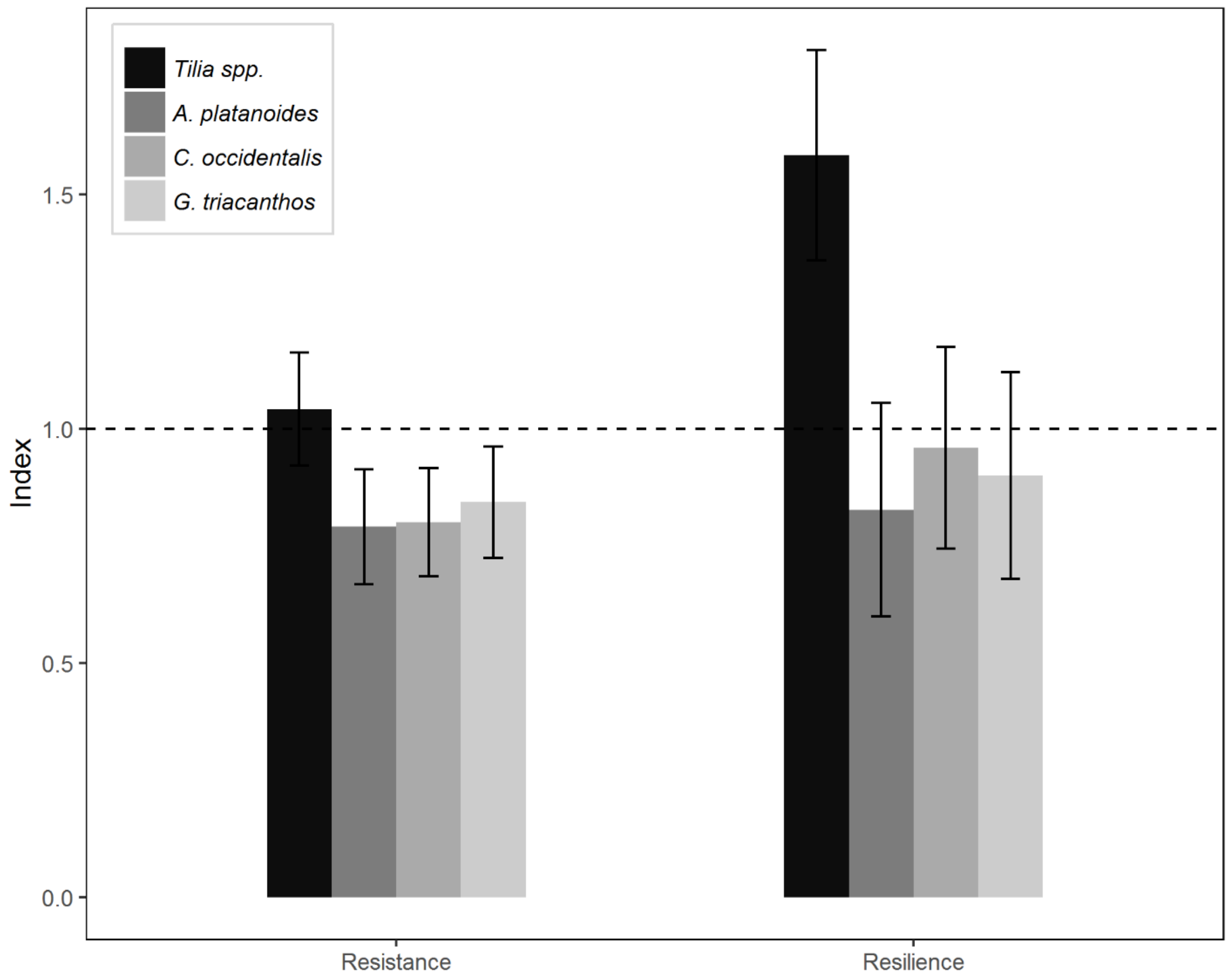

Figure 2.4 Indices of resistance and resilience to sidewalk construction by species. Bars indicate one standard error. Trees, N=142.

Species significantly differed in mean recovery index after controlling for the number of years after sidewalk construction, $p < 0.001$ (Figure 2.5). *Tilia* spp. had mean recovery index values greater than one by the second-year post-sidewalk construction and *C. occidentalis* had a recovery index greater than one by year three post-sidewalk

construction. Neither *G. triacanthos* nor *A. platanoides* had a recovery index greater than one within the five-year post-construction growth period used in the analysis (Figure 2.5).

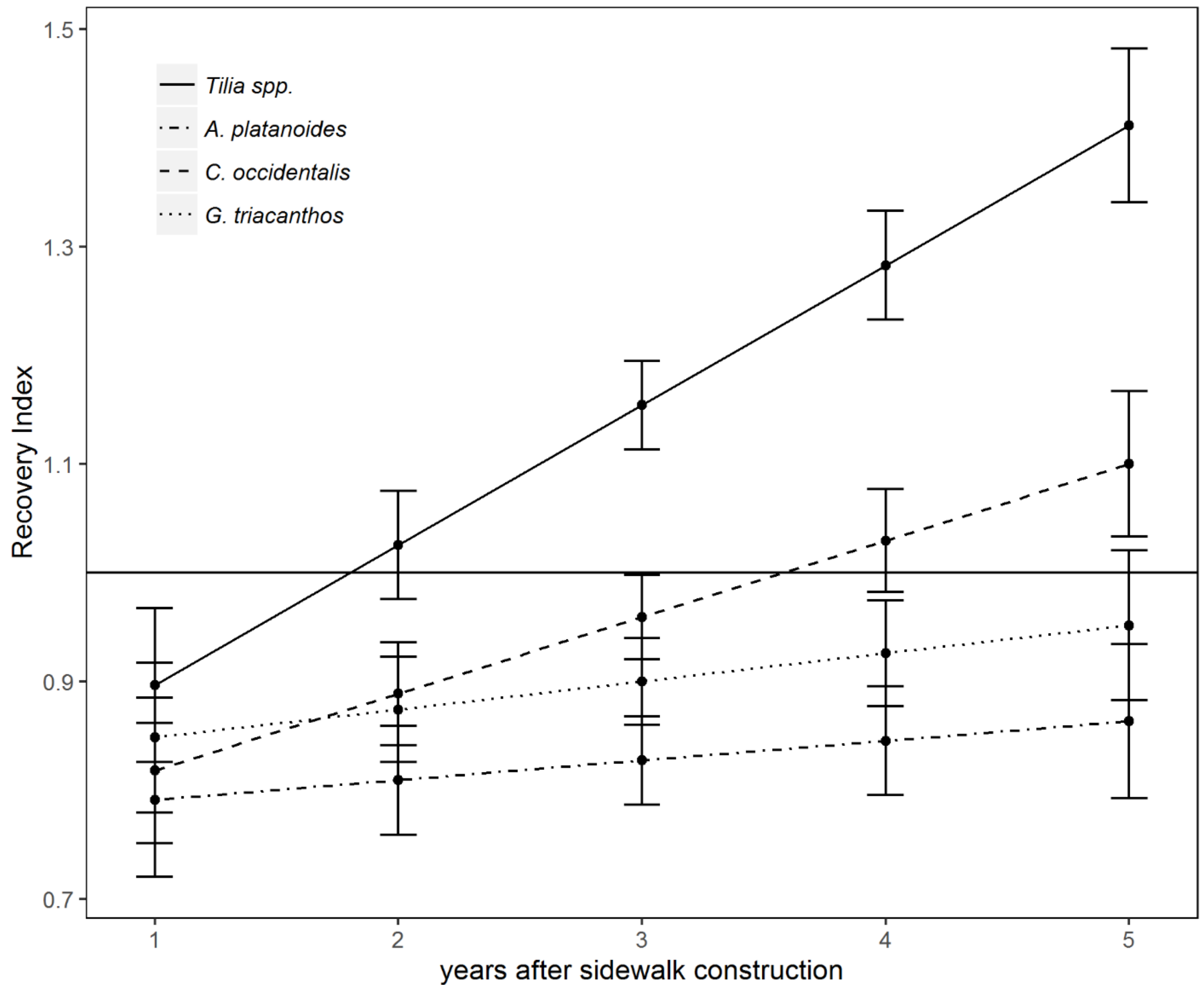

Figure 2.5 Mean growth response for 5 years after sidewalk construction by species; bars indicate one standard error. Tree, N=142. Year of growth, N=710.

Tree growth response by species post-sidewalk construction also varied across planting space widths, $p < 0.001$. Mean recovery index was significantly different between planting spaces ≥ 2m and <1.25m (p=0.0033), and between ≥1.25 to <2m and

<1.25m (p=0.0002). Narrower planting spaces (less than 1.25m) had a lower mean recovery index value for all species and all years post-construction (Figure 2.6).

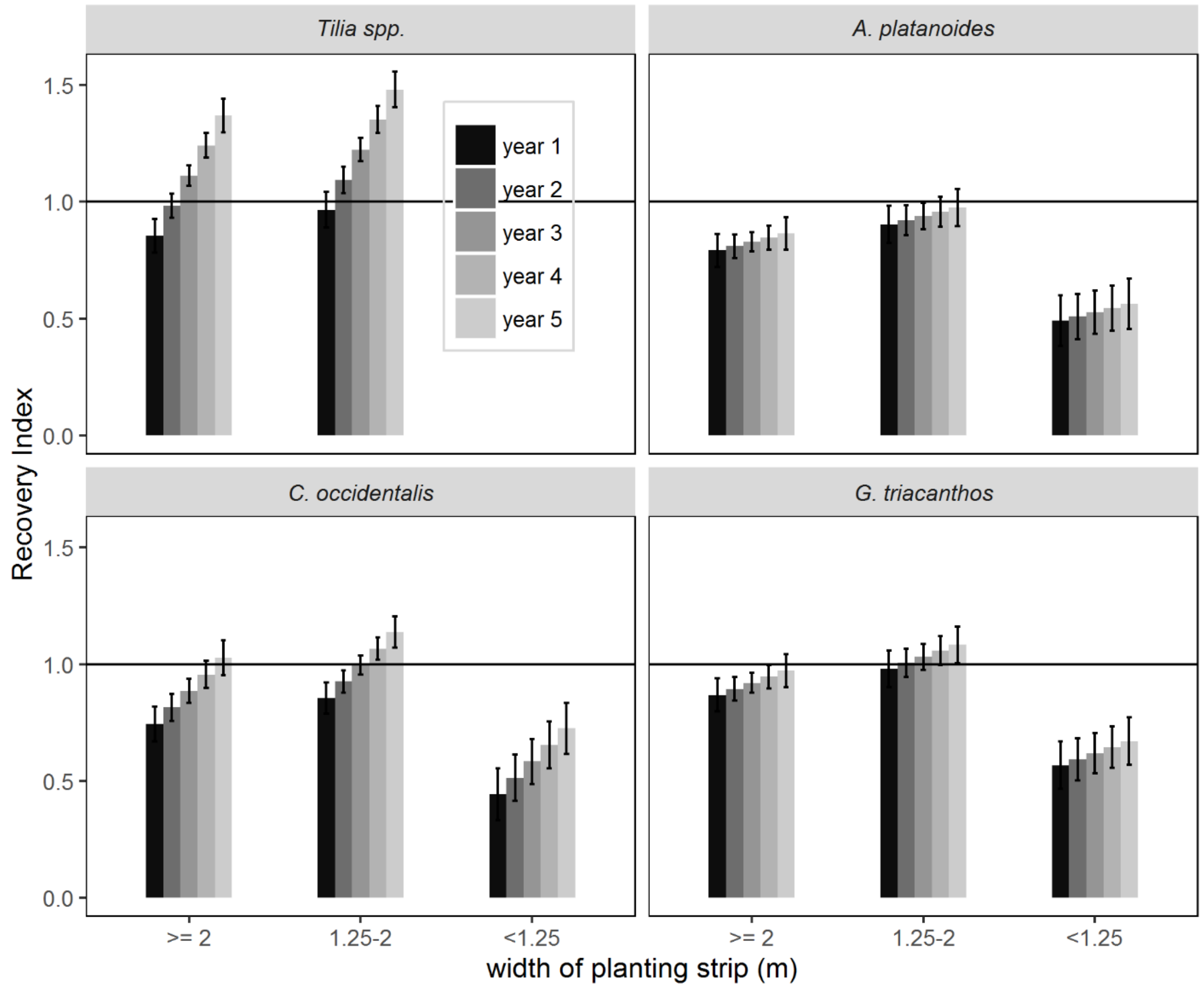

Figure 2.6 Recovery index by years after sidewalk construction, species, and width of planting space. Bars indicate one standard error. Trees, N=142. Years of growth, N=710

Discussion

Overall, for species and tree populations investigated, growth increased with age on average (Figure 2). Increased growth with age is consistent with the finding of Johnson and Abrams (2009), where they found radial growth slowed in older trees but

continued to increase in terms of BAI. Interestingly, trees in the WC population had an increased mean BAI compared to NC trees (model 1; Table2). This finding initially appeared counterintuitive; however, the increased growth rate of WC trees occurred prior to construction which indicates faster growing trees have an increased probability of disrupting sidewalks to a level that requires sidewalk repair, which is consistent with Randup, et al. (2001). Reductions in mean BAI for WC trees are consistent with the results of previous studies that have shown a negative influence of construction on trees (Hauer, et al., 1994; Koeser, et al., 2013).

Differences in species growth response based on urban site conditions and disturbance has been reported in other studies (Fahey, et al., 2013; Grabosky, et al., 2001; Iakovoglou, et al., 2001; Quigley, 2004). Quigley (2004) noted statistical differences in species growth response to site conditions and differences based on species successional status. Early successional species had statistically larger DBH than mid-to-late-successional species (Quigley, 2004). While successional status was not specifically investigated *C. occidentalis* and *Tilia* spp. are generally considered mid-to-late successional species (Sullivan, 1994b; Gucker, 2011) and had statistically higher mean BAI than *A. platanoides*, classified as a mid-to-late-successional species (Munger, 2003), for both WC and NC tree populations (model 1; Table 2). *Gleditsia triacanthos* (Sullivan, 1994a), classified as an early-successional species, did not differ statistically from *A. platanoides* in mean BAI (model 1; Table 2) or mean recovery index (Figure 5). The lower mean BAI and mean recovery index for *A. platanoides* supports Quigley's (2004) results and with reports of central Minnesota as a marginal range for *A. platanoides* (Nowak & Rowntree, 1990). The lower performance of *G. triacanthos* compared to other

species investigated is contrary to in the findings in Quigley (2004). However, Quigley (2004) did not specifically compare response to disturbance between successional statuses.

Drought years were assessed using PDSI, which takes on negative values during drought periods, and all trees experienced a decrease in BAI on average when PDSI was negative (model 1; Table 2). The reduced mean BAI during drier than average years is consistent with existing research (Fahey, et al., 2013; Gillner, et al., 2014; Chen, et al., 2011; Cedro & Nowak, 2006). Helama, et al. (2009), found that variation in moisture during the growing season was the most ubiquitous factor influencing growth in oaks regardless of tree vigor. Differences in species response to drought has also been observed in urban trees (Chen, et al., 2011; Gillner, et al., 2014). Gillner, et al. (2014) noted that *Acer* species in recent years have shown an increased response to precipitation. *Acer platanoides* was observed to have more restricted growth in drier years and greater growth in wetter years, compared to other species studied (Gillner, et al., 2014). Similar growth responses in terms of mean BAI was observed in this study.

The interaction between construction and PDSI had additional reduction in mean BAI for trees adjacent to construction during drought years (model 1; Table 2). The positive parameter estimates for the interaction between construction and PDSI increased mean BAI post-construction during wetter than average years. The slight increase in mean BAI post-construction in wetter years may be due to increased small and fine root production where roots were servered allowing additional uptake of water, as new roots and fine roots have increased water absorbing capability (Rook, 1971; Pallardy, 2008). The slight increase in mean BAI post-construction in wetter years does not fully

counteract the negative influence of construction, but it does reduce the negative influence on mean BAI.

Fahey, et al. (2013) found statistical differences in resistance to drought between *Gymnocladus dioicus* and *Liriodendro tulipifera*, but no differences between species resilience to drought. An opposite relationship was found for species response to construction: resistance to construction did not yield statistical differences between species and statistically significant species responses were found for resilience to construction (Figure 4). Only *Tilia* spp. demonstrated weak resistance (index values greater than 1) to sidewalk construction (Figure 4). *Acer platanoides, C. occidentalis,* and *G. triacanthos* are all noted as being drought tolerant (Nowak & Rowntree, 1990; United States Department of Agriculture, 1990) and it was expected that these three species would demonstrate as least some resistance to construction. The lack of resistance and resilience to construction suggests trees are undergoing more than water stress from root severance. A possible physiological explanation is that trees respond to root severance by redirecting carbohydrates created through photosynthesis to root production instead of increasing stem taper (Kozlowski, 1992; Pallardy, 2008). *Tilia* spp. had the highest mean resilience index at 1.58 and were the only trees with a mean resilience index greater than one. *Tilia amerciana* has been reported as a vigorous sprouter after clear-cutting (United States Department of Agriculture, 1990) and this adaptation may allow for faster post-distrubance recovery. Quigley (2004) investigated the growth of various tree species in rural and urban sites. All species investigate showed greater growth in rural compared to urban sites, except *Tilia* spp. which had a slightly higher mean growth in urban sites versus rural sites (Quigley, 2004). The first year of post-construction growth for all

species was below the recovery line (<1), which supports the lack of resistance to construction for the species studied (Figure 5). The reduction in radial growth for urban trees following tree-level disturbance is consistent with the response observed in natural forests (Merlin, et al., 2015).

Potts and Herrington (1990) reported adaptations to drought allowed *G. triacanthos* to accumulate more carbon that it uses if drought is not prolonged and *G. triacanthos* had long-term survival post-construction in Milwaukee, WI (Koeser, et al., 2013). Given *G. triacanthos'* apparent ability to persist in harsh or adverse conditions it was surprising that mean recovery index showed no recovery five-years post-construction (Figure 5). *Gleditsia triacanthos*' recovery trajectory was relatively flat. While it is difficult to explain the lack of recovery from the data, field observations of tree appearance seemed to indicate that *G. triacanthos* may not recover from construction by returning to pre-construction levels of growth, but rather may adopt a new "normal" growth level. That *Tilia* spp. demonstrated recovery to pre-construction growth levels 2-years post-construction is consistent with Koeser, et al., (2013) findings that *Tilia cordata* showed increased survival and growth post-construction.

Growth recovery post-sidewalk construction had a statistically significant relationship with planting space width (e.g., boulevard or tree-lawns). In fact, trees growing in planting space less than 1.25m wide did not show recovery to pre-construction growth after 5 years. The negative impact of reduced planting space on tree growth is consistent with previous research (Quigley, 2004; Day & Amateis, 2011; Dahlhausen, et al., 2016) and tree condition and survival post-construction are positively associated with increased planting space width (Hauer, et al., 1994; Koeser, et al., 2013).

Conclusion

This research indicated there is a measurable impact on tree growth that varies by species, climate conditions, planting environment, and tree age. Iakovoglou, et al. (2001) noted the mechanistic reasons behind the differences in species response to environmental factors can be difficult to determine. However, species' growth and survival response has been shown to differ based on age, environmental conditions, and disturbance (Hauer, et al., 1994; Johnson & Abrams, 2009; Lloret, et al., 2011; Fahey, et al., 2013; Koeser, et al., 2013).

Tree survival, resistance, resilience, and recovery post-construction are all important factors for maintaining the function of urban forests. However, from a management perspective tree stability should also be considered. A species resilience or recovery from construction activities in terms of growth needs to be weighed against a potential greater vulnerability to additional disturbance events (e.g. drought, wind storms) that may result from root severance. While our finding showed *Tilia* species had a higher resilience and shorter recovery period post-construction compared to other species, additional response factors should be considered. Increased growth, coupled with reduced tree stability after construction disturbance of root systems (Moore, 2014) can lead to higher rates of tree failure during storm events (Johnson G. , 2014). In a 2013 wind and rain storm in Minneapolis, Minnesota *Tilia* species adjacent to sidewalk sections replaced within 5 years where found to be 2.24 times more like to fail (Johnson G. , 2014). Given the reduced stability of trees with root damage (Smiley, 2008; Fini, et al., 2012; Moore,

2014), even species that demonstrate a significant ability to recover from construction may be more liabilities than assets in urban environments.

Increased width of tree planting spaces along streets combined with species selection that considers the mature size of a tree are strategies that can help to reduce the need for sidewalk repair adjacent to trees (Wagar & Barker, 1983; Costello & Jones, 2003; North, et al., 2015) and subsequent reduction in stability, growth, and survival of trees as a result of construction activities. To stabilize and potentially increase future urban canopy cover, the interaction between site design and species adaptations should be considered to reduce effects of anthropogenic disturbance and provide adequate growing space required for large, healthy, and long-lived trees.

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CHAPTER 3
SITE CHARACTERISTICS AND TREE ARCHITECTURE AS INDICATORS OF URBAN TREE PERFORMANCE

Introduction

The field of urban and community forestry has seen a paradigm shift from objectives primarily concerned with aesthetics to objectives encompassing economic, environmental, and societal concerns (McPherson E. G., 2006). Environmental and economic concerns include, but are not limited to: energy savings, air quality improvement, CO_2 reduction, storm-water runoff attenuation, aesthetic valuation, and increased property values (Dwyer, et al., 2000; Brack, 2002; Cappiella, et al., 2005; McPherson E. G., 2003; Sanders, et al., 2010). Collectively, these concerns are referred to as the benefits of trees. Many benefits provided by trees are related to the size and condition of the tree crown. Not surprisingly, larger healthy trees have been shown to provide the greatest societal and economic benefits (McPherson E. G., 2003; Morales, 1980; Dimke, Sydnor, & Gardner, 2013). The continued development of software designed to calculate benefits and assign value to urban trees (e.g. i-Tree™) highlights the ongoing need for research capable of quantifying the capacity of urban sites for supporting healthy trees that confer the highest level of benefits.

While larger canopies appear to be the driving factor in quantity and quality of benefits trees provide (McPherson E. G., 2003), age and sustainable growth of urban forests should not be discounted. A tree's ability to maintain wood tissue and continue to grow is impacted by the ratio of photosynthetic to non-photosynthetic area. Maintenance respiration to support branches and stems has been reported to deplete 25% to 60% of daily photosynthate production (Edwards & Hanson, 1996; Pallardy, 2008). Trees with a lower crown surface area to stem surface consume photosynthates at a greater rate. For example, Drobyshev, et al. (2007) found a positive relationship between crown condition

and change in stem diameter growth, with poor crown conditions having a lower stem increment. Voelker, et al. (2008), created a tree vigor index (TVI) to assess the vigor and growth of red oaks. The TVI is the ratio of canopy surface area to trunk surface area and was used to assess tree vigor and predictor of future growth, as tree size alone was not a good indicator of future tree growth (Voelker, et al., 2008; Lee, et al., 2014). Sustainable urban forests must maintain their ability to provide net services (i.e. benefits) over time (Clark, et al., 1997). Development of a tree performance metric that incorporates tree size, tree vigor, and a growth rate component could help to identify sites capable of supporting not only large trees, but also sustainable tree growth over time.

Approaches for assessing the capacity of a given site to support tree growth have long been a central element of traditional forestry applications and have generally quantified site productivity or site index based on tree height and volume predictions to produce wood (Skovsgaard & Vanclay, 2008). Urban tree performance (e.g. potential to provide benefits) is more difficult to quantify than single metrics like DBH, tree height, growth, or crown size. Assessments of urban environments need to incorporate not only the ability of site to support tree canopies, but also sustained tree growth over time.

When assessing urban tree performance, the first challenge is to identify a tree growth metric that adequately assesses which sites are supporting trees capable of meeting the desired objectives. Because urban tree benefits represent multiple objectives, a single metric, such as canopy size, may not adequately address objectives dealing with longevity or biomass. A second challenge in assessing urban tree growth is to identify site characteristics that can be collected and analyzed in conjunction with the selected tree growth metric. Previous studies have shown difficulty in establishing consistent

relationships between urban site characteristics and tree growth, which have been attributed to the variability between sites and difference in species response to site characteristics (Hodge & Boswell, 1993; Sherman, et al., 2016).

The volume, bulk density, compaction, and pervious surface of soil have been used to explain tree growth in urban environments (Hodge & Boswell, 1993; Day & Amateis, 2011; Sanders & Grabosky, 2014; Dahlhausen, et al., 2016;). Surface area of available soil surrounding trees has produced the most consistent results, whereas bulk density, compaction, and volume have been less consistent in explaining tree performance (Hodge & Boswell, 1993; Day & Amateis, 2011; Iakovoglou, et al., 2001; Sherman, et al., 2016). Some studies have addressed the variability of site characteristics by conducting research in limited sites with even-aged trees (Day & Amateis, 2011; Sanders & Grabosky, 2014). Past research has identified the need for long-term data to incorporate the variability between sites and trees (Hodge & Boswell, 1993; Leibowitz, 2012). Using tree-rings allows a retrospective look at both growth and growth rates through time.

Lee, et al. (2014) established that TVI was a useful metric of tree growth that incorporated tree architecture and concepts of sustained growth, for scarlet and black oaks in the Ozark Highlands of Missouri, USA. However, there have been no applications of this index to urban environments. The objectives of this study were to: 1) investigate how key tree attributes (including size, growth rate, and age) vary by planting location and tree species, 2) adapt TVI to urban environments for assessing tree performance, and 3) investigate the influence of site characteristics and tree defects on tree growth and performance.

Methods

Research was conducted on municipally-managed parks and street trees in the cities of Minneapolis (44.9778° N, 93.2650° W) and Saint Paul (44.9537° N, 93.0900° W), Minnesota, USA. Live trees were selected from inventories provided by the Forestry Department of the Minneapolis Park and Recreation Board and the Forestry Department of the City of Saint Paul. Using the combined inventory data of both cities, the species *Acer platanoides, Celtis occidentalis,* and *Gleditsia triacanthos* were selected as municipally-managed trees common throughout both cities. Due to inconsistencies in identification of hybrid and cultivated varieties the genus *Tilia* was also selected, with no attempt to identify individual trees to species or cultivar.

Park trees (PT) included in the study were growing in managed municipal parks where there was evidence of tree and landscape maintenance (i.e. pruning and lawn mowing) and impervious surfaces were a minimum of 1m from the canopy dripline. Park trees were considered to have unrestricted growing space beneath the canopy. Street trees (ST) included in the study were growing in municipally-managed boulevard planting strips between sidewalk and curb on residential non-arterial streets. Street trees were considered to have restricted growing space beneath the canopy. Species were field-verified by researchers. All sampled trees had a minimum diameter of 25cm DBH which was assumed to indicate trees were at least 20 years of age (Frelich, 1992) and established in their environments (Sherman, et al., 2016). Both street trees and park trees had similar distances to neighboring trees (~9m to 20m). There was no upper limit imposed on DBH. Municipal forestry management districts were used to stratify each city whereupon trees meeting criteria for the study were randomly selected from each district.

Tree variables

Trunk flare diameter (TFD; at ground line), diameter at coring height (DACH; 0.5m from ground line), and DBH (1.3m from ground line) were measured to the nearest 0.1cm using a diameter tape. Four crown radii were measured from the dripline to the trunk. All crown radii measurements were made to the nearest centimeter using a Bosch DLR130K laser distance measurer taken in the cardinal directions. Canopy projection area (CPA) was calculated as the square of the averaged crown radii multiplied by pi. Tree height and crown height were measured as a percent of the distance from the observer to the tree using a Sunto clinometer and a Bosch DLR130K laser distance measurer. Trunk height from ground level to the base of the crown was calculated by subtracting the crown height from the total tree height.

Every sample tree was cored at 0.5m from the ground. One increment core per tree was obtained using a Haglöf 16 inch, 4.30mm core, increment borer. Cores were dried, mounted on wood mounts, and sanded with increasingly finer sandpaper up to 800 grit to produce a smooth, flat surface. Each core was viewed and dated by two researchers. To estimate tree age from core samples where pith was not visible, a series of concentric circles were used to approximate the number of non-visible rings (Applequist, 1958). Tree cores lacking a visible pith and with insufficient ring curvature (e.g. decay of internal rings) to provide a reasonable approximation of age were not used in the analysis. Individual tree rings were measured using a Velmex Tree-Ring measurement system and the software J2X to the nearest 0.001 mm. All data were collected between the months of June and August in 2014 and 2015.

Tree damage (DAMAGE) was visually assessed and recorded as present or absent when decay, cankers, cracks, or ribs on the trunk below the canopy measured 10% or greater of trunk circumference or trunk height. The presence of stem girdling roots (SGR) were determined via a visual assessment and recorded as either present or absent. Stem girdling roots are roots contacting the trunk causing compression or deformation in the trunk issue typically at or near ground line (Johnson & Hauer, 2000), and are thought to reduce tree growth through restriction of the flow of water and nutrients (Hulder & Beale, 1981; Johnson & Hauer, 2000; Wells, et al., 2006).

Site variables

Soil compaction (COMP) was measured using an Eijkelkamp hand penetrometer to a depth of 25cm and recorded in MPa. Pervious surface area (PSA) under the canopy was measured as the contiguously open surface area under the canopy in meters to the nearest centimeter. Canopy overlap (COLAP) was recorded when the crown of the observed tree overlapped an adjacent tree by at least 0.5m.

Data analysis

Basal area increment (BAI) was calculated from tree-ring measurements using R (R Core Team, 2016) with the dplR package (Bunn A. G., 2010). A comparison of raw ring-width increment and BAI showed BAI as a more robust measure of long-term growth trends for comparison of species and age classes (Johnson & Abrams, 2009). A BAI ratio (BRATIO) was derived from the average BAI over the last 10 years of growth divided by average BAI over the life of the tree. A BRATIO greater than 1 indicated an

increase in growth over the last ten years and decreased growth when BRATIO was less than 1.

A tree performance index (TPI) was derived using field measurements of trunk and crown to create a modified form of the TVI (Voelker, et al., 2008; Lee, et al., 2014) multiplied by BRATIO.

$$TPI=\sqrt{TVI*BRATIO} \tag{3.1}$$

Where,

$$TVI = \frac{CSA}{SSA} \tag{3.2}$$

The crown surface area (CSA) was calculated as either the surface area of a cone or the surface area of the sphere based on the approximate crown form of a species. Determination of crown form was based on Wandell (1989) and field observations. *Acer platanoides* and *C. occidentalis* mostly closely resembled spherical form, whereas, *G. triacanthos* and *Tilia* spp. forms where viewed as inverse conical and conical, respectively. Stem surface area (SSA) was calculate as the lateral surface area of a tapered cylinder using the trunk flare diameter (TFD) as the base of the cylinder and the top diameter of the cylinder was the estimated trunk diameter at the base of the crown. The diameter at the base of the crown of each tree was estimated using linear regression to find the mean decrease in diameter over the distance between DACH and DBH by tree species.

Two-way factorial ANOVAs were conducted to examine the influence of site, species, and their interaction on mean age, DBH, height, CPA, BRATIO, and TPI. A Tukey's honest significance test, post-hoc analysis was conducted in cases in which a main factor was significant. Significance level was set at 0.05.

Ordinary least squares regression models were employed to assess differences in tree growth and performance based on TPI, DBH, CPA, and BRATIO and their relationships to site and tree characteristics. Site characteristics included: PSA, COMP, and COLAP. Tree characteristics included: presence of stem girdling roots (SGR), presence of tree damage (DAMAGE), site (PT and ST), soil compaction (COMP), canopy overlap (COLAP), and tree age class (17-25, 26-50, 51-75, and > 75 years). Each model for a given response variable was selected based on the corrected Akaike Information Criterion (AIC_c) using forward and backward selection techniques based on all variables. A key to variable abbreviations is provided in Table 3.1. The final models for each response variable are presented in Table 3.3. The models were fit using the lm function in R (R Core Team, 2016). All statistical analyses were conducted in R (R Core Team, 2016).

Table 3.1 Reference guides to variable abbreviations used in analysis.

abbreviation	full name	definition
BAI	basal area increment	annual growth increment
BRATIO	basal area increment ratio	ratio of mean the of last 10 years of growth to the mean of total tree growth
CPA	canopy projection area	area of the canopy based on crown radii
COLAP	canopy overlap	binary; 1 = canopy overlap, 0 = no canopy overlap
COMP	compaction	soil compaction was measured by penetrometer
DAMAGE	tree damage	binary; 1 = tree damage, 0 = no tree damage
DBH	diameter at breast height	trunk diameter at 1.3m
PSA	pervious surface area	pervious surface area under the canopy
PT	park tree	park tree
SGR	stem girdling root	binary; 1 = stem girdling root, 0 = no stem girdling root
ST	street tree	street tree
TPI	tree performance index	index based on ratio of canopy to stem surface area multiplied by growth rate ratio

Results

In total 320 trees were sampled with species equally represented in the two site types (PT or ST). Of the 320 trees sampled, 292 trees were included in the final analysis. Table 3.2 provides details on tree age, height, DBH, BRATIO, and TPI based on species, site, and species by site.

Table 3.2 Descriptive statistics by species. Age in years, DBH in cm, height in meters, CPA in meters, BRATIO, and TPI. The standard deviation is presented in parenthesis.

Species	n	age	DBH	height	CPA	BRATIO	TPI
ACPL	74	38 (8)	43.4 (8.6)	14 (2)	84.0 (29.7)	1.23 (0.42)	1.07 (0.22)
CEOC	72	49 (25)	47.1 (9.5)	17 (3)	109.4 (47.8)	1.38 (0.41)	1.08 (0.27)
GLTR	75	39 (13)	43.0 (11.0)	16 (4)	124.0 (51.3)	1.47 (0.34)	1.19 (0.21)
TI	71	35 (11)	44.9 (10.6)	15 (3)	71.7 (30.1)	1.54 (0.25)	0.96 (0.22)

Note: ACPL = *A. platanoides,* CEOC = *C. occidentalis*, GLTR = *G. triacanthos,* TI = *Tilia* spp.

The effect of species and site interactions on mean age were significant, p = 0.0000. Park trees were significantly older than street trees (p = 0.0000). Mean age for *C. occidentalis* differed significantly from all other species (p = 0.0000; Figure 3.1). *Celtis occidentalis* in parks had significantly higher mean age (64 years) than other species and had the greatest variability (Figure 3.1). There was no statistical difference in tree age among or within sites for other species (Figure 3.1).

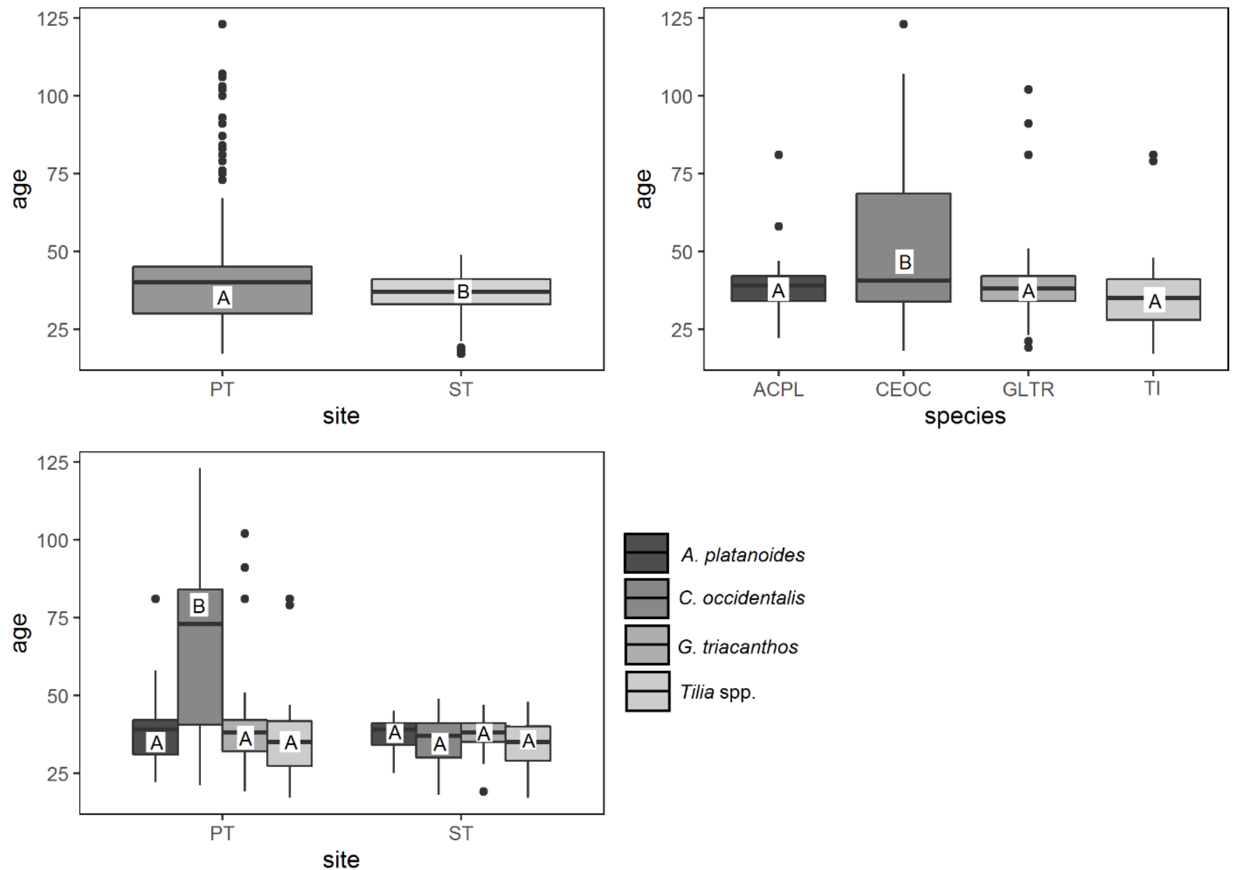

Figure 3.1. Age by site, species, and site:species interactions. Different letters indicate statistically different means. *Acer platanoides* (ACPL), *C. occidentalis* (CEOC), *G. triacanthos* (GLTR), and *Tilia* spp. (TI). Park trees (PT) and street trees (ST). (N = 292)

Comparisons of mean DBH showed no significant differences between species, sites, or the interactions among species and sites ($p > 0.05$; Figure 3.2). *Gleditsia triacanthos* growing in parks did differ statistically from both *A. platanoides* and *C. occidentalis* ($p < 0.05$; Figure 3.2).

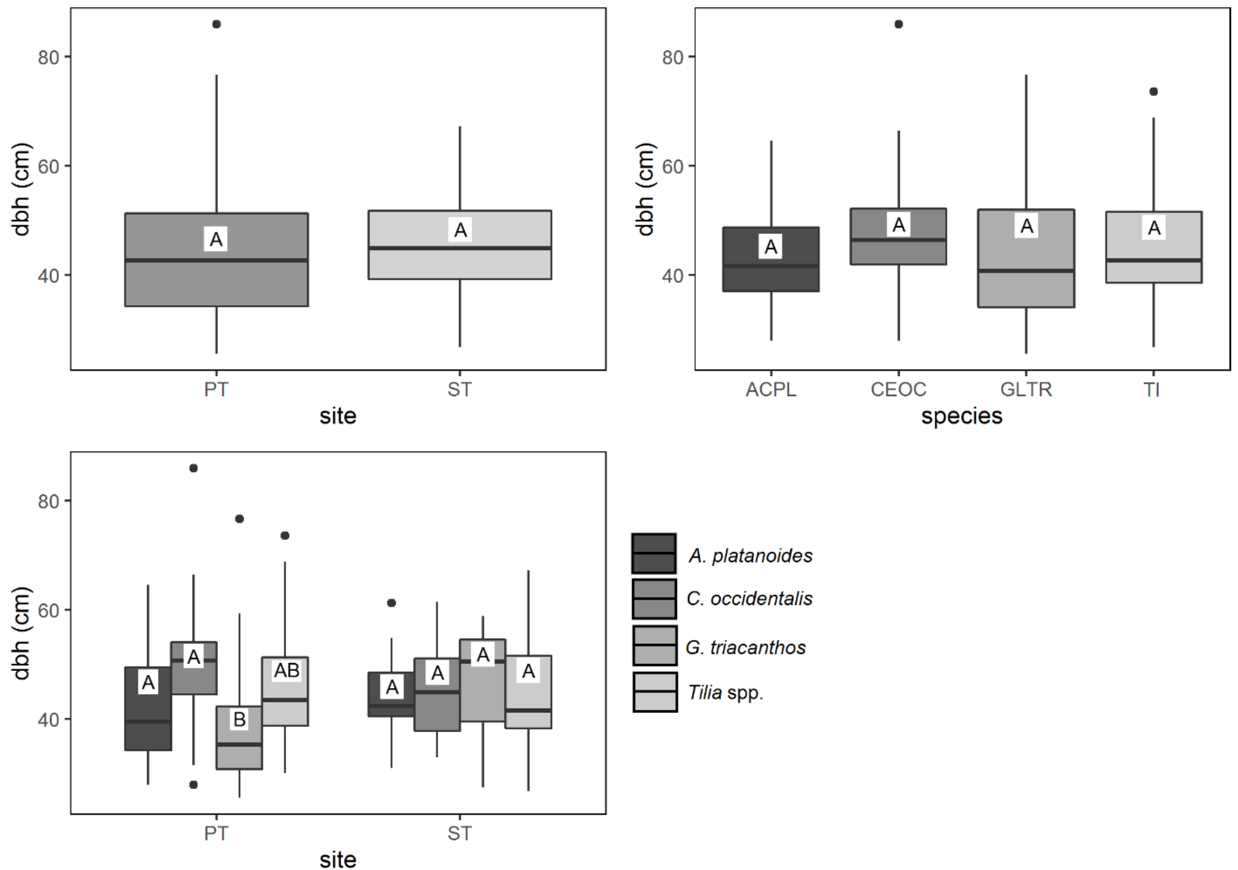

Figure 3.2. DBH by site, species, and site:species interactions. Different letters indicate statistically different means. *Acer platanoides* (ACPL), *C. occidentalis* (CEOC), *G. triacanthos* (GLTR), and *Tilia* spp. (TI). Park trees (PT) and street trees (ST). (N = 292)

Mean CPA was significantly higher for ST than PT ($p = 0.0142$). Canopy projection area was statistically lower for *A. platanoides* than for either *C. occidentalis* ($p = 0.0018$) or *G. triacanthos* ($p < 0.0001$). *Tilia* spp. showed a statistically lower CPA than *A. platanoides* ($p = 0.0262$), *C. occidentalis* ($p < 0.0001$), and *G. triacanthos* ($p < 0.0001$). Comparisons of CPA among species, between sites, and within a site among species can be seen in Figure 3.3.

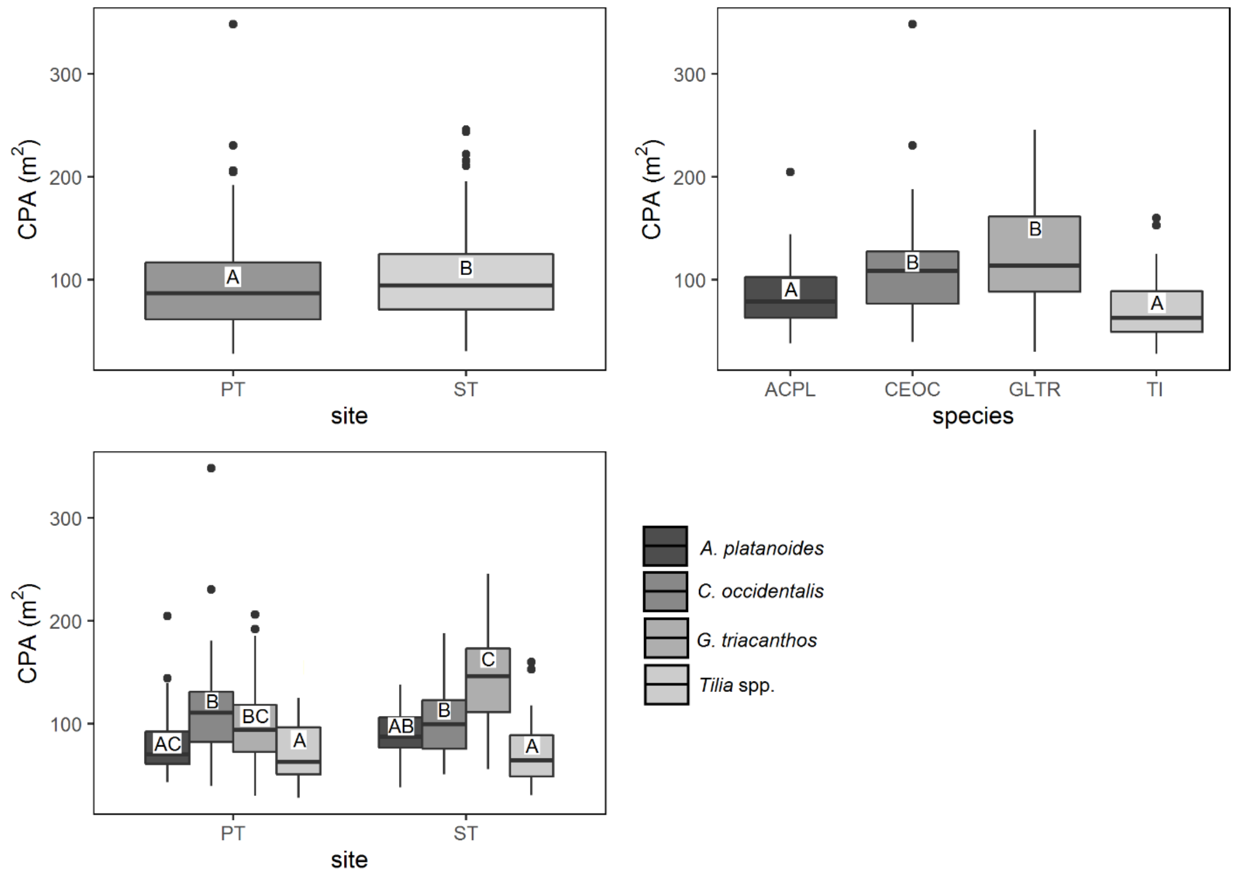

Figure 3.3. CPA by site, species, and site:species interactions. Different letters indicate statistically different means. *Acer platanoides* (ACPL), *C. occidentalis* (CEOC), *G. triacanthos* (GLTR), and *Tilia* spp. (TI). Park trees (PT) and street trees (ST). (N = 292)

Mean tree height for *A. platanoides* was significantly less than both *C. occidentalis* ($p < 0.0001$) and *G. triacanthos* ($p = 0.0092$) and mean tree height of *Tilia* spp. was significantly less than *C. occidentalis* ($p = 0.0082$). All other species comparisons were non-significant ($p > 0.05$). The mean tree height of park trees was significantly larger than street trees ($p = 0.0002$, Table 3.2). For *G. triacanthos* park trees were statistically taller on average than street trees ($p < 0.0001$). Comparisons of mean height for street trees showed *G. triacanthos* was taller on average than *A. platanoides* ($p = 0.0008$). Comparisons of mean height for park trees showed *C. occidentalis* was taller

on average than *G. triacanthos* (p = 0.0001), *A. platanoides* (p < 0.0001), *Tilia* spp. (p = 0.0150). Differences in mean tree height among species, between sites, and within a site among species can be seen in Figure 3.4.

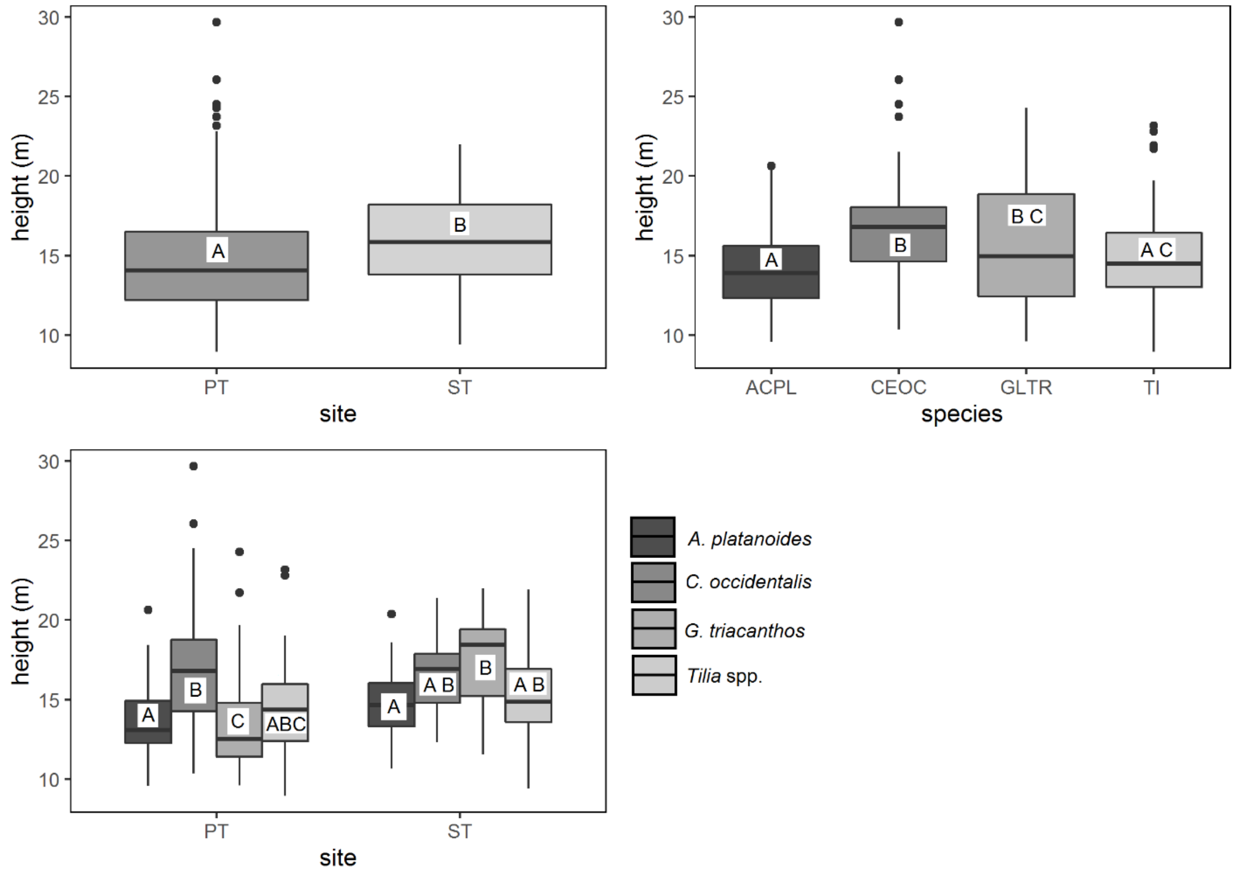

Figure 3.4. Height by site, species, and site:species interactions. Different letters indicate statistically different means. *Acer platanoides* (ACPL), *C. occidentalis* (CEOC), *G. triacanthos* (GLTR), and *Tilia* spp. (TI). Park trees (PT) and street trees (ST). (N = 292)

Mean BRATIO differed significantly between GLTR and ACPL (p < 0.0001), TI and ACPL (p = 0.0000), TI and CEOC (p < 0.0001). All other species comparisons were non-significant. The mean BRATIO was significantly different between PT and ST trees, and between PT and ST trees (p < 0.0001, p < 0.0001). Differences in mean BRATIO among species, between sites, and within a site among species can be seen in Figure 3.5.

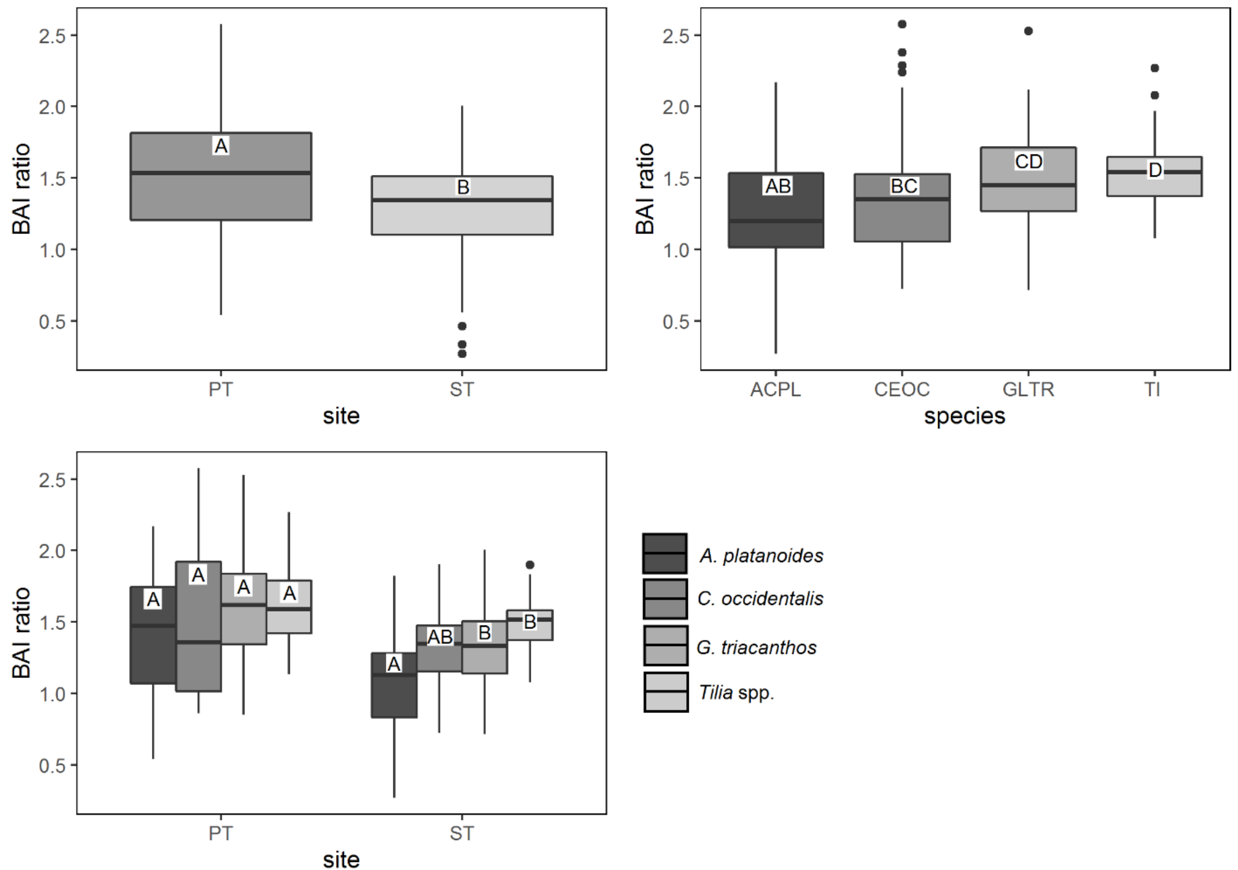

Figure 3.5. Basal area increment (BAI) ratio by site, species, and site:species interactions. Different letters indicate statistically different means. *Acer platanoides* (ACPL), *C. occidentalis* (CEOC), *G. triacanthos* (GLTR), and *Tilia* spp. (TI). Park trees (PT) and street trees (ST). (N = 292)

Mean TPI was significantly higher for park trees than street trees accounting for species (p = 0.0000). Tree species also had a significant effect on mean TPI accounting for the effect of site (p = 0.0000) and the species-site interaction was non-significant (p = 0.5840). Differences in mean TPI among species, between sites, and within a site among species can be seen in Figure 3.6.

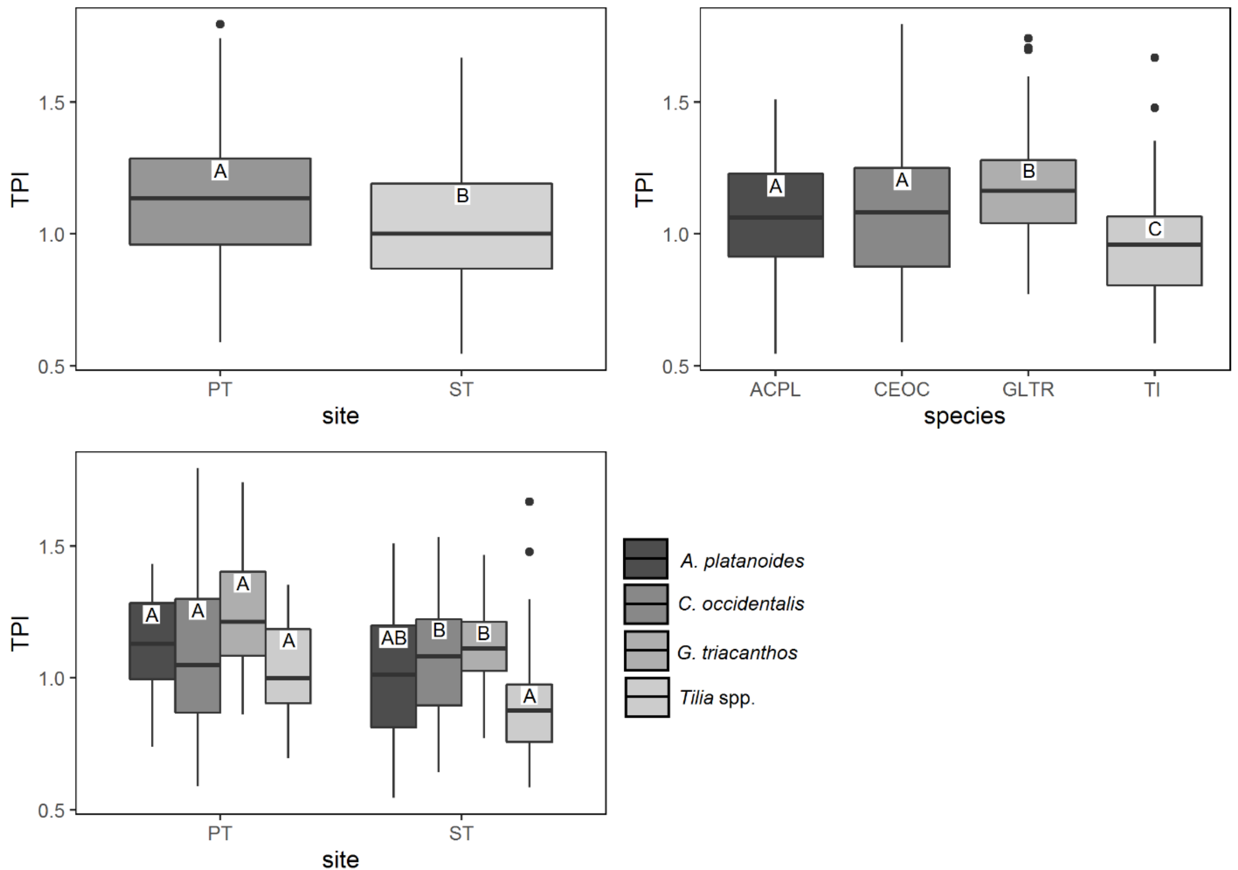

Figure 3.6. Tree performance index (TPI) by site, species, and site:species interactions. Different letters indicate statistically different means. *Acer platanoides* (ACPL), *C. occidentalis* (CEOC), *G. triacanthos* (GLTR), and *Tilia* spp. (TI). Park trees (PT) and street trees (ST). (N = 292)

The available pervious surface area under the canopy (PSA) and species were statistically significant factors explaining variation observed in TPI, DBH, CPA, BRATIO across all models (Table 3.3). In all models except BRATIO, pervious surface area under the canopy had a positive influence on the variables of interest. Although pervious surface area was negative for mean BRATIO, the overall influence was minimal. In terms of R-squared values, the CPA model explained the most amount of observed variation, approximately 67%, and the BRATIO model explained the least

amount of variation, approximately 21% (Table 3.3). Mean BRATIO was statistically higher for trees greater than 75 years old.

Table 3.3. Comparison of model coefficients, standard errors, and R^2 values for models examining the influence of site characteristics and tree defects on TPI, DBH, CPA, and BRATIO.

	Response variables			
variables	TPI	DBH	CPA	BRATIO
intercept	0.050 (0.067) ***	31.567 (2.713) ***	-20.012 (6.129) **	1.828 (0.124) ***
PSA	0.002 (0.000) ***	0.015 (0.013) ***	0.887 (0.045) ***	-0.001 (0.001) *
DAMAGE	-0.115 (0.041) **	2.930 (1.441) *		-0.107 (0.067)
SGR	-0.058 (0.028) *			-0.085 (0.047)
COLAP		2.899 (0.952) **	9.490 (3.385) **	
age 17-25	0.262 (0.069) ***	-15.224 (2.456) ***		-0.241 (0.114) *
age 26-50	0.176 (0.056) **	-7.502 (1.930) ***		-0.262 (0.095) **
age 51-75	-0.167 (0.081) *	-3.082 (2.851)		-0.355 (0.132) **
CEOC	-0.013 (0.039)	-0.157 (1.336)	10.085 (4.439) *	0.068 (0.064)
GLTR	0.041 (0.039)	-2.346 (1.283)	26.979 (4.371) ***	0.174 (0.063) **
TI	-0.140 (0.036) ***	3.436 (1.250) **	-7.042 (4.379)	0.253 (0.059) ***
site:ST		15.907 (1.401) ***	87.237 (4.911) ***	-0.315 (0.065) ***
adjusted R^2	0.28	0.48	0.67	0.21

Note: Standard errors are shown in parentheses. *C. occidentalis* (CEOC), *G. triacanthos* (GLTR), and *Tilia* spp. (TI). N=292
*** $p < 0.001$, ** $p < 0.01$, * $p < 0.05$

The effect of species differed in significance and magnitude depending on the response variable under investigation. Mean TPI and mean CPA were highest for *G. triacanthos* and lowest for *Tilia* spp. (Figure 3.7a & b) for all levels of PSA. *Celtis occidentalis* had the largest mean DBH (Figure 3.7b), whereas, *Tilia* spp. had the highest BRATIO compared to other species at the same amount of available pervious soil surface area (Figure 3.7d). Available soil surface area under the canopy had the steepest slope for *A. platanoides* and *Tilia* spp. in terms of current growth rate (Figure 3.7d).

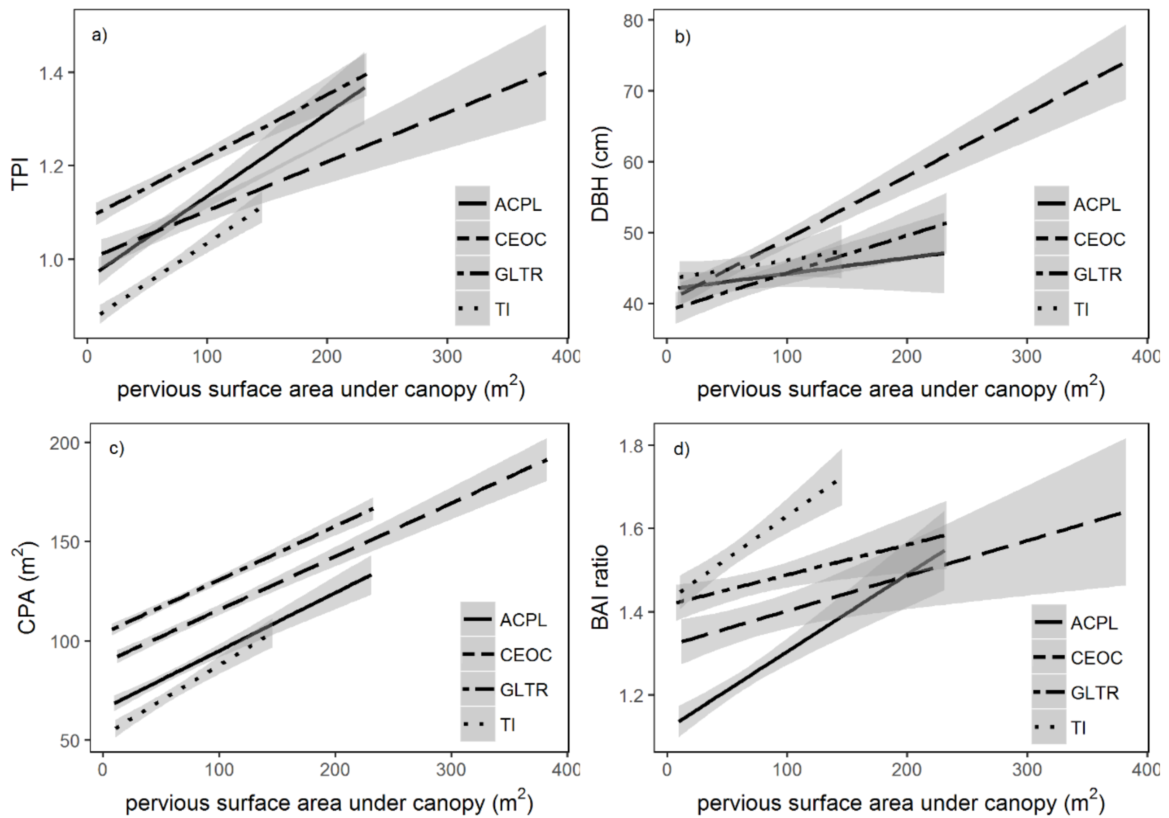

Figure 3.7 Model output for mean TPI, DBH, CPA, and BAI ratio by species on pervious surface area, based on model results presented in Table 4. Grey bans represent a 95% confidence interval. *C. occidentalis* (CEOC), *G. triacanthos* (GLTR), and *Tilia* spp. (TI). N=292

Discussion

Tree attributes

Assessment of tree species performance varied based on the growth metric analyzed. Park trees were statistically older on average than street trees. This difference was largely due to the age of the *C. occidentalis* population, although all species had park trees older than their street tree counterparts. There was no statistical difference in tree age for street trees and no street tree was greater than 50 years old. While there are likely street trees older than 50 years, they appear to be rare in the cities of Minneapolis and Saint Paul, and the reasons for a lack of older street trees cannot be determined from our data. A possible explanation is the arrival of Dutch elm disease to both cities in the 1963, resulting in the removal of thousands of diseased elms and subsequent replanting of different species along city streets (French, 1991). However, park trees older than 50 years old were observed, it is also possible that older street trees either out-grew their planting spaces, were in poor condition, damaged during construction, or were lost during wind-loading events, all reasons that may have required removal.

There was no statistical difference detected for DBH between park trees and street trees nor any difference between species. While no statistical difference was found between park and street trees it is worth noting that park trees had the largest DBH for all species. This is consistent with previous research showing a relationship between available growing space and tree space (Quigley M. E., 2004; Sanders, Grabosky, & Cowie, 2013). The only statistical differences were among species growing in parks. In parks, *G. triacanthos* had the lowest performance in terms of mean DBH compared to other species.

Tree diameter has been shown to increase with crown size and crown condition as higher photosynthetic potential leads to greater accumulation of carbon along the trunk (Drobyshev, et al., 2007). The expectation was that CPA would show a similar relationship as DBH; however, CPA differed statistically between park and street trees with street trees having the largest mean CPA. In terms of canopy performance, street trees out performed park trees. Our results are counter to previous studies that have shown trees growing in restricted growing space have smaller canopies on average (Day & Amateis, 2011; Sanders & Grabosky, 2014). Street trees in Minneapolis and Saint Paul have their crowns raised to a minimum height of approximately 4m to accommodate traffic. Pinkard et al. (1998) found canopies increased in size for *Eucalyptus nitens* two years after their crowns were raised through pruning if at least 50% of the canopy was maintained. Crown raising may be a possible explanation for the observed larger street tree canopies as more resources are diverted from accumulation along the trunk to expansion of the tree canopy. In terms of CPA performance, street trees outperformed park trees and both *C. occidentalis* and *G. triacanthos* outperformed *A. platanoides* and *Tilia* spp. (Figure 3.3). A diversion of carbon from the trunk to canopy could also explain the reduction in growth rate (BRATIO) over the last ten years of growth from street trees as compared to park trees (Figure 3.4 & Table 3.3:BRATIO). While street tree canopies are larger, they may also be in poorer condition, although condition of the canopy was not assessed. Poor crown condition was shown to reduce growth of *Quercus robur* in southern Sweden (Drobyshev, et al., 2007) and the differences in BRATIO may reflect that broad canopy of street trees does not necessarily indicate a more productive canopy in terms of carbon accumulation.

Street trees were taller on average than park trees. *C. occidentalis* and *G. triacanthos* in parks were the tallest on average. Crown raising was shown to increase tree height for *Tectona grandis* in Costa Rica (Viquez & Perez, 2005). Other studies have shown no impact to tree height after pruning 50% or less of the lower canopy (Pinkard & Beadle, 1998; Alcorn, et al., 2008). The literature on urban tree height response to pruning is lacking and pruning information for individual trees was not available.

Tree performance metrics

Urban tree performance has previously been assessed using canopy measures, DBH, or growth increment (Iakovoglou, et al., 2001; Day & Amateis, 2011; Sanders, et al., 2013; Sanders & Grabosky, 2014; Dahlhausen, et al., 2016; Sherman, et al., 2016). Here we used linear models to assess tree performance through analysis of CPA, DBH, BRATIO, and TPI. Variables important in explaining tree performance differed across the metrics evaluated (Table 3). Pervious soil surface area and species were important factors across all models. In all models except those developed for BRATIO, pervious surface area had a positive influence on the growth metric which is consistent with previous research (Iakovoglou, et al., 2001; Day & Amateis, 2011; Sanders, Grabosky, & Cowie, 2013; Sanders & Grabosky, 2014; Dahlhausen, et al., 2016; Sherman, et al., 2016;). The negative influence of PSA on BRATIO is contrary to previous research and all other models, although significant, indicated the effect of PSA on BRATIO was minimal.

Tree damage was included in models for TPI, DBH, and RBATIO (Table 3). In both TPI and BRATIO the variable DAMAGE had a negative influence as expected.

Loss of cambium and damage to trees has been shown to reduce tree growth and stability (Hauer, et al., 1994; Shortle, et al., 2003). Tree damage had a positive influence in the DBH model, which was unexpected. A possible explanation for the positive influence of damage to the trunk on DBH, could be the formation of callus tissue and reaction wood. To remain mechanically stable following an environmental stress, trees will form reaction wood. In angiosperms, a larger volume wood is grown opposite the injury or load and may extend vertically along the trunk to maintain mechanical stability (Du & Yamamoto, 2007). Stem girdling roots had a negative influence on TPI and BRATIO. The negative influence of SGR on above ground growth is consistent with existing research (Hulder & Beale, 1981; Wells, et al., 2006).

Competition with neighboring trees (COLAP) had a positive impact on both DBH and CPA (Table 3). This result is inconsistent with Iakovoglou, et al. (2001) who found a negative relationship between tree growth and proximity to neighboring trees within 9m. Our measure of canopy overlap may not be sensitive enough to detect true competition and may in fact be accounting for tree size. Trees observed in our study were between ~9m to 20m from a nearest neighbor, indicating that only larger trees would have canopies that overlapped.

The influence of age class on DBH and BRATIO was consistent with expectations as, DBH and BRATIO increased with increasing age. The increase in BRATIO is consistent with Johnson and Abrams (2009) who found that trees continue to increase in basal area as they age. In the TPI model, age class appears to have an inverse relationship with trees in the younger age classes out performing older trees (Table 3) and

may indicate a decline in tree condition with age. Age class did not add to the explanatory power for the CPA model.

Species performance varied based on the performance metric assessed. *Gleditsia triacanthos* had the highest performance in TPI and CPA, and the second highest performance in BRATIO, but the lowest performance in DBH models (Table 3, Figure 7). High performance of *G. triacanthos* in urban environments has been noted by other researchers (Koeser, et al., 2013; Swoczyna, et al., 2015). *Tilia* spp. showed the highest performance in models of DBH and BRATIO, but the lowest in terms of CPA and TPI. These performance differences in *Tilia* spp. are likely due to tree morphology where the canopy is oval to pyramidal (Wandell, 1989; Sullivan, Tilia americana, 1994) reducing the CPA in comparison to other species.

Soil compaction based on penetrometer readings was assessed in all models. Soil compaction above 2.3Mpa has been shown to negatively impact root growth outward and downward from the trunk (Day & Bassuk, 1994) and water infiltration into the soil has been shown to be limited in compacted soils (Gilman, et al., 1987; Gregory, et al., 2006; Bartens, et al., 2008); however, the influence on aboveground metrics of tree growth have been more difficult to demonstrate statistically. In our models soil compaction had a negative influence on the variable of interest (data not shown), but was neither significant nor did it add to the explanatory power of the final models. While relatively easy to measure, soil compaction may be a poor proxy for the actual soil information of interest to plant growth: soil moisture and oxygen diffusion in soils (Hodge & Boswell, 1993). Soil compaction in our sites was also a relatively uniform distribution and may not have had enough variability to detect statistical differences. Other studies have used bulk

density to assess soil compaction with varying statistical significance (Hodge & Boswell, 1993; Iakovoglou, et al., 2001; Day & Amateis, 2011), although most studies suggested increased compaction or bulk density have a negative influence on growth.

Conclusion

To successfully create a sustainable urban forest, managers must understand how tree performance is impacted by a variety of site characteristics. Understanding urban tree growth is an important tool not only for managers of urban street tree planting programs, but also for designers of urban infrastructure. Urban infrastructure often contains elements that include restricted planting spaces, and the ability to anticipate the growth response of trees in different sites can assist urban planners and natural resource managers to more effectively manage trees based on site characteristic to meet management objectives.

There are a growing number of studies investigating urban tree performance in different sites. Many studies use similar site characteristics: soil surface area, soil volume, bulk density, soil compaction, competition, yet each use different measures of performance, from biomass to stem caliper to canopy projection area and tree ring analysis (Hauer, et al., 1994; Iakovoglou, et al., 2001; Sanders, et al., 2010; Day & Amateis, 2011; Dahlhausen, et al., 2016; Sherman, et al., 2016). Assessing tree performance or value based on any one metric may be misleading as tree growth and form can vary by species and site, and management objectives can vary from tree biomass accumulation to canopy cover. As demonstrated here two different species were highlighted as having the best performance, *G. triacanthos* for CPA and *Tilia* spp. for DBH. The adaption of TVI (Voelker, Muzika, & Guyette, 2008) to create TPI was our

attempt to create a unified variable capable of identifying tree performance in urban environments by incorporating ideas of a sustainable tree architecture (TVI) and a metric of potential future sustained growth (BRATIO). While the TPI model did not perform the best in terms of variability explained (only 28% compared to the CPA model 67%) it serves to further highlight the challenges in modeling urban tree growth and illustrates the importance of selecting tree performance models based on a communities' most valued objective. Additional research is needed to refine TPI if it is to prove valuable in future tree assessments.

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CONCLUSION

Urban environments include components of both living and non-living infrastructure designed to accommodate and benefit the human inhabitants. City managers and urban forest managers often need to find a balance when managing potentially conflicting infrastructure needs (e.g. sidewalk accessibility and benefits provided by trees). Broadly, the objective of this dissertation was to examine how tree growth is influenced by common factors observed as part of the urban environment in order to provide urban forest practitioners with scientifically sound information to aid in the planning and management urban trees capable of resilience and longevity in urban settings. The results presented in the preceding chapters can guide species selection and site design to help maximize the potential of trees to provide long-term benefits to urban denizens.

This research supported some of the findings of previous authors that tree growth response differs by species and is influenced by complex interactions between natural environmental forces and features of a built environment (Dahlhausen, Biber, Rotzer, Uhl, & Pretzsch, 2016; Monteiro, Levanic, & Doick, 2017; Fahey, Bialecki, & Carter, 2013; Quigley, 2004). Chapter 1 demonstrated that trees growing along streets have growth responses to climate factors that differed significantly from trees growing in parks within the same species. Street trees were, generally, less affected by temperature compared to park trees, and *Tilia* spp. in urban environments demonstrated little sensitivity to climatic features in terms of their growth response (Chapter 1). The differences in resistance and resilience to drought over time highlight a challenge in

understanding tree growth over decades. While the greater resistant and resilience seen in the earlier drought could be related age, it seems more plausible that the resistance and resilience of trees are more directly related to tree size as size is likely to have a greater impact on the balance of resources required to maintain an existing tree body and the resources needed to continue previous levels of growth.

Some of the results from Chapters 1 and 3 challenge the existing literature on tree size and growth as related to the site conditions (Sanders, et al., 2013; Quigley, 2004). Street trees were found to be larger and growing faster on average than trees growing in city parks, even though there was no statistical difference in tree age (Chapter 3). The exact reason of the increased growth of street trees over parks trees could not be determined directly from the data, but may be influenced by urban hydrology that potentially shunts more urban runoff over the boulevard planting space of street trees providing increased access to water.

While several studies have investigated the influence of construction on tree mortality and condition (Hauer, et al., 1994; Koeser, et al., 2013), Chapter 2 provided direct evidence of growth reduction post-construction for the four tree species investigated. Chapter 2 also detailed the length of time trees required to recover to pre-construction levels of growth. Again, *Tilia* spp. stood out from the other species investigated and proved to be highly resilient to the influences of construction. However, given the potential instability of trees after root severance, caution is still recommended when evaluating tree selection for areas where construction activities are probable.

Overall, the results of this dissertation indicates that tree growth response to climate, construction, and urban site characteristics is detectable and can vary by species

and tree attributes of age and size. Tree performance was also species and site dependent, but more than that, tree performance needs to be assessed based on the objectives to be met. Urban tree benefits encompass a wide range of objectives from aesthetics to biomass accumulation or energy reduction. The provision of benefits by urban trees should be weighed against the potential negative impacts of infrastructure disruption (e.g. sidewalk lifting, storm debris and damage, etc.) and guided by the principals of species selection. This emphasizes the need for species selection based on the desired objectives and the available site and should seek to maximize urban infrastructure and tree performance over the long-term.

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