Wissenschaftszentrum Weihenstephan für Ernährung,
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# Structure and Growth of Urban Trees and Their Impacts on the Environment - From Assessment to Model

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*"There are those who say that trees shade the garden too much, and interfere with the growth of the vegetables. There may be something in this: but when I go down the potato rows, the rays of the sun glancing upon my shining blade, the sweat pouring down my face, I should be grateful for shade."*

Charles Dudley Warner

# Acknowledgements - Danksagung

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# List of Abbreviations

| | |
|---|---|
| a | Intercept |
| AIC | Akaike Information Criterion |
| $A_S$ | Shaded Area |
| AT | Air Temperature |
| b | Slope |
| BVOC | Biogenic Volatile Organic Compounds |
| CF | Bias Correction |
| CPA | Crown Projection Area |
| CPS | Circular Paved Square |
| cr | Crown Radius |
| cv | Crown Volume |
| Dbh | Diameter at Breast Height |
| Doy | Day of the Year |
| EPS | Expressed Population Signal |
| f | Correction Factor for the Crown Form |
| GAM | Generalized Additive Model |
| h | Tree Height |
| $h_c$ | Crown Height |
| IPCC | Intergovernmental Panel on Climate Change |
| LA | Leaf Area |
| LAI | Leaf Area Index |
| MA | Moving Average |
| m. a.s.l. | Meters Above Sea Level |
| n | Number of Measurements |
| OGS | Open Green Square |
| OLS | Ordinary Least Square |
| PET | Physiological Equivalent Temperature |
| $PE_T$ | Potential Evapotranspiration |
| PM | Particulate Matter |
| Prec | Precipitation |
| RMA | Reduced Major Axis |
| RSE | Residual Standard Error |
| RWI | Ring Width Index |

| | |
|---|---|
| s | Smooth Function for every Slope Variable |
| SD | Standard Deviation |
| SE | Standard Error |
| SEA | Superposed Epoch Analysis |
| SFD | Sap Flux Density |
| SMP | Soil Matric Potential |
| SPEI | Standardized Precipitation Evapotranspiration Index |
| ST | Soil Temperature |
| SVF | Sky View Factor |
| $tra_a$ | Yearly Transpiration Sum |
| Transp | Transpiration |
| UHI | Urban Heat Island Effect |
| UV | Ultraviolet Radiation |
| Vol | Crown Volume |
| VPD | Vapor Pressure Deficit |
| WS | Wind Speed |
| x | Predictor Variable |
| Y | Elevation Angle of the Sun |
| y | Outcome Variable |

# List of Species

| | | |
|---|---|---|
| *Aesculus hippocastanum* | Horse-chestnut | Gewöhnliche Rosskastanie |
| *Fagus sylvatica* | European beech | Rotbuche |
| *Fraxinus americana* | White ash | Weiß-Esche |
| *Fraxinus pennsylvanica* | Green ash | Rot-Esche |
| *Gleditsia triacanthos* | Honey-locust | Amerikanische Gleditschie |
| *Platanus orientalis* | Oriental plane | Morgenländische Platane |
| *Pyrus calleryana* | Callery pear | Chinesische Wildbirne |
| *Robinia pseudoacacia* | Black locust | Robinie |
| *Tilia cordata* | Small-leaved lime | Winterlinde |

# Abstract

Research focusing on urban trees and their ecosystem services is still a recent field of scientific inquiry, which is getting increased attention by city councils and urban planners, not least due to climate change. However, actual data on the growth and impacts of common urban tree species in Central Europe is still lacking. This thesis deals with the structure and growth patterns of two major urban trees in Germany, small-leaved lime *Tilia cordata* Mill. and black locust *Robinia pseudoacacia* L. Using regression analysis, an overview of the range of growth and tree structures in various age classes was generated, illustrating the full lifecycle of both tree species growing at three typical urban settings (parks, public squares and street canyons). The growth of *T. cordata* and *R. pseudoacacia* regarding their drought tolerance and recovery was analyzed retrospectively with a dendrochronological study, revealing a surprisingly high drought tolerance of *T. cordata* due to the anisohydric character of this species though a faster recovery of *R. pseudoacacia* in the consecutive year after a drought event was found caused by its isohydric strategy of water conservation. In a further step, the influence of the local climate on the growth patterns of *T. cordata* was assessed. Data collection was performed in Munich and Würzburg, two cities differing considerably in average yearly precipitation, with Würzburg being far drier than Munich (580 mm yr$^{-1}$ to 1000 mm yr$^{-1}$). The results show a slightly better growth of both species in Munich, especially during drought as better water availability promotes the growth of urban trees in Munich compared to Würzburg. In an additional study with high-resolution growth data over the exceptionally hot and dry summer 2015, a significant relationship of the microclimate with the growth of *T. cordata* was found. While the precipitation rate was positive correlated with tree growth, all other measured meteorological variables, e.g., air and soil temperature, vapor pressure deficit, wind speed and transpiration as well as soil matric potential were negatively correlated with the growth of *T. cordata*. Most pronounced effects on the tree growth had the transpiration, indicating a trade-off between growth, transpiration and cooling. Due to the high transpiration rate, *T. cordata* can provide high rates of evapotranspiration and hence cooling, however depleting the internal water storage when the soil water availability decreases. Moreover, anisohydric species such as *T. cordata* might suffer from growth reductions several years after a drought. In the future, tree species that achieve a better balance between growth, transpiration and evaporative cooling should be preferred to mitigate the effects of climate change for a better thermal comfort in cities. Finally, the three ecosystem services, carbon storage, shading and cooling by evapotranspiration, were calculated for both species in Munich and Würzburg, highlighting the potential role of urban trees for

moderating the urban climate and contributing to climate change mitigation and adaptation in urban areas. For sustainable planning of the urban forest, future tree selection should be based on in-depth knowledge of the demands of the tree on its planting site as well as on the demands of the tree regarding its benefits for the city climate.

# Zusammenfassung

In den vergangenen Jahrzehnten ist die Forschung zu Stadtbäumen und deren Ökosystemleistungen vermehrt in den Fokus der Wissenschaft und Stadtplanung gerückt, denn auch aufgrund des Klimawandels sind Städte und Planer an grünen Lösungen für ein angenehmes Stadtklima besonders interessiert. Es fehlen jedoch insbesondere für Mitteleuropa Daten bezüglich des Wachstums und der Dimensionen von häufigen Stadtbaumarten. Die vorliegende Arbeit hat sich deshalb mit dem Wachstum und den Strukturdimensionen zweier häufig gepflanzten Baumarten in der Stadt beschäftigt, der Winterlinde *Tilia cordata* Mill. und der Robinie *Robinia pseudoacacia* L. Mit allometrischen Wachstumsbeziehungen wurden Tabellen erstellt, die das Wachstum und die Baumdimensionen in verschiedenen Altersklassen widergeben. Diese Tabellen beschreiben den Lebenszyklus beider Baumarten an drei häufigen Freiraumtypen der Stadt, in Parks, an öffentlichen Plätzen und in Straßenzügen. Weiterhin wurde retrospektiv das Wachstum von *T. cordata* und *R. pseudoacacia* in Bezug auf ihre Austrocknungstoleranz und Erholungsfähigkeit mit einer dendrochronologischen Studie untersucht. Es wurde eine überraschend hohe Austrocknungstoleranz für *T. cordata* nachgewiesen, welche auf dem anisohydrischen Charakter der Art beruht, während *R. pseudoacacia* aufgrund ihrer isohydrischen Strategie bezüglich des Wasserhaushalts eine schnelle Erholungsfähigkeit in dem darauffolgenden Jahr des Trockenereignisses aufwies. Zusätzlich wurden der Einfluss des lokalen Klimas auf die Wachstumsreaktionen von *T. cordata* genauer untersucht. Die Datenaufnahme erfolgte in München und Würzburg, zwei Städte die sich in ihrem Klima essentiell unterscheiden. So ist Würzburg aufgrund eines jährlichen durchschnittlichen Niederschlags von 580 mm gegenüber München (1000 mm) deutlich trockener. Die Ergebnisse zeigten ein geringfügig höheres Wachstum beider Arten in München, wobei insbesondere während einer Trockenperiode die höhere Wasserverfügbarkeit das Wachstum von Stadtbäumen in München im Vergleich zu Würzburg besser fördert. In einer weiteren Studie mit hochaufgelösten Wachstumsdaten während des außergewöhnlich heißem und trockenen Sommers 2015 im Zentrum von München wurde ein signifikanter Zusammenhang des spezifischen Mikroklimas mit dem Wachstum von *T. cordata* nachgewiesen. Während die Niederschlagsrate einen positiven Zusammenhang mit dem Wachstum hatte, waren alle anderen gemessenen meteorologischen Variablen wie die Bodentemperatur, Lufttemperatur, Dampfdruckdefizit, Transpiration, Windgeschwindigkeit und Bodenpotential negativ mit dem Wachstum korreliert. Den größten Einfluss auf das Wachstum hatte jedoch die Transpiration, was auf einen Kompromiss zwischen Wachstum, Transpiration

und Abkühlungswirkung schließen lässt. *T. cordata* kann durch die gezeigte hohe Transpirationsleistung eine dauerhaft hohe Abkühlungswirkung erzielen, die jedoch bei einer geringen Bodenwasserverfügbarkeit zu einer Erschöpfung der internen Wasserspeicher führt. Auch kann eine anisohydrische Art wie *T. cordata* Wachstumseinbrüche mehrere Jahre nach einem Trockenereignis erleiden. Deshalb sollten in Zukunft Arten mit einer besseren Balance von Wachstum, Transpiration und damit Kühlungswirkung bevorzugt werden, denn so können die Auswirkungen des Klimawandels auf das Stadtklima erfolgreicher gemildert werden. Letztendlich wurden drei Ökosystemleistungen, genauer die Kohlenstoffspeicherung, die Beschattung und die Abkühlungswirkung durch Evapotranspiration für beide Arten in München und Würzburg berechnet. Die Ergebnisse bestätigen die potentielle Rolle von Stadtbäumen für ein angenehmes Stadtklima und eine erfolgreiche Anpassung und Minderung an die Folgen des Klimawandels. Zukünftige Baumartenwahl sollte sowohl an den Ansprüchen des Baumes an seinen Pflanzort als auch an den Ansprüchen an den Baum in Bezug auf dessen Ökosystemleistungen erfolgen, um nachhaltiges Grünraummanagement zu gewährleisten.

# I. Cumulative Thesis

**This dissertation is based on investigations that were published and submitted in the following four original articles:**

- **Moser, A., Rötzer, T., Pauleit, S., Pretzsch, H (2015): Structure and ecosystem services of small-leaved lime (*Tilia cordata* Mill.) and black locust (*Robinia pseudoacacia* L.) in urban environments. Urban Forestry & Urban Greening 14, 1110-1121**

Summary:

Knowledge of the structure and morphology of common urban trees is scarce, particularly of the full lifecycle of a tree. The present and future structural dimensions of urban tree species are of an increasing interest because tree growth is associated with its ecosystem services (e. g. carbon storage, pollutant filtering, cooling, shading, biodiversity). The purpose of this study was to characterize the dimensions of two urban tree species (small-leaved lime, *Tilia cordata* Mill. and black locust, *Robinia pseudoacacia* L.) and to predict future structural dimensions based on the diameter at breast height dbh and tree age. Tree dimensions of 425 *T. cordata* and *R. pseudoacacia* trees in three different site categories (park, public square and street) were collected in the two southern German cities Munich and Würzburg. Both differ in their climate characteristics, Würzburg is with 400 mm less precipitation per year considerably drier than Munich. Regression equations were developed for tree height, crown diameter, crown height, crown volume, crown projection area, and open surface area of the tree pit. The data revealed strong relationships ($r^2 > 0.7$) between crown diameter, crown volume, crown projection area, crown height, tree pit for both species, and tree height of *T. cordata*. Based on tree dimensions and the leaf area index (LAI), three ecosystem services (carbon storage, shading, and cooling effects) were estimated for the analyzed trees. The results indicated that urban trees considerably improved the climate in cities, with carbon storage, shading, and cooling of single trees showing a direct relationship with LAI and age. Highest ecosystem services were found for *R. pseudoacacia* trees growing in parks, while *T. cordata* provided more cooling with higher age compared to *R. pseudoacacia*. The associations of tree growth patterns identified in this study can be used as guidelines for tree planting in cities and their ecosystem services; they may improve the management and planning of urban green areas.

- **Moser, A., Rötzer, T., Pauleit, S., Pretzsch, H (2016): The Urban Environment Can Modify Drought Stress of Small-Leaved Lime (*Tilia cordata* Mill.) and Black Locust (*Robinia pseudoacacia* L.). Forests 7, 71**

Summary:

The urban environment characterized by various stresses like dense soil with limited rooting space, less water availability and nutrients, further high temperatures and radiation, pollutants, insect infestations and pruning poses challenges to trees. In particular, water deficits and high temperatures can cause immense drought stress to urban trees, resulting in reduced growth and finally tree die-off. Drought-tolerant species are expected to be resilient to these conditions and are therefore advantageous over other, more susceptible species. However, the drought tolerance of urban trees in relation to the specific growth conditions in urban areas remains poorly researched. This study aimed to analyze the annual growth and drought tolerance of two common urban tree species, namely small-leaved lime (*Tilia cordata* Mill.) and black locust (*Robinia pseudoacacia* L.), in two cities in southern Germany (Munich and Würzburg) in relation to their urban growing conditions. Both cities differ considerably regarding their climate characteristics; Würzburg is characterized by 400 mm/yr less precipitation than Munich. With increment cores, a dendrochronological study about the growth patterns of these tree species in relation to their site conditions was conducted. Marked growth reductions during drought periods and subsequent fast recovery were found for *R. pseudoacacia*, whereas *T. cordata* exhibited continued reduced growth after a drought event, although these results were highly specific to the analyzed city. We further show that individual tree characteristics and environmental conditions significantly influence the growth of urban trees. Especially canopy openness—representing the light availability—and other aspects of the surrounding environment (water supply and open surface area of the tree pit), tree size, and tree species significantly affect urban tree growth and can modify the ability of trees to tolerate the drought stress in urban areas. These findings were strongly dependent on the species, with *T. cordata* being less affected by shade as a shade-tolerant species in comparison with *R. pseudoacacia*, a light-demanding species experiencing growth deficits when growing in shade. Sustainable tree planting of well adapted tree species to their urban environment ensures healthy trees providing ecosystem services for a high quality of life in cities.

- **Moser, A., Rahman, M., Pretzsch, H., Pauleit, S., Rötzer, T. (2016): Inter- and intraannual growth patterns of urban small-leaved lime (*Tilia cordata* mill.) at two public squares with contrasting microclimatic conditions. International Journal of Biometeorology, DOI 10.1007/s00484-016-1290-0**

Summary:

The effects of urban conditions on tree growth have been investigated in an increasing number of studies over the last decades, emphasizing the harsh environment of cities. Urban trees often grow in highly paved, compacted sites with consequently less soil moisture and nutrients, higher soil temperatures, and greater vapor pressure deficits. However, there is still a knowledge gap regarding the impact of those detrimental site conditions, especially in respect of drought events, on the growth patterns of urban trees. The present study investigated the structure and growth of the common urban tree species small-leaved lime (*Tilia cordata*) at a highly paved public square (CPS) compared with a contrasting more open, greener square (OGS) in the city center of the southern German city Munich. Continuously measured high precision dendrometer data along with meteorological data of the extreme dry and warm summer 2015 as well as dendrochronological data of the sampled trees were investigated to analyze tree growth during a drought year. The results highlight different tree dimensions and growth patterns of the trees at both sites, influenced by tree age and distinct site conditions. While the trees at OGS grew up to 2.5 mm until mid of August, the trees at CPS had only 0.4 mm diameter increment. After the initial expansion at CPS, tree diameter contracted again during summer to the point of shrinkage (up to 0.8 mm) at the end of our investigation. Further drought year analysis confirmed the patterns of significant growth reductions in the consecutive two years following the drought. A correlation analysis revealed that transpiration (r = −0.6), air temperature (r = −0.6), and vapor pressure deficit (r = −0.6) were negatively correlated with the daily diameter growth, whereas precipitation had a strong positive effect (r = 0.3). Due to high transpiration rates associated with anisohydric water use behavior, *T. cordata* was able to provide evaporative cooling even during drought. However this water stress behavior resulted in a substantial growth decline afterwards especially at paved sites like CPS. Our results suggest selection of tree species, such as those with isohydric water use behavior, which may achieve a better balance between growth, transpiration, and hence evaporative cooling.

- **Moser, A., Rötzer, T., Pauleit, S., Pretzsch, H. (2016): Stadtbäume: Wachstum, Funktionen und Leistungen – Risiken und Forschungsperspektiven. Allgemeine Forst- und Jagdzeitung AFJZ, in Begutachtung (under review)**

Summary:

Stadtbäume und grüne Infrastrukturen sind ein wesentlicher Bestandteil des Stadtbildes, sie stellen den ästhetischen und erholungswirksamen Mittelpunkt der Stadt dar. Doch nicht nur durch ihre Ästhetik prägen Bäume eine Stadt, sondern auch durch ihre Struktur und Erscheinung sowie die Funktionen, die sie für das Stadtklima erbringen. Neben positiven Eigenschaften wie Kohlenstoffbindung, Abkühlungswirkung, Beschattung, Biodiversität, sowie Abflussminimierung dürfen jedoch auch die negative Eigenschaften des Stadtgrüns wie Kosten für die Pflanzung und Pflege, Allergene und Schäden durch herabfallende Pflanzenteile bzw. Verschmutzungen nicht ignoriert werden. Darüber hinaus ist der Wuchsraum der Stadt durch vielfältige Faktoren gekennzeichnet, welche sich negativ auf die Vitalität, das Wachstum und die Leistungen der Bäume auswirken können und zu erhöhten Kosten führen können. Zu nennen sind hierbei hohen Temperaturen, geringe Wuchsraum-, Nährstoff,- und Wasserverfügbarkeit, Schadstoffbelastungen, Beschattung durch Gebäude und Verletzungen. Auch stellen der urbane Wärmeinseleffekt und die Auswirkungen des Klimawandels für Stadtbäume langfristig wachstumshinderliche Faktoren dar, die in der Grünraumplanung immer mehr im Vordergrund stehen. Städte und Forschung versuchen deshalb, nachhaltige Konzepte zu entwickeln, um weiterhin einen gesunden, vielfältigen und attraktiven Baumbestand in der Stadt zu erhalten. Diese Studie beschäftigt sich mit dem aktuellen Stand des Wissens und stellt die positiven und negativen Aspekte der Stadtbäume dar. Die Wirkungen von Stadtbäumen auf den Menschen und das Stadtklima, aber auch die Effekte des Standortklimas auf den Baum werden diskutiert. Dabei werden weiterhin die Folgen des Klimawandels aufgezeigt sowie moderne Konzepte und zukünftige Forschungsperspektiven dargestellt und analysiert.

The above listed articles are attached in the appendix. Hereinafter, reference to the original articles is made by citation.

**Authors contributions:**

For the above listed publications I-II and IV, the first author A. Moser conducted major parts of data collection, supported by technical staff and students. All data analyses and calculations were done by the first author as well as writing and composing the manuscript. H. Pretzsch, S. Pauleit and T. Rötzer initiated the project 'City Trees', provided the ideas and concepts for the project and the articles. T. Rötzer prepared the model for the calculation of the described eco-system services in paper I. M. Rahman initiated the project 'green lab Munich' presented in paper III and carried out data collection with help of A. Moser. Data evaluation and analysis of paper III was done by the first-author. All co-authors contributed to the design and content of the paper by scientific advice, discussion and language editing.

# II. Thesis: Structure and Growth of Urban Trees and Their Impacts on the Environment - From Assessment to Model

## 1. Introduction

In 2015, Germany suffered from one of the driest and hottest summer for the past decades, especially affecting the living conditions in cities. Particularly, the months of July, August and September were characterized by very low precipitation rates, as illustrated in the climate of Munich: while the average precipitation of the month July is usually around 110 mm (1985-2014), in July of 2015 only 20 mm of rain was recorded (Figure 1). A similar pattern was observed for the months of August and September. Moreover, the temperature in July was around 3°C higher than in the past 30 years.
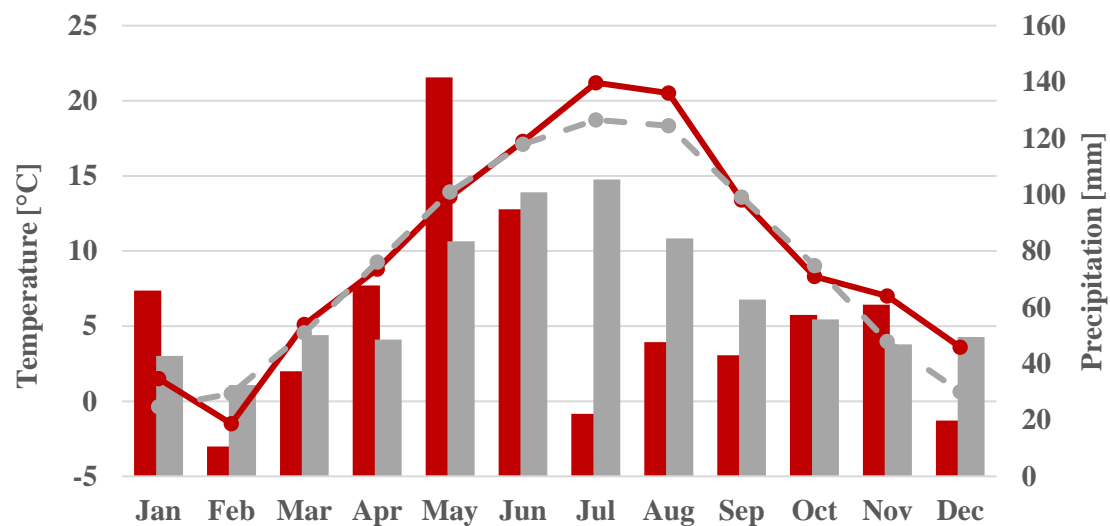


**Figure 1: Monthly temperature (red line) and precipitation (red bars) of Munich in 2015 as well as the average temperature (grey line) and average precipitation (grey bars) of Munich in 1985-2014 (data source: DWD (2016))**

A main reason for the warm climate in large cities like Munich is the urban heat island effect UHI (Oke, 1987). The specific climate of a city varies in comparison to its rural surroundings: in particular the inner city is warmer and dryer, typically with high temperatures during the night. Higher temperature maxima and high thermal storages during day and night are characteristics of large cities and the urban heat island effect. There are various reasons for the urban heat island effect. The higher temperatures are mainly due to the absorption of incoming solar radiation of buildings, which reradiate great parts of the incoming solar radiation as heat. Moreover, sparse vegetation and high rates of paved surfaces result in less transpiration (Collier, 2006; Kuttler, 2004; Oke, 1982). In Figure 2, the urban heat island is graphically illustrated, the

reradiated incoming solar radiation as heat by the highly paved surfaces and buildings is displayed. Less wind movement keeps the warmer air within the city boundaries, creating the urban boundary layer (Oßenbrügge & Bechtel, 2010). Consequences of the UHI are increased costs for cooling and peak electricity demands as well as intensified pollution problems, which increase the urban footprint and can cause human discomfort and health problems (Hassid et al., 2000; Santamouris et al., 2001; Santamouris et al., 2007; Santamouris et al., 2011). The effects of the urban heat island on the human thermal comfort are well researched (Luber & McGeehin, 2008; Sakka et al., 2012), since the urban heat island is most pronounced during midsummer. During nighttime in summer, the thermal strain is especially high for humans, causing sleep problems and diminished recovery during the night (Santamouris, 2014; Tan et al., 2010).
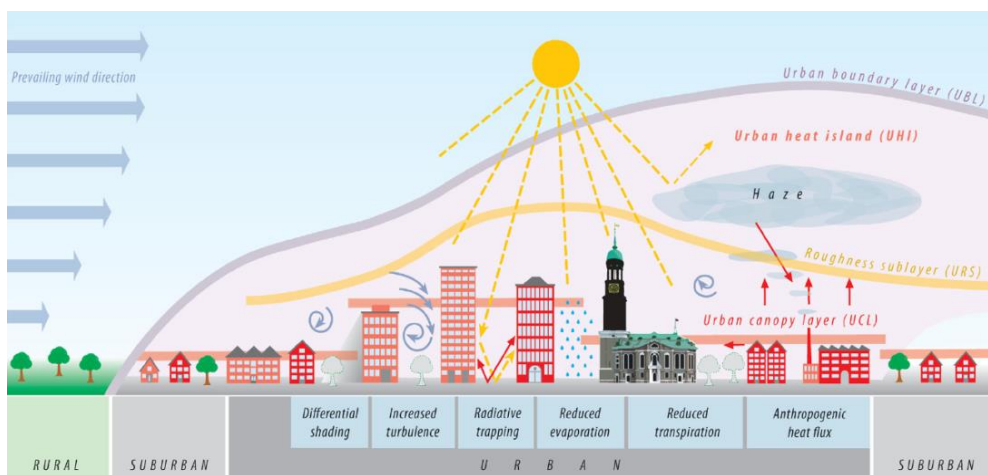


**Figure 2: Urban heat island effect in the city by Oßenbrügge & Bechtel (2010)**

However, the occurrence of very hot summers with low precipitation rates is also an effect of climate change, a widely accepted global phenomenon characterized by higher temperatures and altered precipitation rates as well as frequent and intense extreme climatic events like droughts, floods, and storms (Della-Marta et al., 2007; IPCC, 2007). These extreme events influence all ecosystems worldwide (Cavin et al., 2013; Kramer & Kozlowski, 1979; Orwig & Abrams, 1997). Water and nutrient limitations are direct consequences of drought, leading to low-growth episodes of vegetation (Pretzsch et al., 2013; Rötzer et al., 2012). Forecasts predict an increase of climate change effects and problems like droughts and extreme weather events. This trend can be observed when analyzing the occurrence of drought years since the beginning of the 21st century. When comparing the past drought years of Munich from 1985 to 1999 with those from 2000 to 2015, 11 years with a negative drought index SPEI (standardized precipitation evapotranspiration index) occurred in the 21th century, while only four years with a negative SPEI were recorded from 1985 to 1999 (Figure 3).
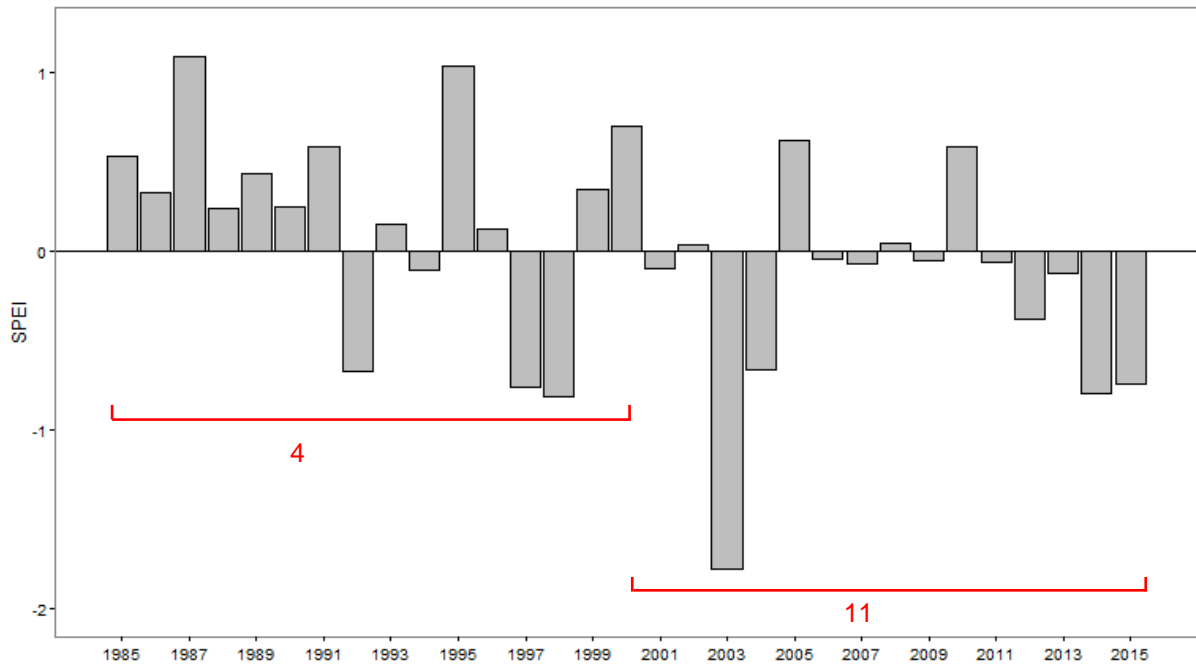
**Figure 3: Drought index SPEI (standardized precipitation evapotranspiration index) of Munich from 1985 to 2015**

The extremely hot summers of 2003 and 2015 may also be considered a result of climate change. The 'Intergovernmental Panel on Climate Change' (IPCC) predicts a warming of the annual average temperatures by 2 to 4.5°C for Germany by 2071 to 2100 based on the emission scenario A1B (IPCC, 2007). This trend is followed by a greater number of summer days (temperature maxima > 25°C), hot days (temperature maxima > 30°C) and tropical nights (temperature > 20°C) as well as longer dry periods together with less precipitation especially in summer. For example, during the period 1961 to 1990 in Munich 30 to 35 summer days per year, two to four hot days and no tropical days were recorded. Climate predictions expect an increase of summer days from 27 to 33 days more than today, 18 to 22 more hot days than today and seven to nine tropical nights in Munich by 2070 to 2100 (scenario A1B, 50% percentile) (DWD, 2016). Therefore, climate change will amplify the effects of the urban heat island in cities with a greater warming, higher maximum temperatures and higher thermal energy storage. Future climates with altered temperatures and precipitation as well as extreme weather events will affect the living conditions of the urban populations and can be detrimental for human quality of life (Fahey et al., 2013; Meehl et al., 2007). Especially risk groups like older people and children will suffer the most from extreme heat waves. Extreme heat conditions can have effects on health due to heat stress, UV radiation, air pollution and heat strokes (Matzarakis & Amelung, 2008). To evaluate the heat stress for humans, several indices were generated, with the physiological equivalent temperature (PET) as one of the most common applied indices. It

is a thermal index derived from the human energy balance and is well suited for the evaluation of the thermal component of different climates (Matzarakis et al., 1999).

To mitigate the effects of climate change on the climate of cities and the thermal comfort of humans, cities need to find effective solutions. The use of green infrastructure to moderate the impacts of the urban heat island and climate change on cities and humans has been researched and discussed recently. Science focuses mainly on the services green infrastructure can provide for the city climate. Vegetation in cities like trees, shrubs, grasses, facades and roof greenings stores and sequesters carbon (Davies et al., 2011; Nowak & Crane, 2002; Yoon et al., 2013; Zheng et al., 2013), cools its environment by transpiration and shading (Akbari et al., 2001; Armson et al., 2013a; Leuzinger et al., 2010; Rahman et al., 2015), improves the air quality by pollutant filtering (Bolund & Hunhammar, 1999; Ennos et al., 2014; Nowak et al., 2006; Ross et al., 2011), provides noise and wind speed reduction (Bolund & Hunhammar, 1999; Ross et al., 2011), mitigates rainwater run-off (Armson et al., 2013b; McPherson et al., 1994), increases biodiversity (Blair, 1996; Melles et al., 2003; Sandström et al., 2006) and offers aesthetic value such as amenity and life quality (Bolund & Hunhammar, 1999; Ennos et al., 2014). Figure 4 summarizes the services and functions vegetation in urban environments provides. Green infra-structure is therefore suggested as a multifunctional solution that not only mitigates the negative effects of climate change but also concurrently provides multiple benefits to society.

However, even if science mainly focuses on the positive effects of green infrastructure, negative consequences and costs need to be taken into account as well. Urban trees can cause material and personal damages due to falling trees and breaking branches (Dobbs et al., 2011). Trees in cities can host hazardous diseases and insect pests (Carinanos & Casares-Porcel, 2011), they can emit allergenic pollen and biogenic volatile organic compounds BVOCs, increase humidity and create green waste due to debris and litter (Carinanos & Casares-Porcel, 2011; Dobbs et al., 2011; Escobedo et al., 2011; Lyytimäki & Sipilä, 2009; Nowak & Dwyer, 2000). The costs for pruning, planting and maintaining are immense. For example, Soares et al. (2011) calculated $1.9 million in costs for urban tree management in Lisbon, Portugal.
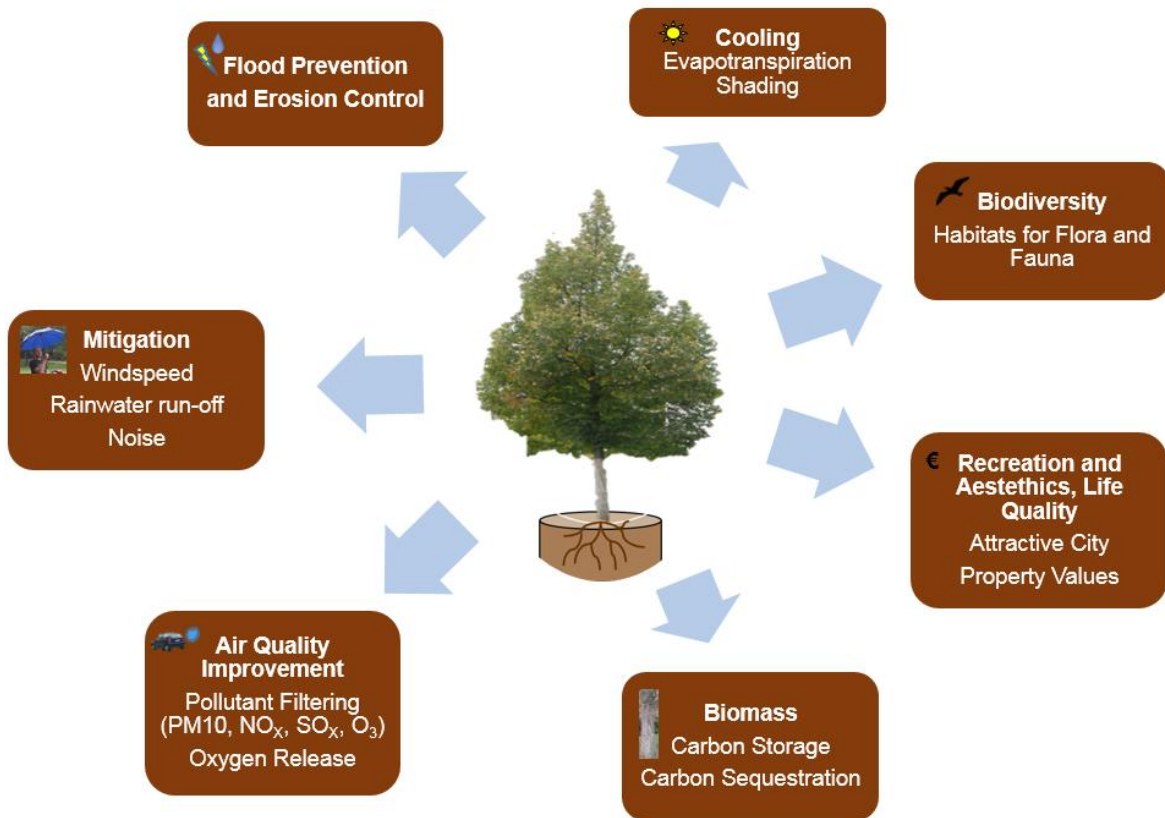
**Figure 4: Ecosystem services of urban vegetation (Moser et al. under review) adapted from Dobbs et al. (2014), Tyrväinen et al. (2005) and Bolund & Hunhammar (1999)**

An increasing number of researchers worldwide is trying to fill the knowledge gap about the quantity and quality of ecosystem services of different types of urban vegetation vegetation—trees in particular—by conducting city-wide studies on the growth, costs and benefits of urban tree stands of a city or studies based on the growth and services of single trees. Moreover, several studies used modeling approaches to answer questions about the structure, diversity and benefits of urban trees (Armson et al., 2013a; Pataki et al., 2011; Rahman et al., 2015; Rust, 2014; Yoon et al., 2013). There is an increasing demand of knowledge and understanding on how trees affect their environment in terms of their benefits for the city climate. For the past two decades, for example, pioneer projects such as the 'Chicago Urban Forest Climate Project' and in particular the software 'i-Trees' have been attempting to quantify the ecosystem services of green vegetation and urban trees and estimate their monetary value (McPherson et al., 1997; Nowak & Crane, 2002; Nowak et al., 1994). Table 1 displays the values of the urban trees in Chicago, which were estimated by calculating the air pollution removal, carbon storage and sequestration as well as energy reduction and reduction of carbon dioxide emission, resulting in a compensatory value of $51.2 billion.

**Table 1: Results of the Chicago Urban Forest Climate Project adapted from Nowak et al. (2013)**

| Feature | Measured |
|---|---|
| Tree number | 157,142,000 |
| Canopy cover (trees and shrubs) | 21% |
| Tree cover | 15.50% |
| Trees smaller than 6 inches dbh (%) | 73.30% |
| Pollution removal | |
|   - Trees | 18,080 tons per year ($137 million per year) |
|   - Trees and shrubs | 24,170 tons per year($183 million per year) |
| VOC emissions | 11,976 tons per year |
| Carbon storage | 16.9 million tons ($349 million) |
| Carbon sequestration | 677,000 tons per year ($14 million per year) |
| Building energy reduction | $44 million per year |
| Reduced carbon emissions | $1.3 million per year |
| Compensatory value | $51.2 billion |

Similar studies have been conducted in Lisbon (Soares et al., 2011) and in four cities in the UK (Rumble et al., 2014). Other researchers have concentrated on the quantity of single ecosystem services that different species can provide. For example, Rahman et al. (2015) analyzed the cooling efficiency of five urban tree species and found the greatest cooling abilities for *Pyrus calleryana* (Figure 5).
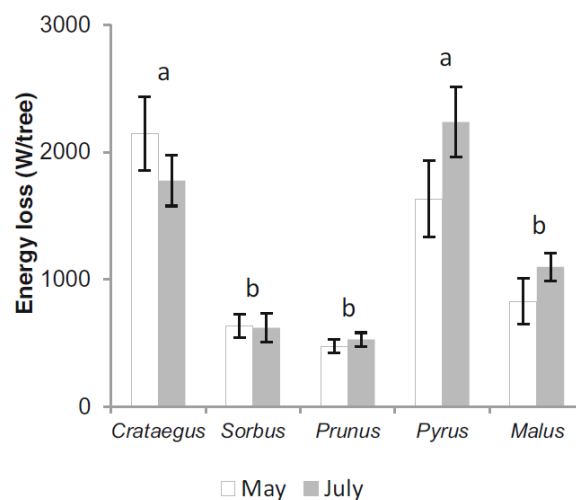


**Figure 5: Evapotranspirational cooling as energy loss per tree of five different tree species grown on different streets in May from Rahman et al. (2015)**

As all the conducted studies emphasized, urban vegetation and in particular urban trees are a key element of cities. Urban trees commonly grow in street canyons, city parks, in front of buildings, at parking sites and at public squares. They are also present at sites like industrial areas, railroad facilities and cemeteries. Large cities like Berlin count 440,000 street trees (Stadt Berlin, 2016), while the city of Barcelona includes 198,775 planted street trees in its municipal area (Chaparro & Terradas, 2009). McPherson et al. (1997) rated the planting and maintaining

of urban trees in Chicago as costing up to $21 million, though their total value due to energy savings, air pollution mitigation and other benefits is $59 million, which amounts to a net present value of $402 per urban tree planted.

Despite their undeniable value for city aesthetics and climate, urban trees face many challenges at their growing site. Compared to forest trees, urban trees have less rooting space (Day et al., 1995; Grabosky & Bassuk, 1995) as well as less water and nutrient availability (Beatty & Heckman, 1981; Whitlow & Bassuk, 1986); the soil is very dense and compact (Beatty & Heckman, 1981; Day et al., 1995); the temperatures are higher with more reflected radiation due to buildings and asphalted surfaces (Akbari et al., 2001; Kjelgren & Clark, 1992). Further, urban trees suffer from mechanical injuries (Beatty & Heckman, 1981; Foster & Blaine, 1978), de-icing salting in winter (Petersen & Eckstein, 1988), pollutants and dog urine (Whitlow & Bassuk, 1988). Figure 6 illustrates a typical urban setting and the harsh environment of urban trees with small planting pots surrounded by compacted, paved surfaces and high levels of pollution. Tall buildings create shade which decreases the photosynthetic activity of the trees though increasing the reflected radiation. Nevertheless, urban trees also create habitats for birds, insects and other animals as well as a comfortable climate for humans.
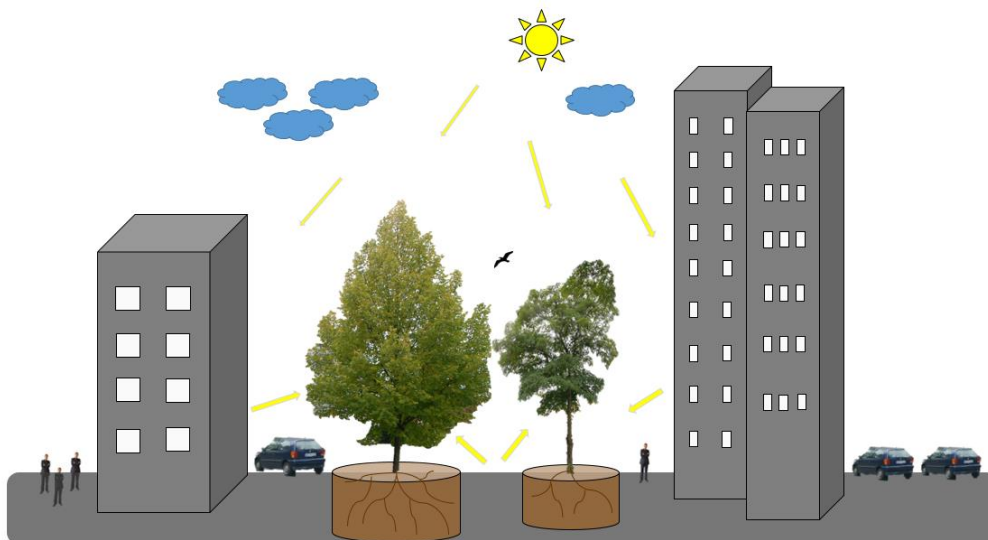


**Figure 6: Challenges and services of trees in an urban environment (Moser et al. under review)**

As demonstrated above, there are several studies worldwide about the value of urban trees and the ecosystem services they provide. However, those studies only focus on one city or few urban tree species typical for certain climatic region. Most existing data on the quantity of the benefits of urban vegetation was provided by i-Trees, which is available for US, UK and Australian cities and urban forest stands. There is a lack of data for Central Europe and typical urban tree species growing in temperate climate regions (Rust, 2014; Semenzato et al., 2011).

Peper et al. (2001b) and Peper et al. (2001a) state that growth data and patterns of a species growing in one region cannot be transferred to the same species in different environmental conditions due to tree maintenance practices and length of the growing season. Moreover, the data available through i-Trees is based on estimations of inventory data and upscaling, dealing with whole city stands and not with single urban trees growing in street canyons, at public plazas, parking areas or parks in urban areas. Accurate assessments and calculations of the actual benefits based on single urban trees are lacking (Peper et al., 2014). While there are many studies on the benefits of urban forest stands and their provided benefits, there are hardly any data on the impact of single trees depending on the species and the individual growing site of a tree. Studies like the analysis of Kjelgren & Clark (1992) and Pretzsch et al. (2015) on the growth of urban trees under different site conditions present in urban areas (parks, streets, public squares, residential areas) are scarcely existing for Europe, different urban tree species or in respect to the effects of climate change on tree growth. However, these different microclimates affect urban tree growth markedly (Kjelgren & Clark, 1992; Peper et al., 2001a, b). Not only do trees affect their environment by providing many functions and services for the city climate, they are affected by their environment and microclimate as well. Trees growing in parks experience environments similar to natural forest stands with high level of competition and water availability, while street trees need to deal with high temperatures, less water availability and often small planting pits. Those different growing sites with various microclimates in terms of air and soil temperature, water and nutrient availability, competition, radiation, sunlight exposure and rooting space influence tree growth extremely. Trees with poor growth conditions may show reduced growth and therefore provide less benefits for the city climate (Iakovoglou et al., 2001; Whitlow & Bassuk, 1986). Ultimately, poor growth conditions can result in tree die-off, leading to increased costs for maintaining and new plantings of trees (Nowak et al., 2004). The growth of trees under different site conditions is depending strongly on the tree species' characteristics and growth patterns (Rais et al., 2014). Some species are more tolerant to drought up to extreme drought-tolerant, enduring long phases of low precipitation rates. Other species are sensitive to long frost periods or late-frosts, while many species cannot deal with the increased nitrogen supply through dog urine or de-icing salting in winter (Czerniawska-Kusza et al., 2014; Mertens et al., 2007). Typical Central European species like horse chestnut *Aesculus hippocastanum* suffer tremendously from insect infestations through alien insect species and fungi (Tubby & Webber, 2010). Moreover, how much a certain tree species or urban forest stand in a city can benefit the climate and humans has been researched in many studies (Bolund & Hunhammar, 1999; Chaparro & Terradas, 2009; Ennos et al., 2014; Escobedo et al., 2011; Gill

et al., 2007; Nowak & Dwyer, 2007), but the demands and growing requirements of trees for optimal growth and provision of highest ecosystem services have been ignored. However, for every planting site the right tree species with appropriate structural and aesthetic demands must be found since each tree species has unique characteristics and demands at the planting site. Comparative studies analyzing the effects of different site conditions on the growth of typical urban tree species and their ecosystem services influenced by their growing site are rare.

To fill the above mentioned knowledge gaps, allometric equations are necessary, which predict the growth and ecosystem services of common urban tree species of all climate regions. Allometric relationships describing the growth of trees over time are common tools in forestry, but seldom available for urban trees although tree variables like crown size are closely related with the services of trees. For example, the leaf area and crown projection area CPA are directly related to the trees' physiological processes like transpiration, photosynthesis, respiration and shading (Kramer & Kozlowski, 1979; Pretzsch, 2014; Stoffberg et al., 2008; Troxel et al., 2013). Figure 7 displays the results of a study conducted by Peper et al. (2014), generating allometric relationships for two common urban tree species in the USA. Based on these allometric relationships, the ecosystem services and benefits of urban trees can be estimated for different age classes and site conditions. Relationships on the structural dimensions over time can help arborists and urban planners to plant the right tree species suitable for a certain site, providing benefits like shade, wind protection or recreation (McPherson et al., 2000).
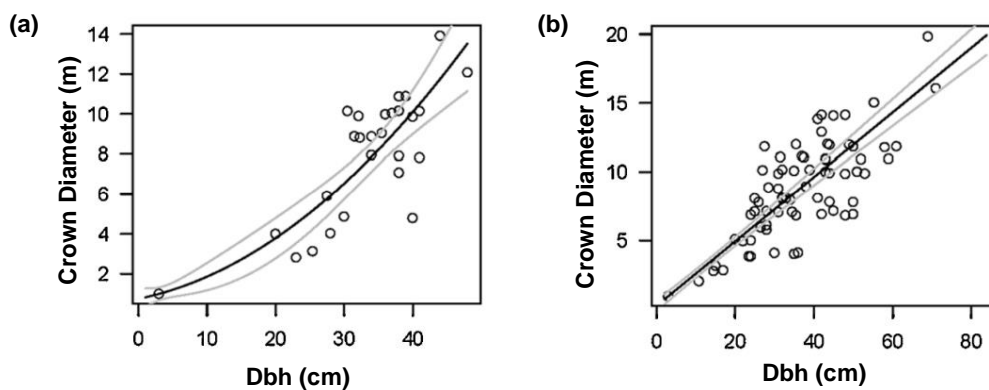


**Figure 7: Allometric equations of crown diameter based on stem diameter of (a) *Fraxinus americana* and (b) *Fraxinus pennsylvanica* of Peper et al. (2014)**

Due to climate change, the effects of the urban heat island effect UHI will be amplified, increasing the heat stress for humans living in cities. For climate-adapted cities, vegetation in cities can play a key role to mitigate the effects of climate change and UHI. Unfortunately, how urban tree growth and further the provided benefits will change under climate change has hardly been researched. There is a lack of knowledge about how climate change and changed growth conditions will affect the growth patterns of common urban tree species and their benefits and

II. Thesis: Introduction

services. Model-based simulations predicting the ecosystem services, thermal comfort and further growth of urban trees are urgently needed tools to mitigate the effects of climate change in cities (Pataki et al., 2011; Wang et al., 2008). Furthermore, dendrochronological studies on past urban tree growth can provide valuable information on the performance of certain tree species under drought stress (Gillner et al., 2014; He et al., 2007), helping scientists and urban planners to select long-living, healthy species less sensitive to drought and heat waves.

With planting climate change adapted species and proper green space design and management, the energy-use of buildings and the release of atmospheric carbon dioxide can be reduced, the air quality improved, rainfall run-off and noise levels can be lowered (Nowak & Dwyer, 2007). Inappropriate tree selection of species unadapted to the impacts of climate change might increase costs to counter pollen production and chemical emissions from trees (BVOCs), to replace dead trees and might even increase building energy-use, waste disposal and water consumption (Gregg et al., 2003; Nowak & Dwyer, 2007; Tyrväinen et al., 2005). Proper planning and planting of the right tree at the right site will ensure well-adapted, vital urban trees providing the greatest benefits for a good city climate, mitigating the effects of climate change and the urban heat island effect on humans.

## 2.    Aims of the Thesis and Research Questions

The aim of this thesis was to analyze the growth patterns of two common Central European urban tree species in relation to their growing site and tree age based on allometric relationships, increment core data and high-resolution growth data. Moreover, the ecosystem services carbon storage, cooling by evapotranspiration and shading were assessed in respect to species and growing site in terms of the present meteorological microclimate and local urban climate. The following research questions were posed and investigated in detail:

- How do the growth patterns of small-leaved lime *Tilia cordata* Mill. and black locust *Robinia pseudoacacia* L. differ in relation to their growing site, their age and their species characteristics?
- What influence does the local urban climate of the city have on the yearly growth of both species, in particular in drought years?
- Which meteorological variable has the most influence on the growth of *Tilia cordata* at two contrasting sites in respect of the drought year 2015 and the overall growth?
- How do the provided ecosystem services carbon storage, shading and cooling of both species differ in terms of species, age, micro- and local climate?


Finally, information sheets were developed, providing data on the growth and structural development as well as the delivered ecosystem services and value of both analyzed species at a certain age.

## 3. Material and Methods

### 3.1 Site Selection

The study sites of this thesis were the cities Munich and Würzburg, both located in South Germany. With around 1.5 million inhabitants, Munich is the third largest city of Germany and largest city of Bavaria. It is characterized by a cool temperate climate, located in the Alpine foothills on the Munich gravel plain. The size of Munich is 31,071 ha, 48°09'N, 11°35'E, 519 m a.s.l. with the river Isar flowing through the city center. Würzburg has approximately 125,000 inhabitants (6th largest city of Bavaria) and a size of 8,760 ha (49°48'N, 9°56'E, 177 m a.s.l.). Located in the northern region of Bavaria in the Main triangle, Würzburg is characterized by a warm, whine-growing area climate with the river Main flowing through the city.

The long-term annual precipitation means of München and Würzburg are 1009 mm and 577 mm, whereas the mean annual temperature of Munich is 9.2°C and of Würzburg is 9.4°C, respectively (DWD, 2016). Figure 8 displays the climate graphs of Munich (a) and Würzburg (b) with the monthly temperature and precipitation values from 1961 to 1990 for Munich and from 1971 to 2000 for Würzburg. While the average temperature is similar in both cities, the summer month of Würzburg are considerably warmer compared to Munich. The precipitation distribution shows similar patterns in both cities with a peak in summer and less precipitation in winter. However, the annual precipitation of Munich is on average 432 mm higher compared to Würzburg.
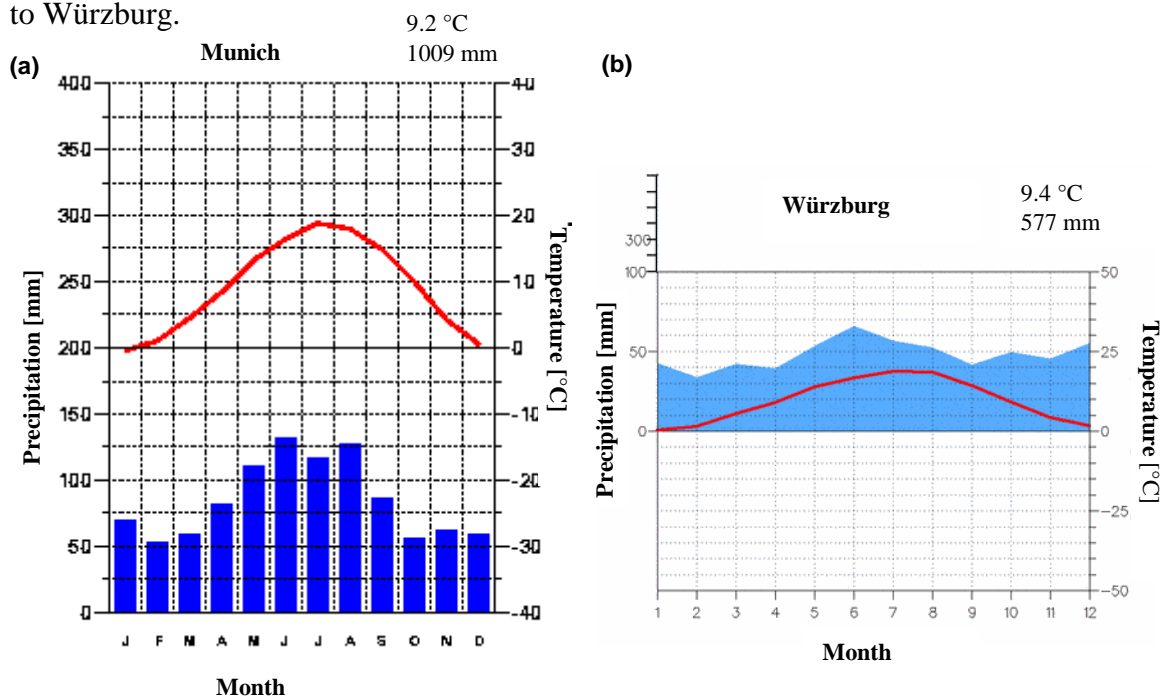


**Figure 8: Climate graphs of Munich from 1961-1990 (a) and Würzburg from 1971-2000 (b) with the monthly temperature (red line) and monthly precipitation (blue line), Source: DWD (2016)**

Moreover, at two public squares in Munich the growth patterns and services of ten *T. cordata* trees were investigated in detail (*'Green Lab Munich'*). The relationships between growth, transpiration and cooling ability of the analyzed trees and the site conditions and the microclimate were studied. Two squares were chosen for data collection, situated within the city center of Munich (Figure 9) therefore representing typical urban conditions. While Bordeaux Platz is an open green square with less pavement (OGS), Pariser Platz is a highly compacted, heavily paved square (CPS). The following criteria were fulfilled for site selection: (1) The two sites must have contrasting characteristics in terms of micrometeorology and surface cover (2) commonly planted and healthy matured street trees of the same species in a sufficient number and (3) in close distance to the city center with significant urban heat island (UHI) effect (Pongracz et al., 2010). Both squares are situated close together (distance of 180 m), with OGS being far bigger than CPS (17,250 m² to 1,945 m²) (Table 2). Pariser Platz is a roundabout connected by six roads with heavy traffic and 10 *T. cordata* trees planted, while Bordeaux Platz is an oval shaped avenue with two roads going in opposite directions with less traffic and 84 *T. cordata* trees in total. The average size of the uncovered tree pit is 55 m² at Bordeaux Platz and 5.8 m² at Pariser Platz. The planting pits of Bordeaux Platz are avenue plantations surrounded by lawn while the planting pits at Pariser Platz are small cut-out pits.

**Table 2: Site characteristics of Bordeaux-Platz and Pariser Platz in Munich with OPS = open, green square with less pavement and CPS = completely paved square**

| square | altitude | longitude | latitude | no. of trees | tree pit size | square shape | square size | square type |
|---|---|---|---|---|---|---|---|---|
| Bordeaux Platz | 530 m | 11.601348 | 48.129749 | 84 | 55 m² | oval | 17,250 m² | OPS |
| Pariser Platz | 531 m | 11.600133 | 48.128384 | 10 | 5.8 m² | roundabout | 1,945 m² | CPS |

*3.2   Species Selection*

For this study the two urban tree species small leaved lime *Tilia cordata* and black locust *Robinia pseudoacacia* were chosen. Species selection was based on abundance and ecological features: Both species are two common tree species in Central European cities (Pauleit et al., 2002) and differ regarding their ecological features. While *T. cordata* is a shade-tolerant and moderately drought tolerant species (Radoglou et al., 2009; Roloff et al., n.y.), *R. pseudoacacia* needs light for growing (shade-intolerant) and is very tolerant to drought (Keresztesi, 1988; Roloff et al., n.y.). Moreover, *T. cordata* can tolerate alkaline site conditions and still accumulate nutrients (Trowbridge & Bassuk, 2004). In contrast, *R. pseudoacacia* is a species with mycorrhiza partnerships and therefore very tolerant towards salt damages (Meyer, 1982).

## 3.3 Data Collection

Tree selection for data collection was mainly based on visual impression; damaged, pruned, or low-forking trees were excluded from measuring. Tree data was collected from November 2013 to September 2014. Data collection for the detailed study on the growth patterns of *T. cordata* at CPS and OGS was done in summer 2015 (for more details see *3.8 Green Lab Munich*). All the measured trees were classified as either park trees, trees in public places, or street trees. Trees were considered park trees when planted in a green area without buildings, while street trees were trees planted in a street canyon. In public places−smaller, mostly paved spaces freely accessible to the public−only free-standing trees with open, detached crowns were selected. Figures 9 respectively 10 display detailed maps of the cities Munich and Würzburg with all measured *T. cordata* and *R. pseudoacacia* trees distributed along the whole city areas.
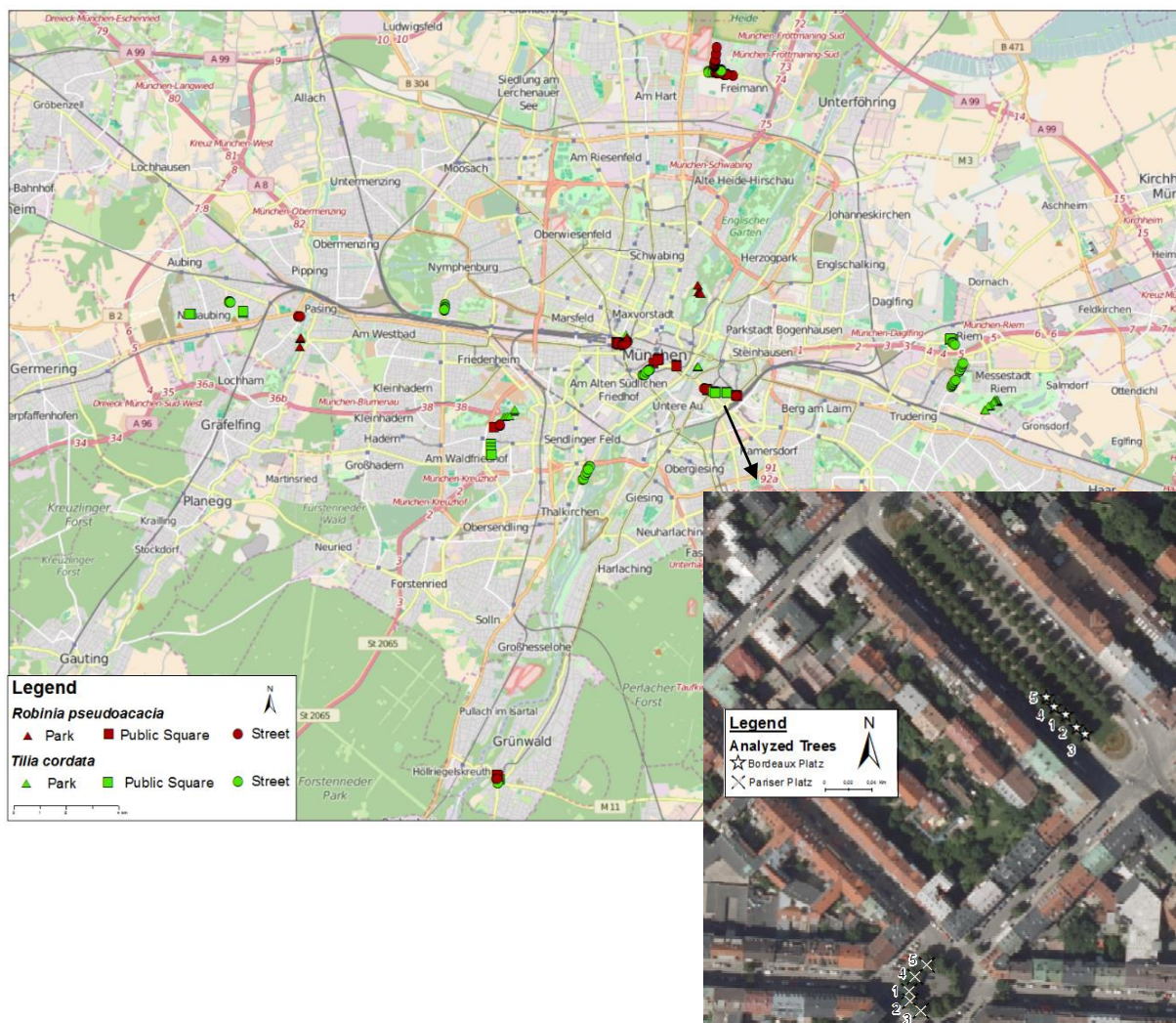


**Figure 9: Measured trees in Munich and the ten *T. cordata* trees at two public squares Bordeaux Platz and Pariser Platz (Source background map: Bayerische Vermessungsverwaltung 2015)**
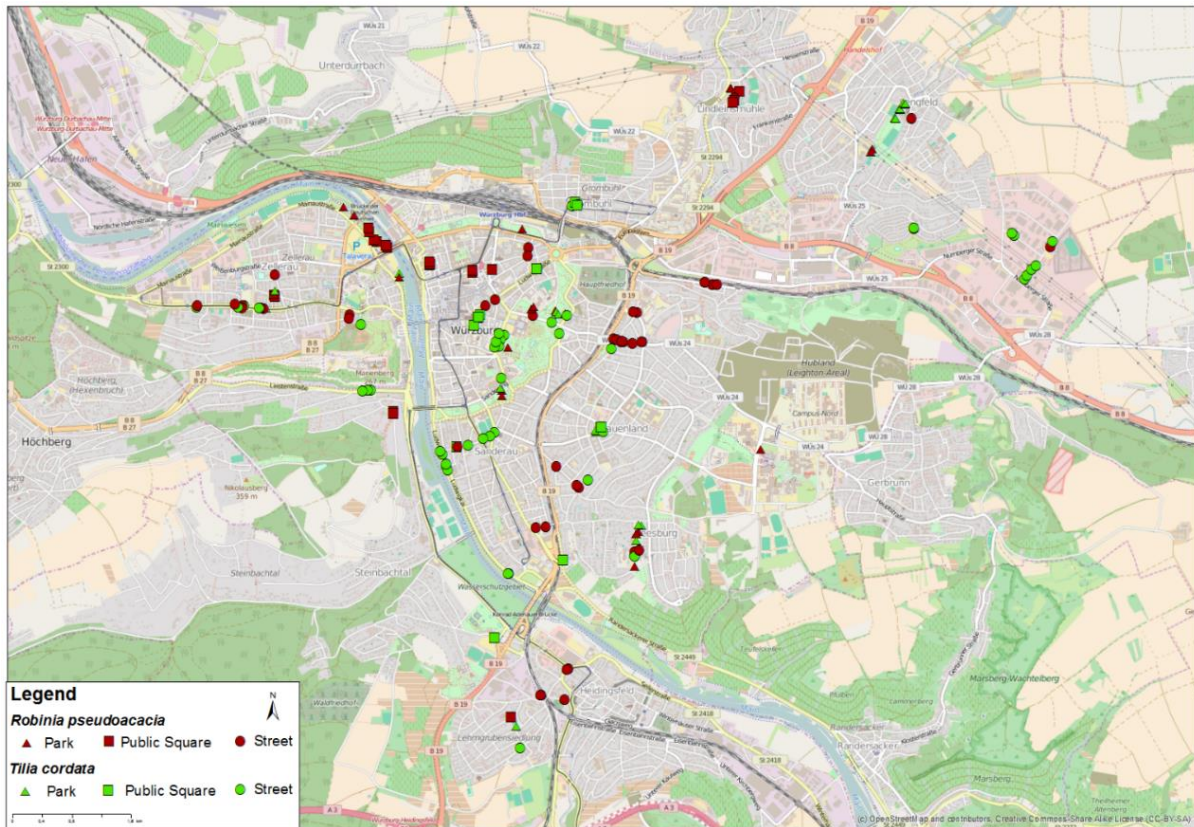
**Figure 10: Measured trees in Würzburg (Source background map: Bayerische Vermessungsverwaltung 2015)**

Overall, 175 trees were measured in Würzburg and 245 trees in Munich, of those 225 *T. cordata* and 195 *R. pseudoacacia* trees, adding up to 420 trees in sum for both cities and species (Table 3).

**Table 3: Number of measured *Tilia cordata* and *Robinia pseudoacacia* trees in Munich and Würzburg distributed in three categories (park, public square, street)**

|  | *T. cordata* | *R. pseudoacacia* | **Sum** |
|---|---|---|---|
| Munich | 139 | 106 | 245 |
| *Park* | 38 | 19 | 57 |
| *Public Square* | 45 | 33 | 78 |
| *Street* | 56 | 54 | 110 |
| Würzburg | 86 | 89 | 175 |
| *Park* | 23 | 26 | 49 |
| *Public Square* | 24 | 30 | 54 |
| *Street* | 39 | 33 | 72 |
| **Sum** | 225 | 195 | 420 |

For each measured tree, the following information was recorded: diameter at breast height (dbh), tree height (h), height to live crown base ($h_c$), crown diameter, tree pit (open surface area OSA), vitality, coordinates and altitude, and distance to adjacent buildings and trees. Crown radii and tree pit were measured in eight intercardinal directions (N, NE, . . ., NW) along the ground surface from the center of the trunk to the tip of the most remote downward-projecting

shoot and to the last visible open, non-asphalted surface of the soil. The vitality of the trees was rated based on the scale of Roloff (2001), ranging from very good (0) to very poor (3) conditions regarding the branching structures of the crown.

### 3.4   Data Derived by Calculations from Measured Data

From the measured data, crown radius (cr), crown diameter, crown volume (cv), crown projection area (CPA), age and biomass were calculated. The mean crown radius was defined as the quadratic mean:

$$cr = \sqrt{(r_N^2 + r_{NE}^2 + \ldots + r_{NW}^2)/8} \tag{1}$$

with $r_N$ as the widest measured crown extension in the northern direction,…, $r_{NW}$ the widest crown extension in the northwest direction.

The crown projection area and the crown volume were calculated as

$$CPA = cr^2 * \pi \tag{2}$$

$$cv = CPA \times crown\ height \tag{3}$$

The aboveground woody biomass of *T. cordata* was obtained by the equation of Čihák et al. (2014), while the biomass of *R. pseudoacacia* was computed following Clark III & Schroeder (1986).

Leaf area index (LAI) was derived from hemispherical photographs captured during the fully leafed phase (July–August) using a Nikon CoolpixP5100 camera with fisheye lens and Mid-OMount. Between the time of measurement and the fully leafed phase, some trees were removed or severely pruned, and these trees were excluded from photography. The resulting hemispherical photos were analyzed with the programs WinSCANOPY (Régent Instruments Inc.). This software allows the calculation of the LAI by several methods. For *R. pseudoacacia*, LAI (Bonhom)-Lin was used, which is based on the Bonhomme & Chartier (1972) linear method. For *T. cordata*, the LAI (2000)-Lin calculated with the Licor LAI2000 linear method (Miller, 1967; Welles & Norman, 1991) and the LAI-2000 generalized method based on the work of Miller (1967) and Welles & Norman (1991) resulted in the most reliable values. Regression analyses with generalized additive models (GAM) and mixed smoothed predictor variables were performed to obtain specific models for calculating LAI on the basis of the measured tree dimensions. The best fitting model with the lowest Akaike Information Criterion (AIC) was chosen, for *T. cordata* resulting in:

$$\ln(\text{LAI}) = a + s(b_1) \times \ln(\text{dbh}) + s(b_2) \times \ln(h) + s(b_3) \times \ln(2 \times cr) \tag{4}$$

and for *R. pseudoacacia* resulting in:

$$\ln(\text{LAI}) = a + s(b_1) \times \ln(h) + s(b_2) \times \ln(cv) + s(b_3) \times \ln(\text{CPA}) \tag{5}$$

with $a$ as intercept, s as smooth function for every slope variable ($b$) and h as tree height.

The age of all trees was estimated based on the measured tree parameters dbh and tree height. For *T. cordata*, the formula of Lukaszkiewicz & Kosmala (2008) was applied:

$$\text{age} = a + \exp^{(b + c \times \text{dbh}/100 + d \times h)} \tag{6}$$

with $a = 264.073$, $b = 5.5834$, $c = 0.3397$, $d = 0.0026$, dbh in cm, and h in m.

To obtain the age of *R. pseudoacacia*, dbh was multiplied with a species-dependent age factor of 0.996, which was computed by the measurements of Dwyer (2009) for *Gleditsia triacanthos*:

$$\text{Age} = \text{dbh} \times 0.996 \tag{7}$$

*3.5   Allometry*

Allometry in forestry refers to the analysis and modeling of relationships of measured tree dimensions with linear regression techniques (Pretzsch et al., 2012). In this study, the relationships of tree height and crown dimensions were analyzed with dbh and age as predictor variable, computing the development of *T. cordata* and *R. pseudoacacia* over time. The modeling of growth is usually based on dbh, due to the pipe model theory of Shinozaki et al. (1964), which predicts leaf area from dbh (Peper et al., 2001a). On the basis of this theory, most modeling of forest stands was conducted, using dbh as an explanatory variable for predicting tree height and crown dimensions (Peper et al., 2001a; Pretzsch et al., 2012; Stage, 1973; Watt & Kirschbaum, 2011).

$$\ln(y) = a + b \times \ln(x) \tag{8}$$

with $y$ as the outcome variable, $x$ as predictor variable, $a$ as intercept and $b$ as slope.

With the performed OLS-regression (ordinary least square), the outcome ($y$) is calculated from the predictor ($x$). Since in application of the models, the growth of trees over time (outcome $y$) will be estimated on basis of the predictor ($x$), OLS-regression was chosen instead of RMA (reduced major axis)- and MA (moving average)-regression (Niklas, 1994). Other models like polynomial regressions were tried but showed no significant improvement; therefore, the log transformation was conducted for all analysis. This is compliant with previous research of forest stands and urban trees (Peper et al., 2014; Peper et al., 2001a, b; Pretzsch et al., 2012; Stoffberg

et al., 2008). For the back-transformation of the logarithm, the bias correction (CF) by Baskerville (1971) and Sprugel (1983) was applied:

$$CF = \exp^{(RSE^2/2)} \tag{9}$$

with RSE = residual standard error.

### 3.6 Increment Core Collection and Dendrochronology

Increment cores were taken from 88 *T. cordata* and 62 *R. pseudoacacia* trees in Munich and Würzburg, including 20 *T. cordata* trees at OGS and CPS during the study Green Lab Munich. For each tree, two perpendicular cores of the stem were extracted at a height of 1.3 m, in northern and eastern direction. Due to the main wind direction (southwest), coring from north and east yields more representative tree ring width (Pretzsch et al., 2013). For further processing, all cores were mounted on grooved boards with glue and sanded using progressively finer sand papers (Figure 11 (a)). The first sanding was applied to flatten the cores, whereas the subsequent sanding episodes polished the cores for better visualization of the cross-sectional area (Speer, 2012). The annual tree-ring widths of the cores were measured using a Lintab digital positioning table with a resolution of 1/100 mm (Figure 11 (b))(Rinn, 2005). For cross-dating of the time-series, the software package TSAP-Win (Rinn Tech, 2010) was used.
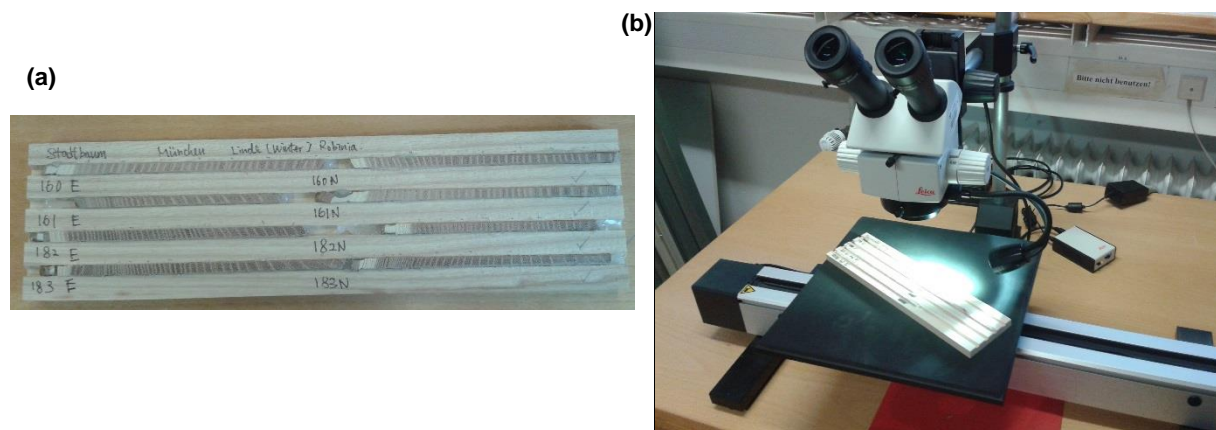


**Figure 11: (a) Increment cores mounted with glue and sanded to increase visibility of year rings and (b) Lintab digital positioning table**

All following analyses of increment core data were carried out in R (R Core Team, 2014) using package dplR (Bunn et al., 2015). With dplR, all tree-ring series were indexed using a double detrending process: first, modified negative exponential curves were applied followed by cubic smoothing splines (20 years rigidity, 50% wavelength cutoff). The detrending was conducted to remove low frequency trends, which are age associated (Gillner et al., 2014; Holmes et al., 1986). The resulting detrended series were averaged using Tukey's biweight robust mean to

build chronologies for both species and each city. The autocorrelation of every series was removed using autoregressive models (maximum order of 3). As a result of detrending, standardized chronologies with a yearly ring width index (RWI) averaging around 1 were obtained. Values smaller than 1 indicate growth below normal, while values greater than 1 indicate growth higher than normal. The statistical validity of the chronologies was assessed using the expressed population signal (EPS) for the common period of the time series of all analyzed tree individuals.

With the derived increment cores of OGS and CPS in Munich, a pointer year analysis was conducted; it computes the individual relative radial growth variation by comparing the ring-width of year t to that of year $t^{-1}$ for each year ring series. The sign and magnitude of the found variation between the series were derived (Bunn et al., 2015; Mérian & Lebourgeois, 2011). Positive pointer years display an extraordinary wide ring-width, while negative pointer years are characterized by exceptionally narrow rings (Gillner et al., 2014).

### 3.7   Climate-growth Relationships

To derive past drought years of Munich and Würzburg, a monthly drought index, namely the SPEI, was calculated to investigate the effects of climate on tree growth (Vicente-Serrano et al., 2010). The SPEI uses precipitation and potential evapotranspiration ($PE_T$) as input data (Vicente-Serrano et al., 2010), whereby $PE_T$ was calculated according to the Hargreaves approach (Droogers & Allen, 2002; Hargreaves, 1994) including minimum and maximum temperature and the Penman-Monteith equation (Allen et al., 1998) including minimum and maximum temperature, wind speed and sunshine hours. Using the calculated SPEI, the influence of the identified drought years on the growth patterns of urban trees was analyzed. A yearly SPEI for 1985–2014 (increment core study of Munich and Würzburg) respectively 1985–2015 (Green Lab Munich) was computed by averaging the monthly index values of the growing season from April to September. Years with an SPEI smaller than −1 were classified as moderate drought years, whereas years with an SPEI smaller than −2 were interpreted as years with severe droughts (Lough & Fritts, 1987; Vicente-Serrano et al., 2010).

The influence of drought years on tree growth was analyzed with a superposed epoch analysis (SEA). SEA assesses the significance of events like droughts on tree growth during drought and to the pre- and post-drought growth periods (Bunn et al., 2015). Following Lough & Fritts (1987), Orwig & Abrams (1997), and Gillner et al. (2014), deviations from the mean ring width index of each core were calculated for the following three periods. The growth of the 5 years prior to the analyzed drought year (pre-drought), of the drought year and of the 5 years after the

drought year (post-drought) were averaged to detect significant departures between those superposed epochs. Further, the influence of past temperature and precipitation on the year ring formation of a certain year, a response function analysis with bootstrap resampling on 1000 samples was conducted (Fritts, 1971; Zang, 2011).

### 3.8 Green Lab Munich: Growth Patterns of T. cordata in Relation to the Site Conditions

To investigate the growth patterns in relation to the site conditions in detail, electronical dendrometer generating high-resolution growth data and a weather station for collecting meteorological data were installed at both sites (Figure 12). For continuous measurements of stem growth precision dendrometer (Ecomatic, Dachau, Germany) were attached at a height of about 4 m at five trees of each site. The dendrometer measures the linear shifting of a small sensing rod fixed at the bark of the trunk. With stem contraction and expansion, the rod is displaced as well, translating the movement in an electrical signal (Drew & Downes, 2009; Mäkinen et al., 2008). June 23rd to October 21st 2015 and stored in a CR800 data logger (Campbell Scientific, U.K.) equipped with Campbell Logger Multiplexer (AM16/32B). The height was chosen to ensure inaccessibility of installed devices to avoid vandalism.

Air temperature, air pressure, relative air humidity, precipitation, wind speed and direction were measured with two Vaisala Weather Transmitters WXT520 (EcoTech Umwelt-Meßsysteme GmbH, Bonn, Germany) at OGS and CPS. The stations were installed on top of street lamp posts at a height of 3.3 m by cross arms and 2 m away from the lamp to decrease influence of lamp and shade of the nearby trees and buildings (Rahman et al. accepted). At OGS the weather station was in a straight line close to the first tree and at CPS right at the center of the square to capture differences in micrometeorology as accurately as possible. All measured data were recorded continuously in a 15-min resolution from July 28th to October 21st 2015 on a enviLog remote data logger (EcoTech Umwelt-Meßsysteme GmbH, Bonn, Germany) attached to one of the sampled trees at each site (Rahman et al. accepted).

Soil matric potential and temperature were measured using Tensiomark 1 (4244/1, range pF 0-pF 7) (EcoTech Umwelt-Meßsysteme GmbH, Bonn, Germany) installed vertically through soil profile to the depth of 30 cm. At Bordeaux Platz the sensors were installed 3.5 m away from the main stem within the fringe of the CPA. Three sensors were installed at the first tree, followed by two sensors for next four. At Pariser Platz two sensors for each tree were installed at the furthest opening point from the main stem. All the sensors were installed in a place which was mostly shaded to minimize the effect of direct radiation on the soil surface (Rahman et al.

accepted). Further, sap flux sensors and crown temperature sensors were installed (Rahman et al. accepted).
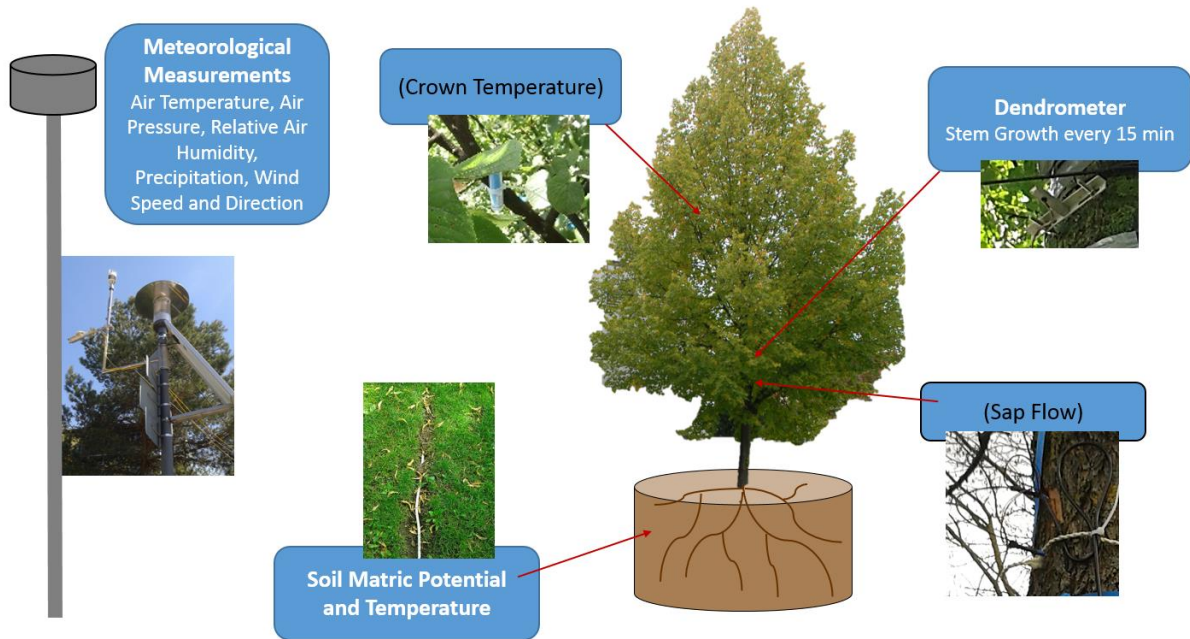


**Figure 12: Measurements at 10** *T. cordata* **trees at Bordeaux Platz OGS and Pariser Platz CPS (white fronts discussed in this thesis) at the Green Lab Munich**

### 3.9    *Ecosystem Services: Carbon Storage, Shading and Cooling*

To evaluate the benefits of urban trees for the city climate, the three ecosystem carbon storage, shading, and transpiration by cooling were analyzed in detail. To calculate the carbon storage of a tree, the biomass was multiplied with 0.5 following IPCC (2003) and Yoon et al. (2013). Shading can be obtained depending on crown height $h_c$ and crown radius cr for each hour of the day by

$$A_S = 2 \times cr \times h_c \times f \times \cot (Y) \tag{10}$$

with $A_S$ = shaded area in m$^2$, $f$ = correction factor for the crown form, $Y$ = elevation angle of the sun which in turn is a function of latitude, day of the year and hour of the day (e.g., DVWK (1996)), assuming a perpendicularly standing object on a horizontal surface (Häckel, 2012). As shading coefficient the average shaded area of all sunlight hours of June 21st was applied. For rough estimations of transpiration, and thus, the cooling effect of individual trees, simulation results of the process-based growth model BALANCE were used (Rötzer et al., 2010). This model simulates the growth and water balance of individual trees depending on their environment (Rötzer et al., 2010). Subsequently, the annual transpiration sums of the deciduous species European beech *Fagus sylvatica* were related to LAI [m$^2$/m$^2$] by regression analysis:

$$\text{tra}_a \ [\text{mm}] = -0.0168 \times \text{LAI}^2 + 0.4555 \times \text{LAI} + 0.2844 \qquad (r^2 = 0.65) \qquad\qquad (11)$$

To estimate the average annual energy withdrawn from the atmosphere for transpiration, the transpiration sum must be multiplied by the potential evaporation energy [2.34 MJ (kg $H_2O$)$^{-1}$].

*3.10 Laserscanning*

Of the measured 425 *R. pseudoacacia* and *T. cordata* trees in Munich and Würzburg, 129 trees were analyzed in detail with the Riegl LMS-Z420i TLS system. This laser scanner fires laser impulses at the target and the sent light is reflected back to the scanner partly. The time between firing and returning of the signal is measured by the scanner, giving the distance of the scanned object. The accuracy for measured distance is about 1 cm, the angles azimuth and inclination to the target are recorded with a precision of 0.002°. The laser scanner can scan in maximum 360° in horizontal direction and 80° in vertical direction. By tilting the scanner of 90° and doing a second scan, a high-resolution scan of the crown can be achieved. The originally spherical coordinates (azimuth, inclination and distance) were then translated to Cartesian coordinates (x, y, z) for further processing, which was conducted with the software RiScanPro (Riegl, Version 1.4.3). Data analysis was done in a Master thesis by Claudia Chreptun (Chreptun, 2015).

## 4.    Results

### 4.1    Structure and Growth of R. pseudoacacia and T. cordata

Most of measured *R. pseudoacacia* trees had greater values regarding the different tree dimen-sions (except LAI) than *T. cordata* on average (Figure 13). When comparing the values for the different dimensions of both species for the two cities, *R. pseudoacacia* had greater tree struc-tures in Würzburg compared to Munich, while *T. cordata* had on average more similar tree structures in both cities (see Moser et al. (2015)).
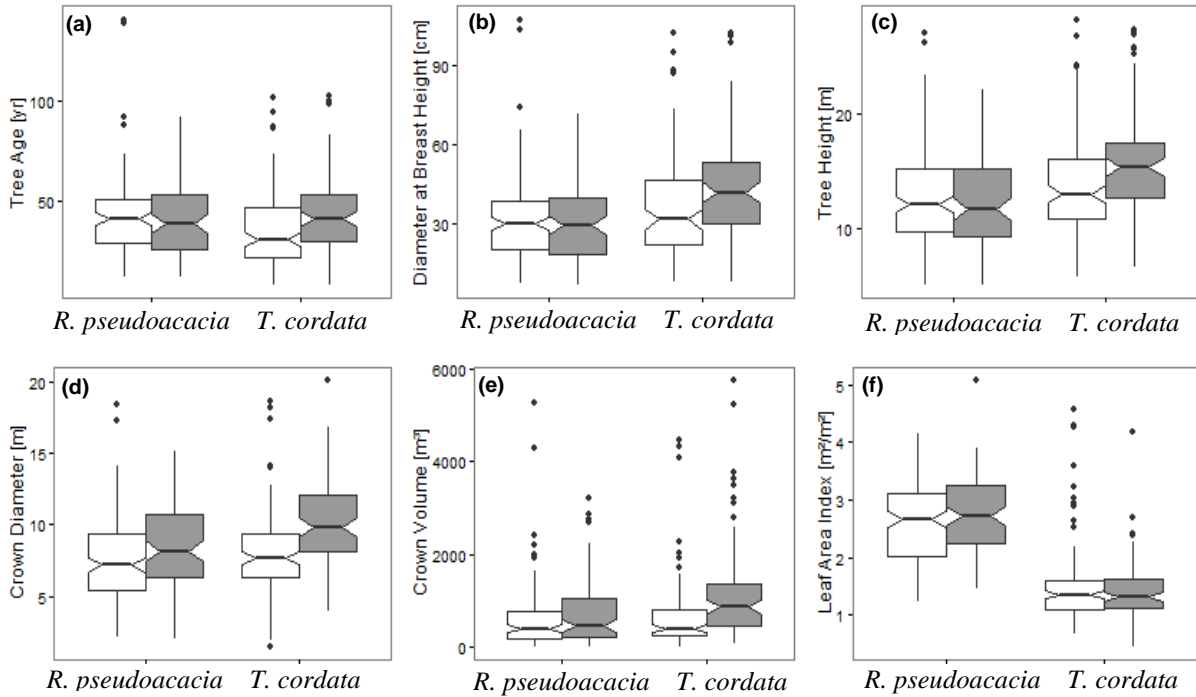


**Figure 13: Structural dimensions (a) age, (b) dbh, (c) tree height, (d) crown dimeter, (e) crown volume and (f) LAI of *R. pseudoacacia* and *T. cordata* in Munich (white) and Würzburg (grey)**

When comparing tree dimensions of *R. pseudoacacia* and *T. cordata* grown at different site categories park, public square and street, park trees often displayed greatest values and street trees the smallest tree sizes except for the openness of the sites (Figure 14). Trees of *T. cordata*, growing in public squares, however, had the greatest dbh, tree height and crown sizes.
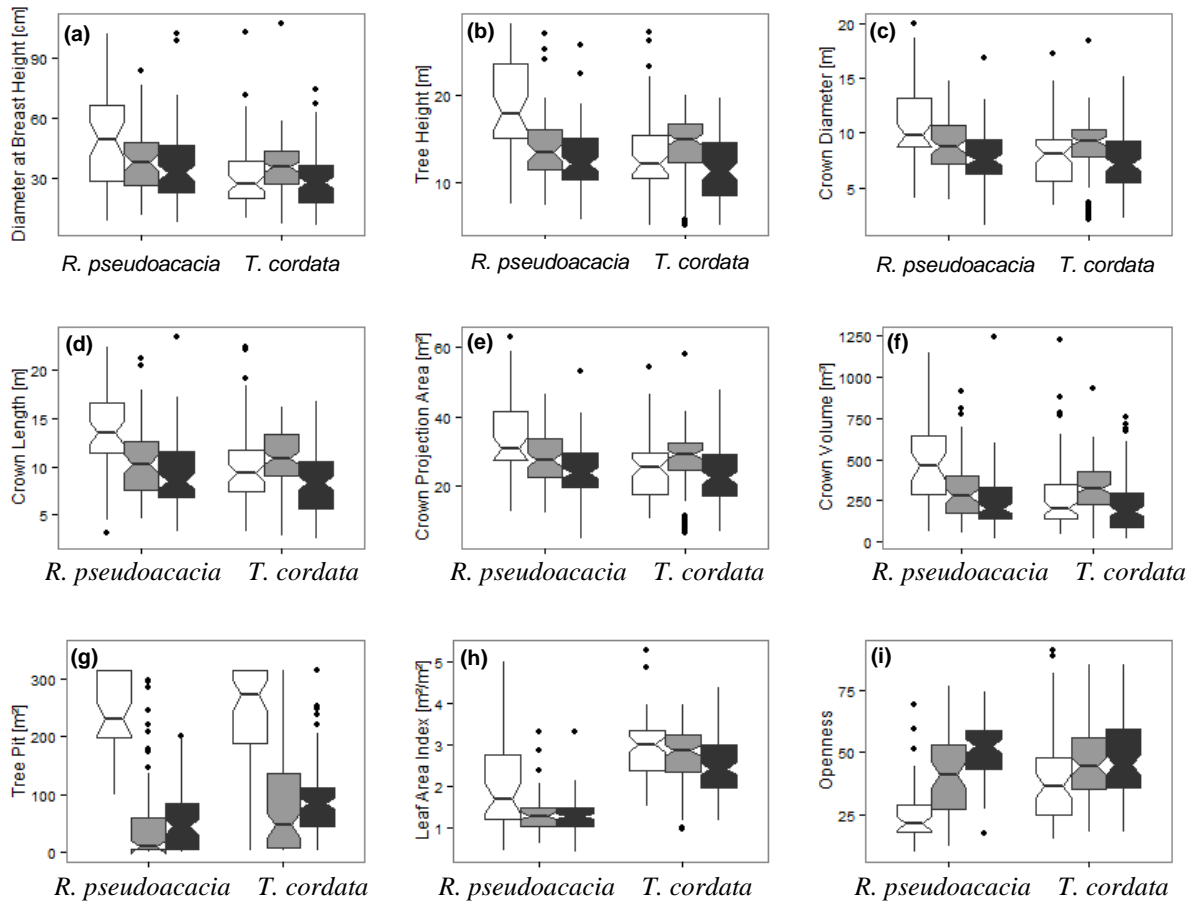
**Figure 14: Structural dimensions (a) dbh, (b) tree height, (c) crown diameter, (d) crown length, (e) crown projection area, (f) crown volume, (g) tree pit, (h) leaf area index and (i) openness of *R. pseudoacacia* and *T. cordata* in the site categories park (white), public square (grey) and street (black)**

Regression analysis revealed different growth patterns of both species for all measured and calculated tree structures (Table 4 and Figure 15). For *T. cordata*, higher coefficients of determination were found, while *R. pseudoacacia* showed a weaker relationship especially with tree height. However, *R. pseudoacacia* grew faster than *T. cordata*, as the steeper slope of the regression line indicates.

**Table 4: Results of linear logistic regressions with dbh as predictor variable and the tree variables tree height, crown base, crown height, crown diameter, CPA (cpa) and crown volume as response variables for *T. cordata* and *R. pseudoacacia* (equation: ln(response) = a + b × ln(predictor)). The table lists the regression coefficients (a, b), coefficients of determination ($r^2$), residual standard errors (RSE) for bias correction, F-values, p-values, and standard errors (SE) of regression coefficients as well as the sample size (n), published in Moser et al. (2015)**

| Species | Parameter | n | a | SE | b | SE | $r^2$ | RSE | F | p |
|---|---|---|---|---|---|---|---|---|---|---|
| *Tilia cordata* | dbh vs tree height | 225 | 0.51 | 0.10 | 0.59 | 0.03 | 0.79 | 0.16 | 863.60 | <0.001 |
| | dbh vs crown base | 225 | 0.35 | 0.09 | 0.23 | 0.03 | 0.25 | 0.22 | 72.41 | <0.001 |
| | dbh vs crown height | 225 | −0.26 | 0.09 | 0.73 | 0.03 | 0.76 | 0.22 | 724.60 | <0.001 |
| | dbh vs crown diameter | 223 | −0.61 | 0.07 | 0.78 | 0.02 | 0.87 | 0.16 | 1479.00 | <0.001 |
| | dbh vs cpa | 223 | −1.47 | 0.14 | 1.56 | 0.04 | 0.87 | 0.33 | 1485.00 | <0.001 |
| | dbh vs crown volume | 223 | −1.73 | 0.20 | 2.29 | 0.06 | 0.87 | 0.48 | 1504.00 | <0.001 |
| *Robinia pseudoacacia* | dbh vs height | 195 | 1.14 | 0.10 | 0.42 | 0.03 | 0.55 | 0.22 | 233.70 | <0.001 |
| | dbh vs crown base | 195 | 0.85 | 0.16 | 0.13 | 0.04 | 0.04 | 0.35 | 8.47 | <0.001 |
| | dbh vs crown height | 195 | 0.38 | 0.12 | 0.54 | 0.03 | 0.59 | 0.25 | 279.70 | <0.001 |
| | dbh vs crown diameter | 195 | −0.01 | 0.09 | 0.60 | 0.02 | 0.76 | 0.19 | 615.90 | <0.001 |
| | dbh vs cpa | 195 | −0.25 | 0.17 | 1.20 | 0.05 | 0.76 | 0.38 | 613.80 | <0.001 |
| | dbh vs crown volume | 195 | 0.13 | 0.23 | 1.74 | 0.06 | 0.79 | 0.50 | 743.60 | <0.001 |



**Figure 15: Allometric logistic relationships between stem diameter and (a) tree height, (b) crown diameter, and (c) crown volume for *T. cordata* and *R. pseudoacacia*. The statistical characteristics of the regression lines are presented in Table 4. Published in Moser et al. (2015)**

The leaf area of both species was differing due to different crown structures; while *T. cordata* had a greater leaf area index LAI and leaf area, *R. pseudoacacia* had a smaller LAI and leaf area. Figure 16 displays the relationships of crown volume with leaf area and LAI based on laser scanning.



$$ln(LA) = -0,09 + 0,88 \cdot ln(Vol)$$
$$ln(LA) = -1,35 + 0,99 \cdot ln(Vol)$$

$$ln(LAI) = 0,11 + 0,16 \cdot ln(Vol)$$
$$ln(LAI) = -0,88 + 0,22 \cdot ln(Vol)$$
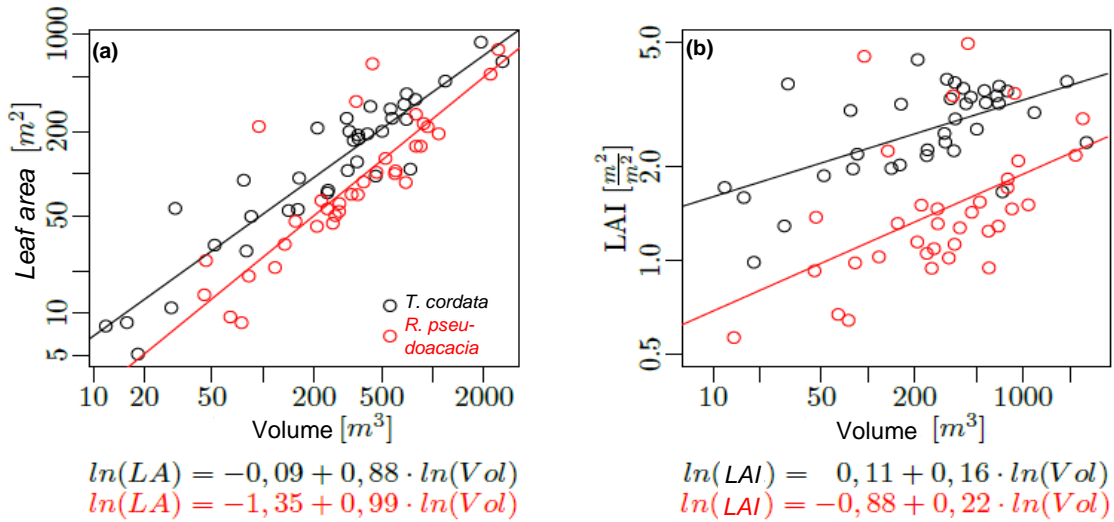
**Figure 16: Relationships of (a) leaf area (LA) with crown volume (Vol) and (b) leaf area index (LAI) with crown volume of *T. cordata* (black) and *R. pseudoacacia* (red) modified after the thesis by Chreptun (2015)**

### 4.2 Yearly Growth Patterns and Drought Resistance

A dendrochronological study regarding the growth patterns revealed a higher increment growth rate of *R pseudoacacia* compared to *T. cordata*. There were no great differences in the growth of both species regarding the city; *R. pseudoacacia* trees in Munich had an average increment growth rate of 3.9 mm yr^{-1} in Munich and 4.0 mm yr^{-1} in Würzburg, while *T. cordata* grew 3.2 mm yr^{-1} on average in Munich and 2.9 mm yr^{-1} in Würzburg (see Moser et al. (2016b)).

A further response analysis of the relationships of monthly increment growth with temperature and precipitation revealed varying impacts of the monthly meteorological values on the growth of both species. The growth of *R. pseudoacacia* in Munich in a certain year was significantly influenced by the October temperature in the previous year of the ring formation, though no other month of the previous and the current years had a significant impact on tree ring formation regarding temperature and precipitation (Figure 17 (a)). However, the temperature in the previous October positively influenced the growth of *R. pseudoacacia*, similar to all month of the current year of the tree ring formation except June. The precipitation mostly had a negative influence on growth, only the precipitation in the summer month of the current year were positive for tree ring formation.

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In Würzburg, the temperature was mostly negatively influencing the growth of *R. pseudoacacia*, only the temperature in the summer month of the current year of the tree ring formation was of positive impact (Figure 17(b)). The precipitation again had no significant influence on increment growth, however the precipitation in late winter and spring of the current year was positive for growth, while the precipitation in July and August decreased the monthly growth of *R. pseudoacacia*.
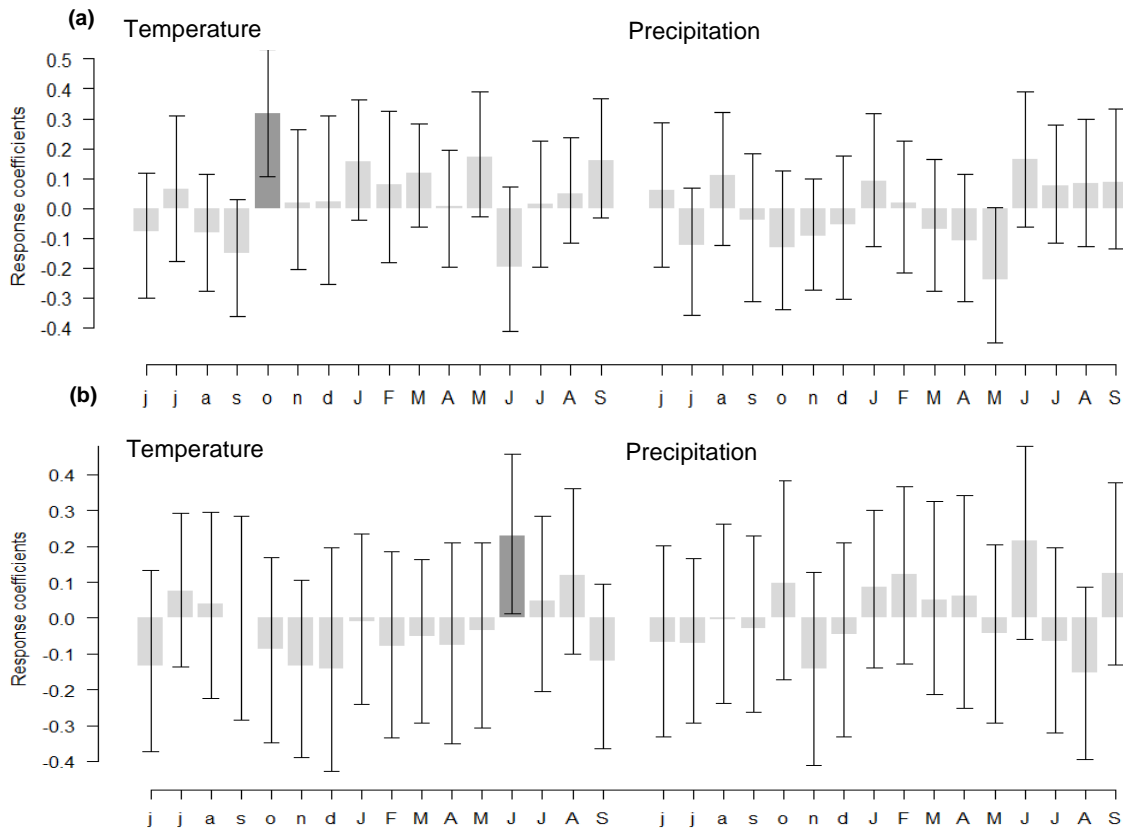


**Figure 17: Response function on the relationship of growth with temperature and precipitation in the month previous of year ring formation (small letters) and the month of the current year of year ring formation (capital letters) of *R. pseudoacacia* in (a) Munich and (b) Würzburg**

*T. cordata* displayed different relationships of growth with temperature and precipitation in Munich and Würzburg (Figure 18 (a) and (b)). While in Munich, the monthly average temperatures (except the moths June and August) in the previous year always had a negative influence on the tree ring formation (significant in June), negative effects of the monthly temperature in the current year of growth were more pronounced. In contrast to *R. pseudoacacia*, precipitation in January and April of the current year had significant negative effects on growth, while the other seasons showed no clear patterns.

In Würzburg, no significant relationships were found neither with temperature and precipitation. Further, the influences of temperature and precipitation on tree ring formation followed no clear pattern depending on year and season.
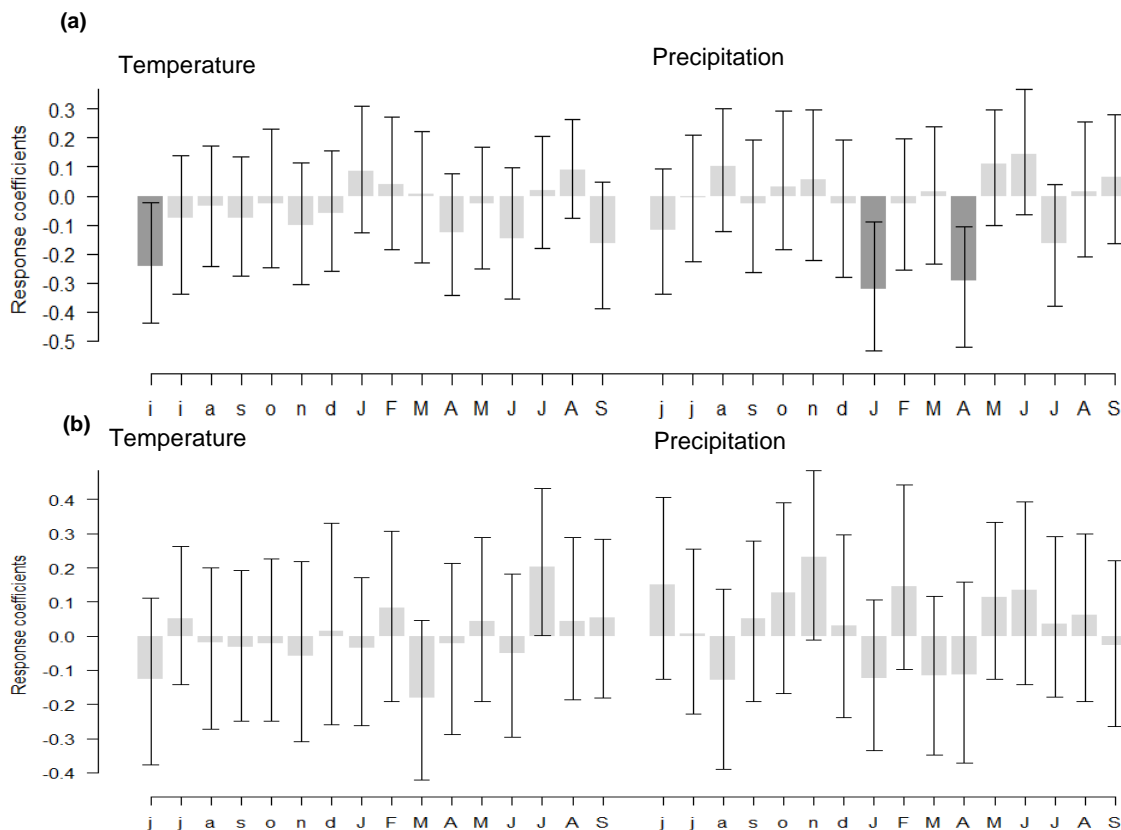


**Figure 18: Response function on the relationship of growth with temperature and precipitation in the month previous of year ring formation (small letters) and the month of the current year of year ring formation (capital letters) of *T. cordata* in Munich (a) and Würzburg (b)**

With a drought year analysis (SEA), the growth patterns of both species during and after drought were analyzed. In Munich, *R. pseudoacacia* displayed the greatest growth decrease during the drought year and a quick recovery afterwards, while *T. cordata* had the greatest growth decrease in the following year after drought events and a slower recovery. In Würzburg, however, the growth patterns were less distinct. For *R. pseudoacacia* no growth decrease was found during the drought year, albeit the growth was very small during drought events with a similar fast recovery. *T. cordata* again displayed the greatest growth decreases after the analyzed drought events (see Moser et al. (2016b)).

A generalized mixed model approach revealed further, that especially the dbh, the canopy openness, the tree pit and the water supply (representing the SPEI) had marginal influence on the past growth of *T. cordata* and *R. pseudoacacia*. During drought years, though, the water supply in terms of SPEI (precipitation) and the canopy openness proved of special importance for

growth, while the tree pit had no further importance (dbh was removed due to age trend correction). Therefore, the size of the pit did not influence the growth during drought years as much as the poor water availability of the soil. The overall water supply was of most importance for tree growth, as well as individual species characteristics and the light availability (see Moser et al. (2016b)).

### 4.3 Growth of T. cordata in Relation to the surrounding Microclimate

Depending on the growing site, different growth curves were found for *T. cordata* at Bordeaux Platz OGS and Pariser Platz CPS in Munich (Figure 19). While the trees at OGS grew rapidly in spring and started to decline slightly in their growth during summer, the trees at CPS only grew slightly in spring. Afterwards, tree growth declined until a diameter shrinkage up to 0.8 mm per tree was observed at CPS (see Moser et al. (2016a)).
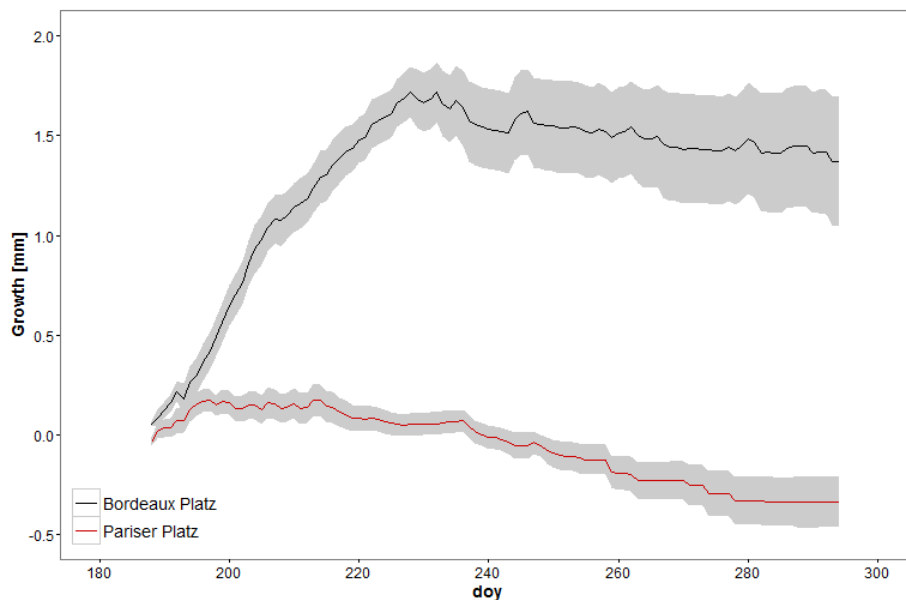


**Figure 19: Daily growth with standard error (grey) of *T. cordata* at Bordeaux Platz (black) and Pariser Platz (red) in Munich (Moser et al., 2016a)**

Further correlation analysis between the minimum growth of *T. cordata* and measured meteorological data revealed strong negative relationships of growth with air temperature, soil temperature, transpiration and vapor pressure deficit. Only precipitation had a positive influence on growth (Figure 20).
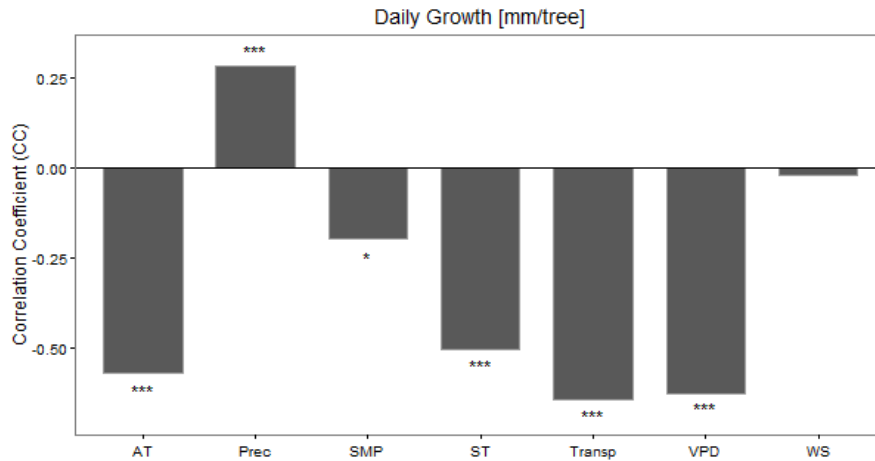
**Figure 20: Correlations of the daily minimum growth of *T. cordata* with air temperature (AT), precipitation (prec), soil matric potential (SMP), soil temperature (ST), transpiration (Transp), vapor pressure deficit (VPD) and wind speed (WS) (Moser et al., 2016a)**

## 4.4 Ecosystem Services of T. cordata and R. pseudoacacia

Finally, the ecosystem services carbon storage, shading and cooling of *R. pseudoacacia* and *T. cordata* were calculated and evaluated. Highest values of all three ecosystem services were observed for *R. pseudoacacia* trees growing in parks. Of *T. cordata*, trees at public squares provided the greatest benefits. All in all, *R. pseudoacacia* could benefit better for the city climate regarding carbon storage and shading, while *T. cordata* fared better in terms of cooling at public squares and streets (Moser et al., 2015). Regarding the ecosystem services of both species in Munich and Würzburg, the trees in Würzburg provided a greater amount of services (Figure 21).
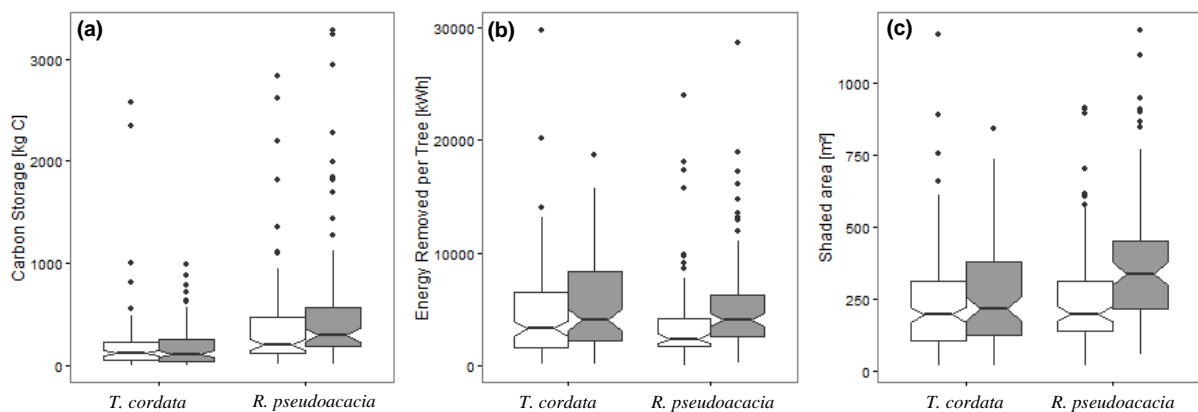


**Figure 21: Ecosystem services (a) carbon storage, (b) shading and (c) cooling effect of *T. cordata* and *R. pseudoacacia* in Munich (white) and Würzburg (grey)**

Age differences were found for all ecosystem services: The greater the age of the trees the more ecosystem services they could provide. *R. pseudoacacia* had greatest values for carbon storage due to the greater tree dimensions with increasing age, while *T. cordata* could remove more

energy from the atmosphere with greater age (see Moser et al. (2015)). The shaded area of both species was analyzed based on the detailed crown structure, there greater values for *T. cordata* were found for all dbh and age classes (Figure 22).



$$A_s = -0,44 + 0,33 \cdot Dbh$$
$$A_s = \phantom{-}0,75 + 0,24 \cdot Dbh$$
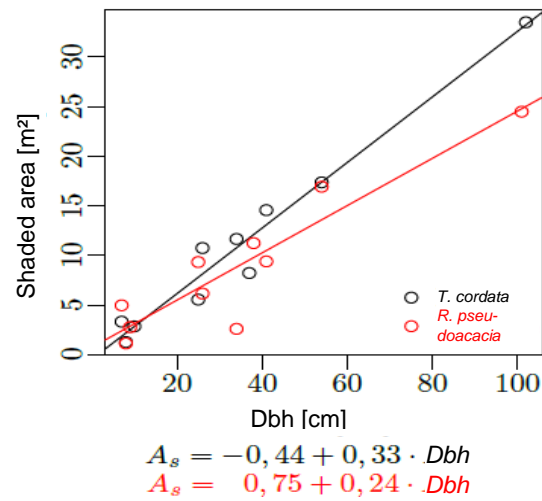
**Figure 22: Relationship of the shaded area AS with dbh of *T. cordata* (black) and *R. pseudo-acacia* (red)**

**5.    Discussion**

This thesis highlights the high diversity of growth dimensions and conditions affecting the growth of two urban tree species in two cities in southern Germany: the growth, structure and services of *R. pseudoacacia* and *T. cordata* were highly influenced by their microclimate and their local urban climate. Some of those differences were due to their different species characteristics. However the conditions at the growing site such as water and light availability (e.g., park trees compared to street trees) and the overall climate (precipitation rate of the city) were highly influential on growth as well.

*5.1  Differences in the Structure and Growth Patterns of T. cordata and R. pseudoacacia*

The analysis of the structure and growth of both species in Munich and Würzburg mostly revealed greater tree dimensions of *R. pseudoacacia* compared to *T. cordata*, which was mainly due to the higher average age of the sampled trees of *R. pseudoacacia*, although older trees of *T. cordata* could also reach great tree dimensions close to those of *R. pseudoacacia*. These differences in growth patterns can be explained by the different life strategies of both species: while *R. pseudoacacia* is a light-demanding pioneer species, characterized by fast growth at sites with favorable and unfavorable conditions, *T. cordata* is a slower but steady growing species that is shade-tolerant. Especially the high light demand of *R. pseudoacacia* was reflected by the weaker relationships of height and crown dimensions with dbh and age. These patterns were also found with the dendrochronological study, the increment cores of *R. pseudoacacia* revealed a faster growth compared to *T. cordata* in both cities Munich and Würzburg. The overall relationships of growth were computed on the basis of all trees together regardless of their growing site. However, street trees experience on average a higher light exposure compared to park trees. Park trees therefore are often characterized by a shift in their tree height/dbh relationship, especially light-demanding species (Harja et al., 2012; Vincent & Harja, 2008; Weiner, 2004). In contrast, the shade-tolerant *T. cordata* displayed stronger relationships of the analyzed tree dimensions ($r^2 > 0.7$).

One of the few studies dealing with the growth of urban trees is the one conducted by Kjelgren & Clark (1992). Similar to the study of this thesis, they analyzed the effects of the microclimate of street canyons, plazas and parks on the growth of urban trees. Differences in radiation, air temperature, vapor pressure deficit, and evaporation between the sites were found, with the greatest values at plazas resulting in the least tree vitality and growth due to higher evaporative demands and limited soil resources (Dobbertin, 2005). The results highlight the tremendously

different growth conditions of the three growth sites, illustrating the great variations affecting tree growth in the vicinity of a city.

As part of this thesis, the effects of the microclimate on tree growth was analyzed in detail with a study about the growth patterns of *T. cordata* at two contrasting sites (green lab Munich, see Moser et al. (2016a)) during the exceptional dry and hot summer of 2015. The studied squares were varied in terms of the planting pits (compacted paved squares of around 6 m² at Pariser Platz CPS and lawn stripes of around 55 m² at Bordeaux Platz OGS) and the surrounding meteorology. The meteorological differences were mainly due to the shape of the canyon: CPS had lower wind speed, showed more channeling effect compared to OGS. Similar to the differing growth of *T. cordata* and *R. pseudoacacia*, the structure of the trees was likely differing due to difference of age. The trees at CPS were on average 52 years older compared to OGS, resulting in greater tree dimensions and an overall smaller growth (age-trend) at CPS. In summary, the study illustrated how the growth patterns of a common urban tree species was influenced by the surrounding environment. The strongest (negative) relationships between the growth of *T. cordata* and influencing factors was found with transpiration, which was an effect of the water stress response of this species caused by the severe drought during the summer 2015. *T. cordata* is an anisohydric species, meaning the stomata close very slowly during droughts. However, while no growth reductions occur during the drought, the slow closure of stomata results in water management disruptions, great amounts of water loss and sustained carbon assimilation especially in late season (Sjöman et al., 2015).

The slow closure of stomata also resulted in high rates of transpiration and hence evaporative cooling. Therefore, *T. cordata* can provide high rates of evaporative cooling even during drought, although growth deficits can be observed as a consequence of the water stress response of this species several years after drought (Pretzsch et al., 2014; Río et al., 2014). In the future, tree species that achieve a better balance between growth, transpiration, and hence evaporative cooling should be preferred to mitigate the effects of climate change for better thermal comfort in cities.

### 5.2  Climate-growth Relationships of T. cordata and R. pseudoacacia

The observed water stress response of *T. cordata* was compared with the drought response of *R. pseudoacacia*. While *T. cordata* is an anisohydric species, *R. pseudoacacia* is characterized as an isohydric species, which close their stomata immediately during a drought event. Further, the stomatal regulation of isohydric species maintains a consistent minimum leaf water potential, while the leaf water potential of anisohydric species markedly decreases with changes in

evaporative demands (Klein, 2014). This strategy prevents great amounts of water loss and disruptions of the water balance. However, the stomata response reduces the carbon dioxide and finally leaf carbon uptake (Roman et al., 2015). Consequently, *R. pseudoacacia* reduces transpiration during drought, providing less evaporative cooling. Comparing the analyzed species during drought, *T. cordata* benefits with higher cooling compared to *R. pseudoacacia*, but reduced growth and ultimately vitality. *R. pseudoacacia* suffers growth decreases during drought, though quick recovery when stomata conductance is increased again. Both species achieved a trade-off between growth, transpiration and hence cooling. For future considerations, *T. cordata* can provide better cooling effects, however irrigation might be necessary to prevent excessive growth decreases, while *R. pseudoacacia* can sustain drought events better, providing less cooling for the environment.

Nevertheless, the local urban climate (climate of a city) affects tree growth and drought responses as well. In Munich, the dendrochronological study illustrated the observed patterns of water stress response of *T. cordata* and *R. pseudoacacia* nicely. Whereas the response analysis revealed no clear patterns regarding the influence of the monthly precipitation and temperature on tree ring formation in terms of city and species, the drought year analysis highlighted distinct differences. The drought year analysis SEA confirmed the greatest growth decrease of *T. cordata* in the years after a drought event, while *R. pseudoacacia* suffered most during drought but recovered quickly. On the contrary, in Würzburg, the patterns of the water stress response of both species were less distinct. Again, *T. cordata* displayed growth reductions in the years after drought, however less severe than in Munich. *R. pseudoacacia* showed no growth decrease in the drought year, only a very small positive growth was found, followed by a year with high growth rates and two consecutive years with great growth decreases. More surprisingly, both species had the greatest growth decline five years after the analyzed drought events. This can be explained with the occurrence of another drought event, since the SPEI revealed a high frequency of droughts of each five years in Würzburg. All in all, the different water stress strategies of the isohydric *R. pseudoacacia* and the anisohydric *T. cordata* can be found in Würzburg as well, but they are slightly changed and masked by the harsh growth conditions in Würzburg with an average of 430 mm less precipitation per year compared to Munich. Moreover, the frequency of drought events is higher in Würzburg than in Munich. Consequently, urban trees in Würzburg are affected by more water stress and have only short times to recover. Together with frequently occurring late-frosts, smaller planting pits in Würzburg compared to Munich (LÖK et al., 1990; Stadt Würzburg, 2014), the factors driving the growth of trees in Würzburg are more various and complex. Moreover, urban tree growth can be influenced by other factors

besides the extreme climate, e.g., different planting methods, the soil material or different fertilization methods affecting the water management and growth of urban trees (Rahman et al., 2011; Zaharah & Razi, 2009).

To analyze the underlying patterns of tree growth with the surrounding microclimate and local urban climate, the growth of *T. cordata* and *R. pseudoacacia* in normal years, in drought years and years with exceptional good years was studied in detail. All three models highlighted that especially the light availability drives the growth of both species, due to their species characteristics. *R. pseudoacacia* as a light demanding species showed growth decreases under low light conditions, while *T. cordata* was less affected by shade. During drought, the overall climate of the city and the species had distinct effects on growth. *R. pseudoacacia* displayed greater growth reductions during drought triggered by its isohydric strategy. The trees in Würzburg however, suffered more of drought than those in Munich enhanced by the overall low precipitation rates in Würzburg. During favorable years *R. pseudoacacia* again displayed its higher growth rate compared to *T. cordata*.

In conclusion, the growth patterns of urban trees are affected by various factors: While during normal years, species characteristics (fast growing to slower growing) dominate growth, under drought years, water stress strategies, the surrounding site conditions and the overall climatic conditions of the city control growth.

### 5.3 Impacts of T. cordata and R. pseudoacacia

Urban trees provide various ecosystem services, e.g., mitigation of rain water run-off, air quality improvement. The carbon storage, shading and cooling by evapotranspiration have been calculated in detail for the two analyzed species in the two cities of Munich and Würzburg. Regarding the ecosystem services carbon storage, shading and cooling, *R. pseudoacacia* could store more carbon and shade a greater area than *T. cordata* on average, while the latter had a greater cooling availability in average. The greater carbon storage and shading can be explained by the greater tree structures, the thick bark of *R. pseudoacacia* and great crown diameter. With increasing age, the provided ecosystem services of both species increased as well. When comparing the plantings in the three different types of open space, park trees of *R. pseudoacacia* and *T. cordata* trees on public squares provided the greatest benefits.

The found result can be compared with other studies. Especially regarding *T. cordata*, the results are similar to those by Yoon et al. (2013), who developed growth equations to quantify the above-ground biomass and carbon storage of five urban tree species in Daegu, Korea. They found a carbon storage capacity of *Platanus orientalis* ranging from 100 kg C in young trees to

350 kg C at a stem diameter of 50 cm. Further, Rahman et al. (2011) analyzed the cooling ability by transpiration of *Pyrus calleryana* and found maximum values of 7500 W for ten year old trees in August (max. 180 kW h per tree in one month), which are comparable values to the young *T. cordata* trees in this thesis.

When comparing the benefits of both species in Munich and Würzburg, the trees in Würzburg always provided the greatest amounts of services, due to their higher age and greater tree dimensions (especially greater for *R. pseudoacacia*, more similar for *T. cordata*).

Following Nowak & Crane (2002), the monetary value of the measured trees in Munich and Würzburg can be calculated by multiplying the stored carbon by \$20.3/t C based on the estimated marginal social costs of carbon dioxide emissions (Fankhauser, 1994). Due to the higher carbon storage, the measured *R. pseudoacacia* trees in Würzburg have an overall value of \$1,015,926 (€904,453, exchange rate on 16/09/2016), followed by the *R. pseudoacacia* trees in Munich (\$810,000 and €721,090, exchange rate at 16/09/2016), *T. cordata* in Munich (\$737,120 and €656,326, exchange rate on 16/09/2016) and lastly the analyzed *T. cordata* trees in Würzburg with \$472,821 (€420,978, exchange rate on 16/09/2016).

*5.4 Conclusion and Prospects*

For an adequate management and planning of urban green areas, knowledge about the growth and the services of urban trees is crucial. Mostly, urban trees need to fulfill certain demands in terms of aesthetics and spatial functions. Up to now, only limited data has been available on urban tree growth in central Europe, providing accurate data on tree growth over time together with provided benefits of urban trees at a certain age. With the results of this thesis, easy comprehensible spreadsheets were created (see Appendix), illustrating the life cycles of *T. cordata* and *R. pseudoacacia* with the development of certain tree structures over age and associated ecosystem services carbon storage, shading and cooling. Those spreadsheets enable city planners and scientists to obtain knowledge on the expected growth and benefits of two common urban tree species.

However, urban tree growth is influenced by several factors such as the local climate of a city (temperature, precipitation, altitude), the microclimate (growing site, tree pit, light availability), species characteristics (light- versus shade-tolerant, water stress response) and the climate of a certain year (drought year, favorable year). Therefore, the presented results can only be regarded as a rough overview of the growth ranges of *R. pseudoacacia* and *T. cordata* valid exclusively for Munich and Würzburg. Also, the measured trees represent a small proportion of the entirety of planted urban trees in both cities. As Peper et al. (2001b) mention, the growth

patterns of a species growing in a certain region should not be transferred to the same species growing under different climatic conditions, since the length of the growing season or the tree maintenance practices might be different. In addition, the dendrochronological approach was similarly based on a subset of *T. cordata* and *R. pseudoacacia* trees distributed over the whole city area, which proved to be a difficulty for chronology building. To provide a better understanding of the past growth of trees at different sites, a greater number of trees at each site should be sampled. The presented studies in Munich and Würzburg as well as the green lab in Munich analyzed tree growth during a short period of time under the present local and micro-climate conditions.

Nevertheless, this thesis highlights the impacts of urban trees on their environment and reversely the effects of the environment on urban tree growth. Understanding the development of urban tree dimensions can ensure a better assessment of tree services. For a proper tree management of vital, well-growing urban trees beneficial for the urban climate, the right tree species needs to be selected. As shown, *T. cordata* can provide high rates of cooling even in a drought year, though irrigation might be necessary to preserve tree vitality and growth. *R. pseudoacacia* can sustain droughts better, resulting in reduced cooling ability. Both species displayed different water stress strategies and an individual trade-off between drought endurance, growth and transpiration hence cooling.

Up to now, only limited data are available on urban tree growth, the impacts of trees on their environment and reversely, the effects of the stand conditions on urban tree growth. Especially knowledge under climate change conditions is still lacking, data with other typical urban tree species in more cities is necessary to close the mentioned knowledge gaps to gain more expertise for climate-fit, green cities with high biodiversity and vital, aesthetic urban trees. The performance of other species, foreign as well as introduced urban trees, also needs to be tested under current and future climate scenarios to find suitable tree species for the harsh urban environment to prevent reduced tree vitality and high management costs. There is an increasing number of studies worldwide trying to find climate-fit urban tree species. For example, the project 'Urban Green 2021' deals with the growth of foreign species in three very contrasting South German cities in terms of climate conditions with the aim to find suitable tree species under climate change (Böll et al., 2014). Moreover, based on the research of this thesis, a follow-up study with more species and additional cities is now underway to understand growth and functions of common urban trees. The provided ecosystem services of *T. cordata*, *R. pseudoacacia* and other studied species will further be validated and tested under climate change scenarios.

This thesis presents a first approach to gain more scientific data for sustainable tree planting to preserve diverse, green cities with healthy urban trees providing various benefits for the urban climate to mitigate the effects of climate change.

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# Contributions

- <u>Peer-reviewed journal paper:</u>
    - Rahman, M. A., Moser, A., Rötzer, T., Pauleit, S. (2016): Within canopy temperature differences and cooling ability of *Tilia cordata* trees grown in urban conditions. Building and Environment, in press, DOI 10.1016/j.buildenv.2016.12.013

    - Rahman, M. A., Moser, A., Rötzer, T., Pauleit, S. (2017): Microclimatic differences and their influence on transpirational cooling of *Tilia cordata* in two contrasting street canyons. Agricultural and Forest Meteorology 232: 443–456

    - Rötzer, T., Biber, P., Moser, A., Schäfer, C., Pretzsch, H.: Trees species mixing can mitigate inter-annual competition for water and stabilize growth by temporal diversification, particularly in drought periods, under review.

    - Rötzer, T., Moser, A., Rahman, M., Pauleit, S., Pretzsch, H. (2016): CityTree – climate sensitive model to simulate tree growth and environmental services. In prep.


- <u>Conference papers:</u>
    - Moser, A., Uhl, E., Pretzsch, H., Rötzer, T. (2014): Strukturerfassung und Ökosystemleistungen von Winterlinden und Robinien in München und Würzburg. BIOMET Konferenz Dresden

    - Moser, A., Rötzer, T., Pauleit, S., Pretzsch, H. (2016): Risk assessment: How do urban tree species respond to drought? Analyzing tree growth in relation to the surrounding urban environment. Urban Tree Diversity 2 Conference Melbourne

# Curriculum Vitae

## 1. PERÖNLICHE DATEN

| | |
|---|---|
| Name | **Moser, Astrid** |
| Geburtsort | 92237 Sulzbach-Rosenberg, Deutschland |
| Geburtstag | 06.06.1988 |
| Staatsangehörigkeit | Deutsch |
| Adresse | Giggenhauser Straße 1, 85354 Freising, Deutschland |
| E-Mail | astrid.moser@freenet.de |
| Tel. | +49 176 50990168 |

## 2. BERUFSERFAHRUNG

- seit 03/2014:

**Doktorarbeit** an der Technischen Universität München TUM, am Lehrstuhl für Wald-wachstumskunde (Prof. Dr. Dr. Pretzsch) zum Thema „Structure and Growth of Urban Trees and Their Impacts on the Environment - From Assessment to Model"
Bearbeitung der Projekte Stadtbäume im Klimawandel I + II, Baumlabor München, KROOF

- 03/2013 – 03/2014:

**Bioplan Höxter:** Erstellung naturschutzfachlicher Gutachten insbesondere zu Windkraft; Kartierungen von Großvogelarten, Fledermäusen, Brutvögel, Amphibien, Tagfalter, Flora

- 10/2010 – 12/2012:

**Wissenschaftliche Hilfskraft** am Lehrstuhl für Zoologie und am Lehrstuhl für Botanik

## 3. STUDIUM

- 10/2010 – 01/2013:

**M. Sc. Biologie** an der Universität Regensburg, Vertiefung in den Bereichen Naturschutz und Ökologie, Evolutionsbiologie, Botanik und Zoologie
Masterarbeit zum Thema „Die Teichbodenvegetation des Charlottenhofer Weihergebietes in Abhängigkeit von Bewirtschaftung und Weihergröße" (Note 1,3)

- 10/2007 – 09/2010:

**B. Sc. Biologie** an der Universität Regensburg
Bachelorarbeit zum Thema „Mehrfachpaarung der Ameisenart *Cardiocondyla obscurior"* (Note 1,7)

## 4. SCHULBILDUNG

- 09/2006 – 07/2007:

**Erwerb der allgemeinen Hochschulreife** an der Staatlichen Fachoberschule Schwandorf

- 09/2004 – 07/2006:
  **Erwerb der Fachhochschulreife** an der Staatlichen Fachoberschule Amberg

## 5. SOFTWARE

Sehr gute Kenntnisse in MS Office-Anwendungen (Word, Excel, Powerpoint), gute Kenntnisse von GIS (ArcGIS 9.x, ArcView) und Statistikprogrammen (R, SPSS).

## 6. SPRACHEN

- Deutsch (Muttersprache)
- English (berufliche Kompetenz)

# Appendix

## A Additional Material

**Table 5: Minimum, average and maximum tree dimension dbh, tree height, crown height, crown diameter, CPA and crown volume of *T. cordata* and *R. pseudoacacia* in five age classes with the sample number n**

| Age | n | Dbh [cm] | | | Tree Height [m] | | | Crown Height[m] | | | Crown Diameter[m] | | | CPA [m²] | | | Crown Volume [m³] | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | min | avg | max | min | avg | max | min | avg | max | min | avg | max | min | avg | max | min | avg | max |
| *Tilia cordata* | | | | | | | | | | | | | | | | | | | |
| <20 | 27 | 6,4 | 9,6 | 13,0 | 5,0 | 6,2 | 8,0 | 2,5 | 3,8 | 5,4 | 2,0 | 3,1 | 4,7 | 3,2 | 7,8 | 17,6 | 10,3 | 31,8 | 93,3 |
| 20-40 | 86 | 11,9 | 21,7 | 31,0 | 6,1 | 10,3 | 16,1 | 3,2 | 7,4 | 12,2 | 2,9 | 6,2 | 10,1 | 6,5 | 31,9 | 79,4 | 27,2 | 253,2 | 808,9 |
| 40-60 | 83 | 27,2 | 37,1 | 46,5 | 9,8 | 14,3 | 18,6 | 6,6 | 10,9 | 15,3 | 5,8 | 9,3 | 13,1 | 26,2 | 69,4 | 134,6 | 196,6 | 781,5 | 1561,2 |
| 60-80 | 22 | 44,8 | 51,7 | 60,7 | 14,2 | 18,1 | 23,3 | 9,7 | 14,5 | 19,2 | 8,9 | 12,0 | 15,1 | 61,8 | 115,1 | 179,3 | 797,8 | 1716,0 | 3206,3 |
| >80 | 8 | 63,0 | 78,1 | 107,0 | 11,4 | 19,1 | 27,1 | 7,0 | 15,6 | 22,5 | 9,1 | 13,2 | 18,4 | 65,1 | 144,8 | 266,8 | 455,7 | 2476,5 | 5276,4 |
| *Robinia pseudoacacia* | | | | | | | | | | | | | | | | | | | |
| <20 | 34 | 7,6 | 13,7 | 21 | 5,7 | 9,6 | 18 | 3,1 | 6,2 | 14,5 | 1,5 | 5,0 | 9,4 | 1,8 | 21,6 | 69,1 | 7,6 | 140,5 | 448,6 |
| 20 -40 | 67 | 21,2 | 30,4 | 40 | 7,3 | 13,6 | 24 | 4,7 | 9,6 | 20,5 | 5,2 | 7,9 | 12,6 | 21,5 | 50,6 | 124,1 | 101,0 | 506,6 | 1559,7 |
| 40-60 | 47 | 31,5 | 45,1 | 60 | 9,1 | 16,0 | 28 | 6,2 | 11,5 | 20,8 | 4,7 | 9,6 | 13,1 | 17,4 | 74,6 | 134,7 | 114,7 | 903,4 | 2188,2 |
| 60-80 | 33 | 41,1 | 57,4 | 80 | 12 | 16,8 | 27 | 8,4 | 13,0 | 21,8 | 5,3 | 11,5 | 17,4 | 21,8 | 107,3 | 238,3 | 274,4 | 1448,3 | 4075,4 |
| >80 | 14 | 60,9 | 84,8 | 102 | 14,7 | 21,3 | 27 | 11 | 17,3 | 23,5 | 10,1 | 14,9 | 20,1 | 80,7 | 179,2 | 315,9 | 887,6 | 3164,2 | 5748,9 |

**Table 6: Average dbh and tree height as well as minimum, average and maximum carbon storage, shading and cooling energy of *T. cordata* and *R. pseudoacacia* in five age classes with the sample size n**

| Age | n | Avg. Dbh [cm] | Avg. Tree Height [m] | Carbon Storage [kg C] | | | Shading [m²] | | | Cooling Energy [kWh] |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | | min | avg | max | min | avg | max | |
| *Tilia cordata* | | | | | | | | | | |
| <20 | 27 | 9,6 | 6,2 | 2 | 9 | 20 | 41 | 72 | 138 | 517 |
| 20–40 | 86 | 21,7 | 10,3 | 21 | 77 | 191 | 76 | 246 | 472 | 2553 |
| 40–60 | 83 | 37,1 | 14,3 | 157 | 301 | 638 | 235 | 508 | 868 | 6350 |
| 60–80 | 22 | 51,7 | 18,1 | 481 | 685 | 1098 | 586 | 829 | 1213 | 10797 |
| >80 | 8 | 78,1 | 19,1 | 876 | 1757 | 3465 | 390 | 1065 | 1704 | 14772 |
| *Robinia pseudoacacia* | | | | | | | | | | |
| <20 | 34 | 13,7 | 9,6 | 10 | 36 | 84 | 39 | 187 | 424 | 1164 |
| 20–40 | 67 | 30,4 | 13,6 | 70 | 226 | 593 | 142 | 413 | 1113 | 2916 |
| 40–60 | 47 | 45,1 | 16,0 | 177 | 597 | 1230 | 207 | 583 | 1341 | 4797 |
| 60–80 | 33 | 57,4 | 16,8 | 410 | 1016 | 2648 | 175 | 734 | 1444 | 6683 |
| >80 | 14 | 84,8 | 21,3 | 989 | 2699 | 4081 | 558 | 1207 | 1674 | 13644 |

# B Published Articles and Manuscripts

1. **Paper I**

   Moser, A., Rötzer, T., Pauleit, S., Pretzsch, H (2015): Structure and ecosystem services of small-leaved lime (*Tilia cordata* Mill.) and black locust (*Robinia pseudoacacia* L.) in urban environments. Urban Forestry & Urban Greening 14, 1110-1121

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## Urban Forestry & Urban Greening

# Structure and ecosystem services of small-leaved lime (*Tilia cordata* Mill.) and black locust (*Robinia pseudoacacia* L.) in urban environments

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### ARTICLE INFO

### ABSTRACT

Knowledge of the structure and morphology of common urban trees is scarce, particularly of the full life cycle of a tree. The present and future structural dimensions of urban tree species are of an increasing interest because tree growth is associated with its ecosystem services. The purpose of this study was to characterize the dimensions of two urban tree species (small-leaved lime, *Tilia cordata* Mill. and black locust, *Robinia pseudoacacia* L.) and to predict future structural dimensions based on the diameter at breast height and tree age. Regression equations were developed for tree height, crown diameter, crown height, crown volume, crown projection area, and open surface area of the tree pit. The data revealed strong relationships ($r^2 > 0.7$) between crown diameter, crown volume, crown projection area, crown height, tree pit for both species, and tree height of *T. cordata*. Based on tree dimensions and the leaf area index (LAI), three ecosystem services (carbon storage, shading, and cooling effects) were estimated for the analyzed trees. The results indicated that urban trees considerably improved the climate in cities, with carbon storage, shading, and cooling of single trees showing a direct relationship with LAI and age. The associations of tree growth patterns identified in this study can be used as guidelines for tree planting in cities and their ecosystem services; they may improve the management and planning of urban green areas.

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## Introduction

Urban trees are a central element in green space planning. During tree planting, city planners need to consider the fitness of trees in their specific environment and their future growth. However, the establishment of trees in urban areas faces several restrictions. Because of various influences, including limited root volume (Day et al., 1995; Grabosky and Bassuk, 1995), soil compaction (Beatty and Heckman, 1981; Day et al., 1995), high temperatures (Akbari et al., 2001; Kjelgren and Clark, 1992), less water availability (Beatty and Heckman, 1981; Whitlow and Bassuk, 1986), and mechanical injury (Beatty and Heckman, 1981; Foster and Blaine, 1978), urban sites considerably differ from forests (Nowak et al., 1990; Quigley, 2004). These stressful conditions in urban environments can hinder tree development. The growth and the corresponding tree age in urban areas, particularly of newly planted trees along streets, are often more limited than those of trees in parks or rural areas (Foster and Blaine, 1978; Rhoades and Stipes, 1999). Therefore, knowledge of the growth dimensions of tree species in association with the stand conditions is important. For selecting the right tree species for a location, quantitative information on tree growth from planting to maturity and decline is required (Peper et al., 2014). However, knowledge of the size and growth at a particular age and the effects of urban climates on tree growth remain unclear and less researched (Kjelgren and Clark, 1992; Peper et al., 2001a; Rust, 2014).

Modelling of tree growth is usually based on diameter at breast height (dbh). According to the pipe model theory (Chiba, 1998; Shinozaki et al., 1964a,b) and the functional carbon balance theory (Mäkelä, 1990), tree structures (e.g., crown volume) can be estimated from tree dimensions (e.g., dbh). Several studies are based on these theories and use dbh as an explanatory variable for forest stands, predicting tree height and crown dimensions (Pretzsch et al., 2012; Stage, 1973; Watt and Kirschbaum, 2011). McMahon (1975), however, promoted three similarity models to describe growth of trees as power law functions (Dahle and Grabosky, 2009). Structural developments of urban trees are of interest because of

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a direct association between tree dimensions and their ecological features and benefits (Troxel et al., 2013). Moreover, urban trees provide numerous ecosystem services to moderate microclimates and improve environmental conditions (Bolund and Hunhammar, 1999; Troxel et al., 2013), including carbon storage (McPherson, 1998; Nowak and Crane, 2002), mitigation of the heat island effect (Akbari et al., 2001), reduction of rainwater runoff (Bolund and Hunhammar, 1999; Xiao et al., 2000a,b), pollutant filtering (Bolund and Hunhammar, 1999; McPherson et al., 1997; Pretzsch et al., 2015), shading (Akbari et al., 2001), and cooling (Dimoudi and Nikolopoulou, 2003; Shashua-Bar and Hoffman, 2003).

Tree parameters are associated with these ecosystem services (Fig. 1). For example carbon storage-closely linked to the biomass of the tree-increases steadily with age (Var. 2, Yoon et al. (2013)), while the shading intensity of trees decreases with age (Var. 1, Chreptun 2015), whereas the leaf area index of a stand peaks at a young tree age (Var. 3, Ryan et al., 1997; Nock et al., 2008). Several studies (Scott et al., 1998; Stoffberg et al., 2008; Xiao et al., 2000a,b) revealed a direct association of these benefits with tree canopy and leaf area. The crown projection area and crown volume form the basis for leaf area calculation (Binkley et al., 2013; Pretzsch, 2014; Pretzsch et al., 2015). Precise leaf area estimates are important because leaf area influences key physiological processes and atmospheric fluxes such as air pollutant and rainfall interception (Peper and McPherson, 1998), photosynthesis (Kramer and Kozlowski, 1979), evapotranspiration (McPherson and Peper, 2012), respiration (McPherson and Peper, 2012), and shading (McPherson and Peper, 2012; Rijal et al., 2012).

Methods for predicting tree dimensions, including the crown projection area, crown volume, and leaf area, enable arborists, planners, and researchers to model the growth and benefits of urban trees. Based on tree allometry, alternative and improved management practices for urban trees can be defined and applied (McPherson et al., 2000; Peper et al., 2001b). However, most growth equations for urban trees have been developed for only a few species with limited size and age ranges (Rust, 2014; Troxel et al., 2013). Particularly, studies dealing with the development of urban trees over time and changes in their ecosystem services in association with their growth are limited. The growth of various urban trees in US cities was studied by Peper et al. (2001a,b), Peper et al. (2014), Rijal et al. (2012), and Troxel et al. (2013). However, only a few common European species have been analyzed to date, particularly regarding their ecological features. Larsen and Kristoffersen (2002) and Hasenauer (1997) studied the growth of *Tilia cordata*, a very common urban tree species of temperate regions. Although both studies provided information on the growth patterns of *T. cordata* over time, they did not establish associations with their ecosystem services. To the best of our knowledge, the dimensions and growth patterns of other urban species have not been systematically researched in European cities. Therefore, the aims of this study are

- to investigate growth associations of two common urban tree species covering several ecological niches: a shade-tolerant (small-leaved lime, *T. cordata* Mill.) and light-demanding (black locust, *Robinia pseudoacacia* L.) species,
- to establish tree growth associations from young, newly planted to very old trees,
- to predict tree height and crown dimensions (diameter, height, volume, and projection area) on the basis of stem diameter and age for *T. cordata* and *R. pseudoacacia* trees in street canyons, public places, and parks from low to high tree age,
- to determine and calculate ecosystem services of urban trees (carbon storage, shading, and cooling) depending on their growth

dimensions and age in order to illustrate the impact of trees for urban climates,
- closing the knowledge gap about the relationships of tree parameters with their ecosystem services and the change of the associated services of urban trees with tree age.

## Material and methods

### Site description and data collection

Tree data were collected in München (48°09′N, 11°35′E, 519 m a.s.l.) and Würzburg (49°48′N, 9°56′E, 177 m a.s.l.), southern Germany. They considerably differ in their climatic characteristics. The long-term annual precipitation means of München and Würzburg are 959 mm and 596 mm, respectively, whereas the mean annual temperature (1961–1990) in both cities is 9.1 °C (DWD, 2015). According to tree inventories, 750,000 trees have been planted in München and 40,000 in the municipal area of Würzburg. For this study, 225 *T. cordata* trees and 195 *R. pseudoacacia* trees were selected for measurement. We chose these two species because they markedly differ regarding their ecological features and are two of the more common tree species in the two cities and elsewhere (Pauleit et al., 2002). While *T. cordata* is a shade-tolerant species *R. pseudoacacia* requires a certain amount of light, therefore *T. cordata* is expected to have a higher leaf area index (LAI) than *R. pseudoacacia*. Tree selection was based on visual impression, and damaged, pruned, or low-forking trees were excluded. Tree data was collected from November 2013 to September 2014.

All the measured trees were classified as either park trees, trees in public places, or street trees. Trees were considered park trees when planted in a green area without buildings. Street trees were trees planted in a street canyon. As trees in public places-smaller, mostly paved spaces freely accessible to the public-only free-standing trees with open, detached crowns were selected. For each tree, the following information was recorded: diameter at breast height (dbh), tree height, height to live crown base, crown diameter, tree pit, vitality, coordinates and altitude, and distance to adjacent buildings and trees.

The stem diameter of all trees was measured with a diameter measurement tape at a height of 1.3 m. Because nursery workers and green space planners measure dbh at a height of 1 m, for the 64 *T. cordata* and 84 *R. pseudoacacia* trees, stem diameters at 1- and 1.3-m tree height, respectively, were measured to calculate a conversion factor with linear regressions and related regression coefficients (*a* and *b*) of the form:

$$\text{dbh}(1.0 \text{ m}) = a + b \times \text{dbh}(1.3 \text{ m}) \tag{1}$$

Tree height and crown base height (from the lowest primary branch to the top of the crown) were calculated using a Vertex Forestor. Crown radii and tree pit (open surface area) were measured in eight intercardinal directions (N, NE, ..., NW) along the ground surface with a measuring tape from the center of the trunk to the tip of the most remote downward-projecting shoot and to the last visible open, non-asphalted surface of the soil. The distances to the neighboring trees and buildings were estimated. The vitality of the trees was rated based on the scale of Roloff (2001), ranging from very good (0) to very poor (3) conditions regarding the branching structures of the crown.

### Calculations based on measured data

From the measured data, crown radius, crown diameter, crown volume, crown projection area, and biomass were

## Ecosystem services over age
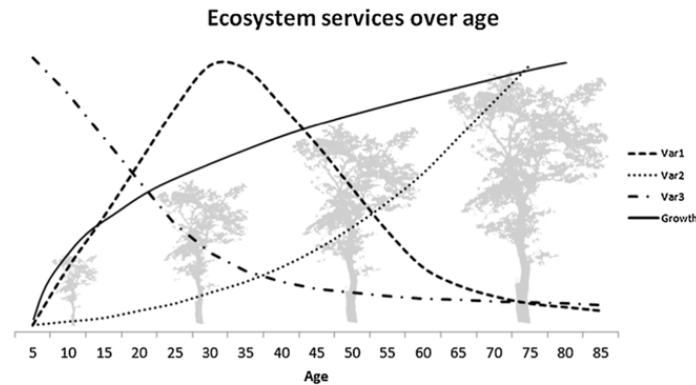


**Fig. 1.** Growth of an urban tree over time and hypothetical development of associated ecosystem services (Var1, Var2, and Var3).

calculated. The mean crown radius $cr$ was defined as the quadratic mean:

$$cr = \sqrt{\frac{(r_N^2 + r_{NE}^2 + \cdots + r_{NW}^2)}{8}} \qquad (2)$$

with $r_N$ as the widest measured crown extension in the northern direction, …, $r_{NW}$ the widest crown extension in the northwest direction.

The crown projection area CPA and the crown volume $cv$ were calculated as

$$CPA = cr^2 \times \pi \qquad (3)$$

$$cv = CPA \times \text{crown height} \qquad (4)$$

To obtain the aboveground woody biomass of *T. cordata*, the equations of Čihák et al. (2014) were applied. The woody biomass of *R. pseudoacacia* was computed with the functions of Clark and Schroeder (1986).

### Leaf area index

LAI was derived from hemispherical photographs captured during the fully leafed phase (July–August) using a Nikon Coolpix P5100 camera with fisheye lens and Mid-OMount. Between the time of measurement and the fully leafed phase, some trees were removed or severely pruned, and these trees were excluded from photography. The resulting hemispherical photos were analyzed with the programs WinSCANOPY (Régent Instruments Inc.). This software allows the calculation of the LAI by several methods. For *R. pseudoacacia*, LAI (Bonhom)-Lin was used, which is based on the Bonhomme and Chartier (1972) linear method. For *T. cordata*, the LAI (2000)-Lin calculated with the Licor LAI2000 linear method (Miller, 1967; Welles and Norman, 1991) resulted in the most reliable values. We performed regression analyses with generalized additive models (GAM, package mgvc) and mixed smoothed predictor variables to obtain specific models for calculating LAI on the basis of the measured tree dimensions. The best fitting model with the lowest Akaike Information Criterion (AIC) was chosen, for *T. cordata* resulting in:

$$\ln(LAI) = a + s(b_1) \times \ln(dbh) + s(b_2) \times \ln(h) + s(b_3) \times \ln(2 \times cr) \qquad (5)$$

and for *R. pseudoacacia* resulting in:

$$\ln(LAI) = a + s(b_1) \times \ln(h) + s(b_2) \times \ln(cv) + s(b_3) \times \ln(CPA) \qquad (6)$$

with $a$ as intercept, $s$ as smooth function for every slope variable ($b$) and $h$ as tree height.

### Age estimation

The age of all trees was estimated based on the measured tree parameters dbh and tree height. For *T. cordata*, the formula of Lukaszkiewicz and Kosmala (2008) was applied:

$$age = a + e^{((b + c \times dbh)/(100 + d \times h))} \qquad (7)$$

with $a$ = 264.073, $b$ = 5.5834, $c$ = 0.3397, $d$ = 0.0026, dbh in cm, and $h$ in m.

To obtain the age of *R. pseudoacacia*, we multiplied dbh with a species-dependent age factor of 0.996, which was computed by the measurements of Dwyer (2009) for *Gleditsia triacanthos*.

### Ecosystem services

To evaluate the benefits of urban trees for the city climate, we estimated three ecosystem services as examples "carbon storage, shading, and transpiration by cooling". To calculate the *carbon storage* of a tree, the biomass was multiplied with 0.5 following IPCC (2003) and Yoon et al. (2013). *Shading* can be obtained depending on crown height $h_c$ and crown radius $cr$ for each hour of the day by

$$A_S = 2 \times cr \times h_c \times f \times \cot(Y) \qquad (8)$$

with $A_S$ = shaded area in $m^2$, $f$ = correction factor for the crown form, $Y$ = elevation angle of the sun which in turn is a function of latitude, day of the year and hour of the day (e.g. DVWK, 1996), assuming a perpendicularly standing object on a horizontal surface (Häckel, 2012).

As shading coefficient we used the average shaded area of all sunlight hours of June 21st. For rough estimations of *transpiration*, and thus, the *cooling effect* of individual trees, simulation results of the process-based growth model BALANCE were used. This model simulates the growth and water balance of individual trees depending on their environment (Rötzer et al., 2010). Subsequently, the annual transpiration sums of the deciduous species European beech *Fagus sylvatica* were related to LAI [$m^2/m^2$] by regression analysis:

$$tra_a \, [mm] = -0.0168 \times LAI^2 + 0.4555 \times LAI + 0.2844 \quad (r^2 = 0.65) \qquad (9)$$

**Table 1**

Regression parameters for predicting diameter at breast height (dbh) at 1 m from dbh at 1.3 m for *T. cordata* and *R. pseudoacacia* of the form: dbh (1.0 m) = $a + b \times$ dbh (1.3 m). Listed are the regression coefficients (*a* and *b*) with standard error (SE), coefficients of determination ($r^2$), *F* and *p* values, and sample size (*n*). Dbh was measured and calculated in centimeters.

| Species | n | Coefficients | | | | F | $r^2$ | p |
|---|---|---|---|---|---|---|---|---|
| | | a | SE | b | SE | | | |
| Tilia cordata | 64 | 0.37 | 0.35 | 1.02 | 0.01 | 6269 | 0.99 | <0.001 |
| Robinia pseudoacacia | 84 | 0.17 | 0.22 | 1.01 | 0.01 | 21,740 | 0.99 | <0.001 |

To estimate the average annual energy withdrawn from the atmosphere for transpiration, the transpiration sum must be multiplied by the potential evaporation energy [2.34 MJ (kg H$_2$O)$^{-1}$].

### Statistical analysis

The software packages R, version 3.0.3 (R Core Team, 2014) was used for statistical analysis. First, regression analyses were performed to determine the associations between stem diameter and age, tree height, crown height, crown base, crown diameter, crown volume, crown projection area, and open surface area of the tree pit. Analysis of variance (ANOVA) with Tukey's HSD test was then performed to identify differences between the measured tree dimensions in the abovementioned categories (park, public place, street). Two Sample *t*-tests were applied to check for species differences regarding the ecosystem services. Log transformation ensured homogeneity of variances. Normal distribution of data was partially given, but is not a necessary assumption for ANOVA with high sample size (Stevens, 1999). Assumption of normality of residuals was assessed with graphical displace. All regressions were performed using log transformation of the tree dimensions, following Pretzsch et al. (2012), Stofberg et al. (2008), and Peper et al. (2001a):

$$\ln(y) = a + b \times \ln(x) \qquad (10)$$

With ordinary least squares (OLS) regression, the response (*y*) is calculated from the predictor (*x*). Because in application of the models, the growth of trees over time (*y*) will be estimated based on *x*, we selected OLS instead of reduced major axis or moving average regression (Niklas, 1994). For the back-transformation of the logarithm, the bias correction (CF) by Baskerville (1971) and Sprugel (1983) was applied:

$$CF = e^{\left(RSE^2/2\right)} \qquad (11)$$

with RSE = residual standard error.

### Results and discussion

#### Stem diameter conversion

A regression analysis revealed highly significant relationships between stem diameters at 1 m and 1.3 m for both species. The regression results are presented in Table 1.

#### Tree dimensions and growth equations

All measured and calculated tree dimensions are presented in Table 2. The minimum, maximum, and mean values are given in ascending age classes. The results indicate similar growth patterns of both species, and with increasing age, the tree dimensions also increased. Roman and Scatena (2011) cited several authors stating the average age of urban trees ranging from 13 years (Skiera and Moll, 1992), 15 years (Nowak et al., 2004) up to 73 years in *Acer saccharinum* (Richards, 1979). The average age of the trees in this study was 41 years for *T. cordata* and 39 years for *R. pseudoacacia*. Up to an age of approximately 80 years, the annual growth was high (crown diameter of *R. pseudoacacia* increased on average by 15 cm annually for all trees older than 60 years). For both species, a wide range of growth dimensions was included in the samples. The dbh of *T. cordata* ranged from 6.4 to 107 cm and that of *R. pseudoacacia* from 7.6 to 102 cm. *R. pseudoacacia* had the largest size at ages greater than 80 years, reaching a maximum stem diameter of 102 cm, tree height of 27 m, and crown diameter of 20.1 m. Overall, we found markedly higher growth rates for *R. pseudoacacia* than for *T. cordata*, resulting in larger trees at equal ages. For example, at ages of up to 20 years, *T. cordata* had an average tree height of 6.2 m and crown diameter of 3.1 m, whereas *R. pseudoacacia* reached an average height of 9.6 m and crown diameter of 5 m.

Linear regression analyses were used to determine the associations between dbh, tree height, and crown dimensions (Table 3). The regressions were highly significant at alpha levels of 0.05. The high $r^2$ (>0.7) for most tree variables indicates strong relationships. For *T. cordata*, dbh associated with the crown projection area, crown volume, and crown diameter displayed the strongest dependences.

**Table 2**

Measured and calculated tree dimensions: dbh, tree height, crown height, crown diameter, crown projection area (cpa), and crown volume of *T. cordata* and *R. pseudoacacia* in 5 ascending age classes of 20 years.

| Age | n | DBH [cm] | | | Tree height [m] | | | Crown height [m] | | | Crown diameter [m] | | | CPA [m²] | | | Crown volume [m³] | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Min | Mean | Max | Min | Mean | Max | Min | Mean | Max | Min | Mean | Max | Min | Mean | Max | Min | Mean | Max |
| *Tilia cordata* | | | | | | | | | | | | | | | | | | | |
| <20 | 27 | 6.4 | 9.6 | 13.0 | 5.0 | 6.2 | 8.0 | 2.5 | 3.8 | 5.4 | 2.0 | 3.1 | 4.7 | 3 | 8 | 18 | 10 | 32 | 93 |
| 20–40 | 86 | 11.9 | 21.7 | 31.0 | 6.1 | 10.3 | 16.1 | 3.2 | 7.4 | 12.2 | 2.9 | 6.2 | 10.1 | 7 | 32 | 79 | 27 | 253 | 809 |
| 40–60 | 83 | 27.2 | 37.1 | 46.5 | 9.8 | 14.3 | 18.6 | 6.6 | 10.9 | 15.3 | 5.8 | 9.3 | 13.1 | 26 | 69 | 135 | 197 | 782 | 1561 |
| 60–80 | 22 | 44.8 | 51.7 | 60.7 | 14.2 | 18.1 | 23.3 | 9.7 | 14.5 | 19.2 | 8.9 | 12.0 | 15.1 | 62 | 115 | 179 | 798 | 1716 | 3206 |
| >80 | 8 | 63.0 | 78.1 | 107.0 | 11.4 | 19.1 | 27.1 | 7.0 | 15.6 | 22.5 | 9.1 | 13.2 | 18.4 | 65 | 145 | 267 | 456 | 2477 | 5276 |
| *Robinia pseudoacacia* | | | | | | | | | | | | | | | | | | | |
| <20 | 34 | 7.6 | 13.7 | 21 | 5.7 | 9.6 | 18 | 3.1 | 6.2 | 14.5 | 1.5 | 5.0 | 9.4 | 2 | 22 | 69 | 8 | 141 | 449 |
| 20–40 | 67 | 21.2 | 30.4 | 40 | 7.3 | 13.6 | 24 | 4.7 | 9.6 | 20.5 | 5.2 | 7.9 | 12.6 | 22 | 51 | 124 | 101 | 507 | 1560 |
| 40–60 | 47 | 31.5 | 45.1 | 60 | 9.1 | 16.0 | 28 | 6.2 | 11.5 | 20.8 | 4.7 | 9.6 | 13.1 | 17 | 75 | 135 | 115 | 903 | 2188 |
| 60–80 | 33 | 41.1 | 57.4 | 80 | 12 | 16.8 | 27 | 8.4 | 13.0 | 21.8 | 5.3 | 11.5 | 17.4 | 22 | 107 | 238 | 274 | 1448 | 4075 |
| >80 | 14 | 60.9 | 84.8 | 102 | 14.7 | 21.3 | 27 | 11 | 17.3 | 23.5 | 10.1 | 14.9 | 20.1 | 81 | 179 | 316 | 888 | 3164 | 5749 |

**Table 3**

Results of linear logistic regressions with stem diameter (dbh) as predictor variable and the tree variables tree height, crown base, crown height, crown diameter, cpa, and crown volume as response variables for *T. cordata* and *R. pseudoacacia* (equation: ln(response) = $a + b \times$ ln(predictor)). The table lists the regression coefficients ($a$, $b$), coefficients of determination ($r^2$), residual standard errors (RSE) for bias correction, $F$-values, $p$-values, and standard errors (SE) of regression coefficients as well as the sample size ($n$).

| Species | Parameter | $n$ | $a$ | SE | $b$ | SE | $r^2$ | RSE | $F$ | $p$ |
|---|---|---|---|---|---|---|---|---|---|---|
| Tilia cordata | dbh vs tree height | 225 | 0.51 | 0.10 | 0.59 | 0.03 | 0.79 | 0.16 | 863.60 | <0.001 |
| | dbh vs crown base | 225 | 0.35 | 0.09 | 0.23 | 0.03 | 0.25 | 0.22 | 72.41 | <0.001 |
| | dbh vs crown height | 225 | −0.26 | 0.09 | 0.73 | 0.03 | 0.76 | 0.22 | 724.60 | <0.001 |
| | dbh vs crown diameter | 223 | −0.61 | 0.07 | 0.78 | 0.02 | 0.87 | 0.16 | 1479.00 | <0.001 |
| | dbh vs cpa | 223 | −1.47 | 0.14 | 1.56 | 0.04 | 0.87 | 0.33 | 1485.00 | <0.001 |
| | dbh vs crown volume | 223 | −1.73 | 0.20 | 2.29 | 0.06 | 0.87 | 0.48 | 1504.00 | <0.001 |
| Robinia pseudoacacia | dbh vs height | 195 | 1.14 | 0.10 | 0.42 | 0.03 | 0.55 | 0.22 | 233.70 | <0.001 |
| | dbh vs crown base | 195 | 0.85 | 0.16 | 0.13 | 0.04 | 0.04 | 0.35 | 8.47 | <0.001 |
| | dbh vs crown height | 195 | 0.38 | 0.12 | 0.54 | 0.03 | 0.59 | 0.25 | 279.70 | <0.001 |
| | dbh vs crown diameter | 195 | −0.01 | 0.09 | 0.60 | 0.02 | 0.76 | 0.19 | 615.90 | <0.001 |
| | dbh vs cpa | 195 | −0.25 | 0.17 | 1.20 | 0.05 | 0.76 | 0.38 | 613.80 | <0.001 |
| | dbh vs crown volume | 195 | 0.13 | 0.23 | 1.74 | 0.06 | 0.79 | 0.50 | 743.60 | <0.001 |

Crown base showed only a very weak association with stem diameter for both species. The correlation between dbh and height of *R. pseudoacacia* was highly significant; however, the $r^2$ value was only 0.55. Comparing both species, *T. cordata* revealed stronger relationships than *R. pseudoacacia*. Residual standard errors (RSE) ranged from 0.16 to 0.50, with the maximum value for crown volume.

Fig. 2 illustrates the log–log relationships of tree height, crown diameter, and crown volume with diameter at breast height (dbh) for both species. The models were fitted to the sampled data points, and the regression equation and associated $r^2$ values are shown. The weaker relationships of tree dimensions (particularly of dbh to height) of *R. pseudoacacia* compared to *T. cordata* possibly reflect that *R. pseudoacacia* is a very light-demanding species and that the measured trees grew unshaded as well as in the shade of high buildings or sheltered by other trees in parks. The growth allocation of trees can greatly change in response to light availability, particularly with respect to the shape of light-demanding species (Harja et al., 2012; Vincent and Harja, 2008; Weiner, 2004). The stronger relationship between the height and stem diameter of the shade-tolerant *T. cordata* supports this idea.

To illustrate the growth of *T. cordata* and *R. pseudoacacia* over time, the allometric relationships of height, crown diameter, and crown volume were plotted against age (Fig. 3). The relationships between age and tree dimensions show excellent fits ($r^2 > 0.8$) for *T. cordata* and moderate ($r^2 > 0.55$) to strong fits ($r^2 > 0.7$) for *R. pseudoacacia*. The development of space requirements of both species can be estimated from tree regressions. The results in Table 3 and the plots shown in Figs. 2 and 3 present relationships between the age and tree dimensions of both tree species in urban environments. All growth curves show the development from low to high age and represent a large part of a tree's lifecycle in urban surroundings. Therefore, it is possible to estimate the space occupied by a tree for any stem diameter and age. For example, at an age of 50, *T. cordata* will have an average height of 18 m, a crown diameter of 12.5 m, and a crown volume of 1379 m³. In comparison, at the same age *R. pseudoacacia* will have a height of 17 m, a crown diameter of 11 m, and a crown volume of 1031 m³.

Our results for the growth patterns of *T. cordata* show slightly weaker relationships for tree dimensions than do those of the studies conducted by Larsen and Kristoffersen (2002) and Hasenauer (1997). This difference is possibly based on the high variation of our sampled trees, which cover two cities with different climatic conditions, each with three different types of sites (park, public place, street). Compared with the results by Larsen and Kristoffersen (2002) who studied *T. cordata* in Copenhagen, *T. cordata* in München and Würzburg grew marginally larger. Considering the growth of both species in the studied cities in detail, we found only minor

**Table 4**

Regression statistics of the form ln(tree height) = $a + b \times$ ln(dbh) + CF with dbh as predictor variable and tree height as response.

| Species | City | $a$ | $b$ | $p$ |
|---|---|---|---|---|
| T. cordata | München | 0.52 | 0.59 | <0.001 |
| | Würzburg | 0.50 | 0.59 | <0.001 |
| R. pseudoacacia | München | 1.21 | 0.40 | <0.001 |
| | Würzburg | 1.04 | 0.45 | <0.001 |

differences for all tree parameters. Regression analysis revealed small alterations of the intercept ($a$), as Table 4 displays exemplarily for the tree dimensions stem diameter and height. The slope ($b$) of *T. cordata* is similar in München and Würzburg, while *R. pseudoacacia* shows a slightly faster height development in Würzburg ($b = 0.4$ in München to $b = 0.45$ in Würzburg).

To further interpret the growth patterns of *T. cordata* and *R. pseudoacacia* in urban surroundings, growth changes over age were investigated. The development of both species reveals slight changes over age; Fig. 4 exemplarily highlights the change in height in relation to age. While there is a minor difference in the growth patterns of *R. pseudoacacia* with age, *T. cordata* trees show a changed allometry with ages higher than 40 years. This is in line with the findings by Bertram (1989) and Genet et al. (2011) that constant scaling coefficients can mean an oversimplification especially in advanced tree development phases. The same effect can be found when regarding the branching allometry of both species. For *T. cordata* a shift toward less crown diameter increase with increasing dbh and age was found, similar to the growth patterns Dahle and Grabosky (2010) described for *Acer platanoides*. *R. pseudoacacia* showed only minor shifts in allometry.

These results support city planners in selecting tree species for public places, streets, and parks. Future space requirements of *T. cordata* and *R. pseudoacacia* trees in urban environments can be calculated on the basis of the presented relationships, and the functions and benefits of trees can be derived. Shading (Akbari et al., 2001), carbon storage (Yoon et al., 2013), cooling (Akbari et al., 2001), fine particle filtering (Bolund and Hunhammar, 1999), noise reduction (Bolund and Hunhammar, 1999; Troxel et al., 2013), and other ecosystem services can be estimated from growth patterns and tree dimensions.

*Site differences*

Analysis of variance (ANOVA) was used to determine the influence of tree species and site (park, public place, street) on tree structure and growth. Tables 5 and 6 present mean values and
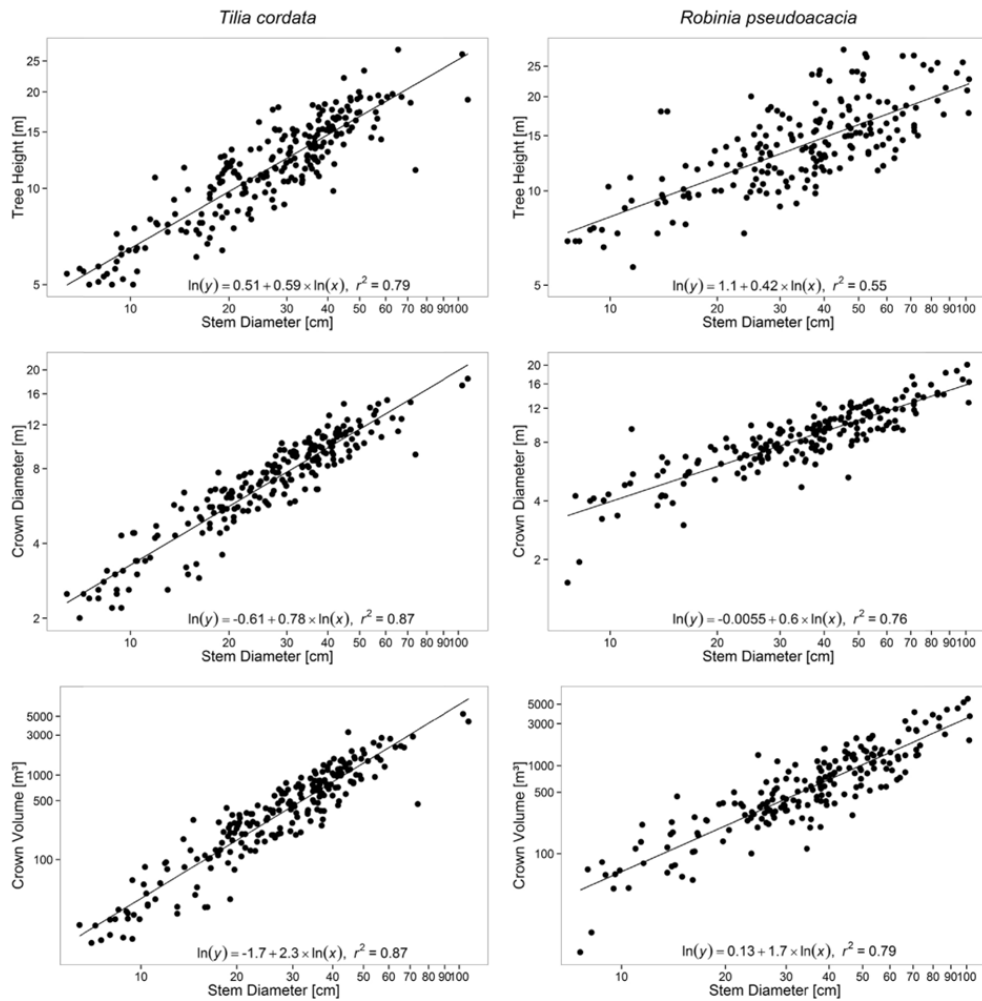
**Fig. 2.** Allometric logistic relationships between stem diameter and (1) tree height, (2) crown diameter, and (3) crown volume for *T. cordata* and *R. pseudoacacia*. The statistical characteristics of the regression lines are presented in Table 3.

standard deviations for all measured and calculated tree variables in the three categories and the ANOVA results.

The variables dbh, age, tree height, crown base, crown diameter, crown projection area, crown volume, and vitality of *T. cordata* revealed no significant differences among the site categories. Only for crown height and open surface area, park trees considerably differed from street trees ($p = 0.02$ for crown height and $p < 0.001$ for open surface area). *T. cordata* planted in public places had crown heights similar to those of park and street trees (Table 5). For *R. pseudoacacia*, all tree variables except vitality markedly differed between site types ($p = 0.13$ for vitality). Park trees had markedly greater tree dimensions than street trees. Trees in public places had stem diameters, ages, tree heights, crown heights, and open surface areas similar to those of park and street trees ($p < 0.001$). For crown diameter, crown projection area, and crown volume, *R. pseudoacacia* at public places significantly differed from both park and street

trees ($p < 0.001$). The overall vitality of *T. cordata* was slightly better than that of *R. pseudoacacia*, with means of 0.9 and 1.2, respectively, both ranged from very good to good ($p = 0.3$).

The results presented in Tables 5 and 6 indicate that *T. cordata* grew in a similar pattern independent of the site type, whereas *R. pseudoacacia* was more variable in its growth dimensions based on the site type. ANOVA revealed that park trees of *R. pseudoacacia* are markedly older than street trees ($p < 0.001$), an age difference that should be carefully considered. Owing to this difference, park trees may have greater tree dimensions, possibly leading to misinterpretations of the impact of site conditions on tree growth.

Some of the species differences in the growth patterns related to the growth site, city, and age can also be caused by the open surface area of the tree (tree pit). Regression analyses indicate that the growth of *T. cordata* and *R. pseudoacacia* is highly influenced by the tree pit (Table 7). Sanders et al. (2013) and Grabosky and Gilman
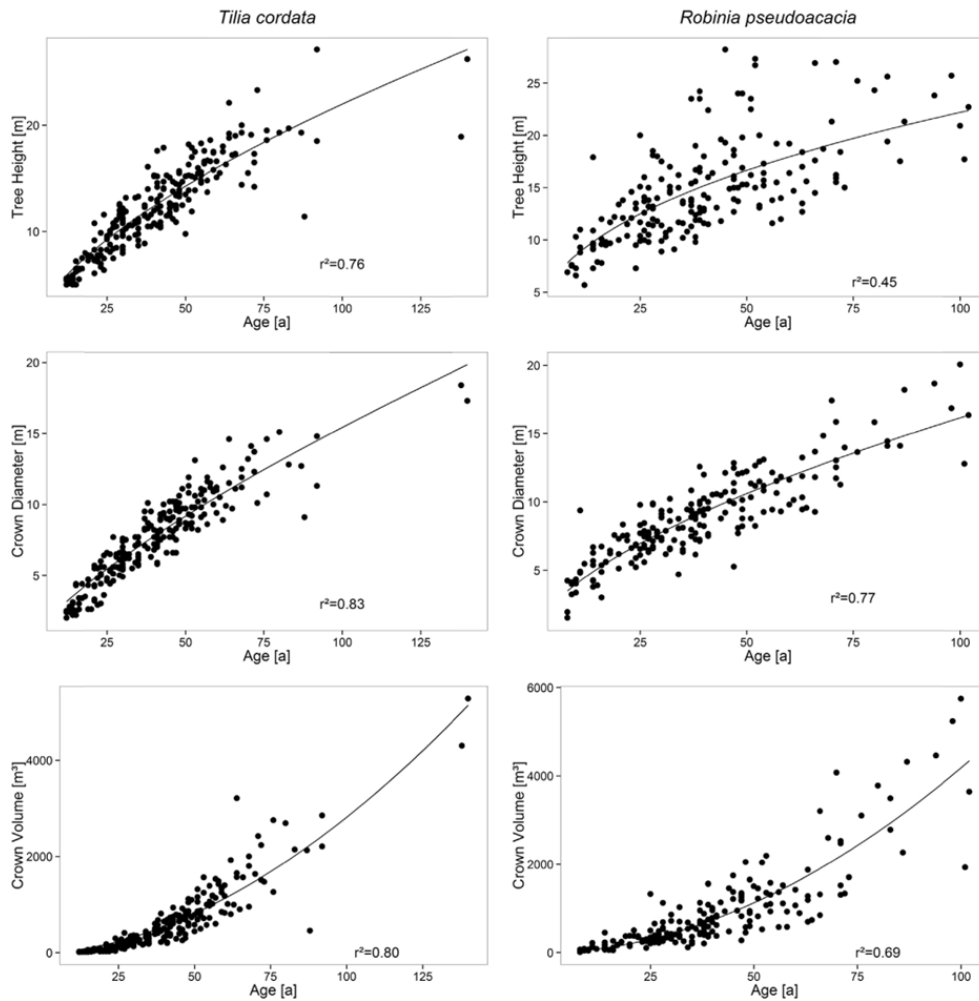
**Fig. 3.** Allometric nonlinear least square relationships between age and (1) tree height, (2) crown diameter, and (3) crown volume of the equation $yY = aY \times x^b$ for *T. cordata* and *R. pseudoacacia*, with resulting $r^2$ values.

(2007) as well found significant growth differences in relation to planting space, revealing that trees with reduced planting space show reduced maximum size. Since reduced planting space induces a limited soil volume and limited water availability, tree growth is often inhibited leading to higher tree mortality.

Further, because park trees may be exposed to less water stress, higher growth rates in parks may be expected (Kjelgren and Clark, 1992; Whitlow and Bassuk, 1988). Our results support this expectation for most tree dimensions of *R. pseudoacacia*. They revealed smaller tree dimensions in streets but considerably greater dimensions in parks. This difference can reflect light and hence photosynthetic limitations in street canyons (Kjelgren and Clark, 1992). Kjelgren and Clark (1992) investigated the microclimates and growth of urban trees in park, streets, and public places and found marked growth differences between the sites. Trees in

public places had a substantially lower leaf area and diameter increment, a finding mainly explained by higher water stress in plazas. Depending on the species, differences between site conditions can considerably influence the growth of urban trees.

*Ecosystem services*

The statistical relationships between tree dimensions and LAI ($m^2/m^2$) derived with generalized additive models are presented in Table 8.

The mean LAI of *T. cordata* was 2.60, ranging from 0.98 to 5.29, while that of *R. pseudoacacia* trees was 1.49, ranging from 0.42 to 5.0. Rauner (1976) found a higher LAI of 4.78 for *T. cordata*. The LAI of *R. pseudoacacia* is lower than that stated by Dickmann et al. (1985) for pure plantings (4.3–4.9) and mixed stands (3.2–3.3) in the US

1117
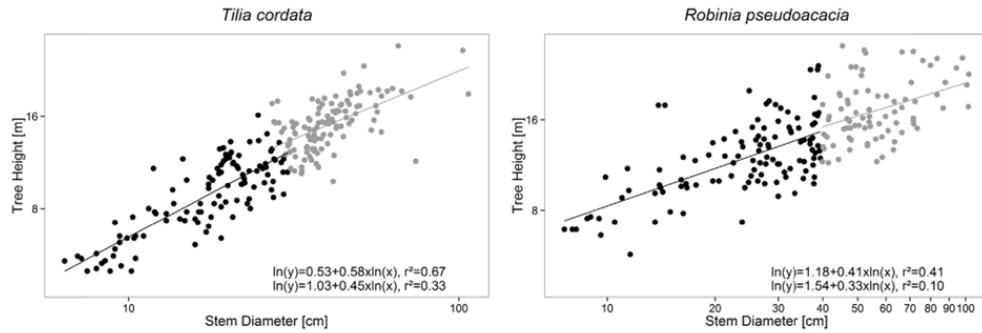


**Fig. 4.** Allometric logistic relationships between stem diameter and tree height for *T. cordata* and *R. pseudoacacia* at an age lower than 40 (black) and over 40 (grey). Upper regression equations are for age lower than 40, lower equations are for age over 40.

**Table 5**

Mean values of the measured and calculated tree variables age, dbh, tree height, crown base, crown height, and related standard deviation (SD) in response to growth site for *T. cordata* and *R. pseudoacacia* and *p*-values for each ANOVA. Mean values in the same column differ significantly when followed by different letters (Tukey's test, $p < 0.05$), $n$ = sample size.

| Site | $n$ | Age [predicted] | | DBH [cm] | | Tree height [m] | | Crown base [m] | | Crown height [m] | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Mean | SD | Mean | SD | Mean | SD | Mean | SD | Mean | SD |
| | | $p = 0.16$ | | $p = 0.18$ | | $p = 0.08$ | | $p = 0.13$ | | $p = 0.02$ | |
| *Tilia cordata* | | | | | | | | | | | |
| Park | 61 | 44 a | 23 | 30.5 a | 15.9 | 13.0 a | 4.4 | 2.9 a | 0.7 | 10.1 a | 4.1 |
| Public place | 69 | 48 a | 24 | 33.6 a | 16.6 | 12.8 a | 4.2 | 3.1 a | 0.8 | 9.7 ab | 3.9 |
| Street | 95 | 41 a | 20 | 28.6 a | 14.3 | 11.6 a | 3.9 | 3.1 a | 0.8 | 8.4 b | 3.5 |
| | | $p < 0.001$ | | $p < 0.001$ | | $p < 0.001$ | | $p < 0.001$ | | $p < 0.001$ | |
| *Robinia pseudoacacia* | | | | | | | | | | | |
| Park | 45 | 78 a | 39 | 49.7 a | 25.2 | 18.6 a | 5.1 | 4.9 a | 2.7 | 13.7 a | 4.5 |
| Public place | 63 | 60 ab | 25 | 38.6 ab | 16.5 | 14.1 b | 4.1 | 3.7 b | 1.1 | 10.4 b | 4.4 |
| Street | 87 | 55 b | 29 | 35.6 b | 18.9 | 12.9 b | 3.7 | 3.7 b | 1.1 | 9.2 b | 3.3 |

**Table 6**

Mean values of the measured and calculated tree variables crown diameter, crown projection area (cpa), crown volume, open surface area, vitality, and related standard deviation (SD) in response to growth site for *T. cordata* and *R. pseudoacacia* and *p*-values for each ANOVA. Mean values in the same column differ significantly when followed by different letters (Tukey's test, $p < 0.05$), $n$ = sample size.

| Site | $n$ | Crown diameter [m]* | | CPA [m²]* | | Crown volume [m³]* | | Open surface area [m²]* | | Vitality | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Mean | SD | Mean | SD | Mean | SD | Mean | SD | Mean | SD |
| | | $p = 0.23$ | | $p = 0.24$ | | $p = 0.11$ | | $p < 0.001$ | | $p = 0.13$ | |
| *Tilia cordata* | | | | | | | | | | | |
| Park | 61 | 7.9 a | 2.9 | 55 a | 42 | 701 a | 889 | 40.0 a | 33.8 | 0.8 a | 0.6 |
| Public place | 69 | 8.2 a | 3.3 | 61 a | 43 | 723 a | 691 | 26.1 b | 26.9 | 1.0 a | 0.6 |
| Street | 95 | 7.3 a | 3.0 | 48 a | 38 | 525 a | 595 | 19.5 b | 21.7 | 0.8 a | 0.6 |
| | | $p < 0.001$ | | $p < 0.001$ | | $p < 0.001$ | | $p < 0.001$ | | $p = 0.35$ | |
| *Robinia pseudoacacia* | | | | | | | | | | | |
| Park | 45 | 10.9 a | 3.9 | 105 a | 72 | 1589 a | 1362 | 74.7 a | 59.1 | 1.3 a | 0.8 |
| Public place | 63 | 8.9 b | 2.6 | 67 b | 37 | 805 b | 687 | 17.7 b | 24.2 | 1.3 a | 0.7 |
| Street | 87 | 7.9 c | 2.7 | 55 c | 36 | 596 c | 657 | 12.0 b | 21.4 | 1.1 a | 0.6 |

* $n = 67$ of *T. cordata* at public places.

during the early growing season but very similar to the computed LAI in August (1.7).

Based on the calculated LAI and biomass, the ecosystem services of shading, carbon storage, and cooling effect were estimated for both species of different age classes (Table 9).

The carbon storage capacity of both tree species considerably differs ($t = -8.12$, $p < 0.001$), with *R. pseudoacacia* storing more than twice as much carbon than *T. cordata* in average. Although the average carbon storage of a *T. cordata* tree at the age of 80 years can reach

1341 kg, *R. pseudoacacia* of the same age can store up to 2119 kg C. A maximum storage capacity of more than 3000 kg C was found for park and street trees of *R. pseudoacacia*. These results are similar to those by Yoon et al. (2013), who developed growth equations to quantify the aboveground biomass and carbon storage of five urban tree species in Daegu, Korea. They found a carbon storage capacity of *Platanus orientalis* ranging from 100 kg C in young trees to 350 kg C at a stem diameter of 50 cm. *R. pseudoacacia* achieved higher benefits than *T. cordata* up to an age of 40 to 60 years

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**Table 7**

Results of linear logistic regressions with tree pit (open surface area) as predictor variable and the tree variables dbh, tree height, crown base, crown diameter, crown height, cpa, crown volume, and vitality as response variables for *T. cordata* and *R. pseudoacacia* (equation: $\ln(\text{response}) = a + b \times \ln(\text{predictor})$). The table lists the regression coefficients $(a, b)$, coefficients of determination $(r^2)$, residual standard errors (RSE) for bias correction, F-values, p-values, and standard errors (SE) of regression coefficients as well as the sample size $(n)$.

| Species | Parameter | $n$ | $a$ | SE | $b$ | SE | $r^2$ | RSE | F | $p$ |
|---|---|---|---|---|---|---|---|---|---|---|
| Tilia cordata | Tree pit vs dbh | 221 | 2.36 | 0.09 | 0.33 | 0.03 | 0.35 | 0.44 | 122.2 | <0.001 |
| | Tree pit vs height | 221 | 1.84 | 0.06 | 0.21 | 0.02 | 0.35 | 0.29 | 118.3 | <0.001 |
| | Tree pit vs crown base | 221 | 1.00 | 0.05 | 0.03 | 0.02 | 0.01 | 0.25 | 3.9 | 0.05 |
| | Tree pit vs crown diameter | 221 | 1.12 | 0.07 | 0.30 | 0.02 | 0.41 | 0.35 | 157.7 | <0.001 |
| | Tree pit vs crown height | 221 | 1.34 | 0.07 | 0.28 | 0.02 | 0.37 | 0.36 | 133.1 | <0.001 |
| | Tree pit vs cpa | 221 | 1.99 | 0.14 | 0.59 | 0.05 | 0.41 | 0.70 | 158.1 | <0.001 |
| | Tree pit vs crown volume | 221 | 3.33 | 0.21 | 0.87 | 0.07 | 0.42 | 1.02 | 161.1 | <0.001 |
| | Tree pit vs vitality | 221 | 1.21 | 0.12 | −0.12 | 0.04 | 0.03 | 0.59 | 8.6 | 0.004 |
| Robinia pseudoacacia | Tree pit vs dbh | 193 | 2.97 | 0.08 | 0.22 | 0.03 | 0.23 | 0.50 | 58.4 | <0.001 |
| | Tree pit vs height | 193 | 2.27 | 0.04 | 0.14 | 0.02 | 0.29 | 0.27 | 81.8 | <0.001 |
| | Tree pit vs crown base | 195 | 1.21 | 0.06 | 0.04 | 0.02 | 0.01 | 0.36 | 3.7 | 0.06 |
| | Tree pit vs crown diameter | 193 | 1.70 | 0.05 | 0.16 | 0.02 | 0.27 | 0.33 | 72.7 | <0.001 |
| | Tree pit vs crown height | 193 | 1.85 | 0.06 | 0.17 | 0.02 | 0.28 | 0.34 | 76.7 | <0.001 |
| | Tree pit vs cpa | 195 | 3.15 | 0.11 | 0.32 | 0.04 | 0.27 | 0.67 | 72.6 | <0.001 |
| | Tree pit vs crown volume | 193 | 5.00 | 0.15 | 0.49 | 0.05 | 0.31 | 0.92 | 88.1 | <0.001 |
| | Tree pit vs vitality | 193 | 1.35 | 0.11 | −0.05 | 0.04 | 0.01 | 0.68 | 2.0 | 0.16 |

**Table 8**

Model parameters and sample size $(n)$ of the calculated models for *T. cordata* $[\ln(\text{LAI}) = a + s(b_1) \times \ln(\text{dbh}) + s(b_2) \times \ln(\text{tree height}) + s(b_3) \times \ln(\text{crown diameter})]$ and *R. pseudoacacia* $[\ln(\text{LAI}) = a + s(b_1) \times \ln(\text{tree height}) + s(b_2) \times \ln(\text{crown volume}) + s(b_3) \times \ln(\text{CPA})]$ with smooth function $s$ for every slope variable $(b)$.

| Species | $n$ | $a$ | SE | $r^2$ | $p$ |
|---|---|---|---|---|---|
| Tilia cordata | 193 | 0.93 | 0.03 | 0.35 | <0.001 |
| Robinia pseudoacacia | 129 | 0.26 | 0.07 | 0.22 | <0.001 |

**Table 9**

Mean, minimum (min), and maximum (max) predicted carbon storage in kg C per tree, and shaded area in m² per tree, as well as the cooling effect as mean energy removed from the atmosphere per tree in kW h in summer month for *T. cordata* and *R. pseudoacacia* from planting at <20 to high age >80 in ascending age classes with overall average values for all trees per species and sample size $(n)$.

| Age | $n$ | C-storage [kg C] | | | Shading [m²] | | | Energy removed from atmosphere [kW h] |
|---|---|---|---|---|---|---|---|---|
| | | Min | Mean | Max | Min | Mean | Max | |
| *Tilia cordata* | | | | | | | | |
| <20 | 27 | 3 | 8 | 17 | 18 | 36 | 75 | 517 |
| 20–40 | 86 | 17 | 62 | 126 | 32 | 144 | 323 | 2553 |
| 40–60 | 83 | 124 | 219 | 356 | 131 | 310 | 505 | 6350 |
| 60–80 | 22 | 376 | 494 | 714 | 327 | 533 | 843 | 10,797 |
| >80 | 8 | 781 | 1341 | 2571 | 192 | 718 | 1171 | 14,772 |
| Average | | | 196 | | | 246 | | 4843 |
| *Robinia pseudoacacia* | | | | | | | | |
| <20 | 34 | 11 | 55 | 132 | 19 | 96 | 275 | 1164 |
| 20–40 | 67 | 99 | 207 | 402 | 74 | 230 | 604 | 2916 |
| 40–60 | 47 | 124 | 424 | 856 | 93 | 341 | 701 | 4797 |
| 60–80 | 33 | 289 | 746 | 1990 | 201 | 456 | 913 | 6683 |
| >80 | 14 | 718 | 2119 | 3284 | 334 | 781 | 1188 | 13,644 |
| Average | | | 461 | | | 312 | | 4472 |

regarding the ecosystem functions of shading and cooling (energy removed from the atmosphere). With higher age, *T. cordata* provided more cooling and similar shade than *R. pseudoacacia*. Rahman et al. (2011) analyzed the cooling ability by transpiration of *Pyrus calleryana* and found maximum values of 7500 W for ten year old trees in August (max. 180 kW h per tree in one month), which are comparable values to the young *T. cordata* trees in our study. In average, *R. pseudoacacia* can provide significant higher shade compared to *T. cordata* ($t = -3.95$, $p < 0.001$), while the cooling ability of both species is more similar ($t = 0.39$, $p = 0.69$).

Fig. 5 reveals the average carbon storage, shading, and cooling effect of *T. cordata* and *R. pseudoacacia* for the different site categories park, public place, and street; Table 10 provides the associated p-values for each comparison.

Street trees of *R. pseudoacacia* provided the least carbon storage ($F = 9.9$, $p < 0.001$), shading ($F = 16.8$, $p < 0.001$), and cooling ability ($F = 17.4$, $p < 0.001$) of the measured sites, while park trees

exhibited the highest ecosystem services compared to public places and street. *T. cordata* trees in public places had higher ecosystem services than did park and street trees, but all values were not significant (carbon storage: $F = 1.8$, $p = 0.16$, shading: $F = 2.6$, $p = 0.07$, cooling effect: $F = 2.3$, $p = 0.10$). Because park trees of *R. pseudoacacia* were markedly older than trees at streets and public places, the greater services also may be because of the age differences.

Urban trees can considerably improve city climates by providing ecosystem services. The results of our research support those of other studies dealing with microclimates amelioration by temperature reduction (Souch and Souch, 1993), shading and cooling (Akbari et al., 2001; Dimoudi and Nikolopoulou, 2003), humidity increase (Dimoudi and Nikolopoulou, 2003; Georgi and Zafiriadis, 2006), and air quality improvement (Akbari et al., 2001). Further investigations will enable comparisons of tree species growth and their benefits to urban green space planning. Particularly with dendrochronology analyses and the integration of climate and
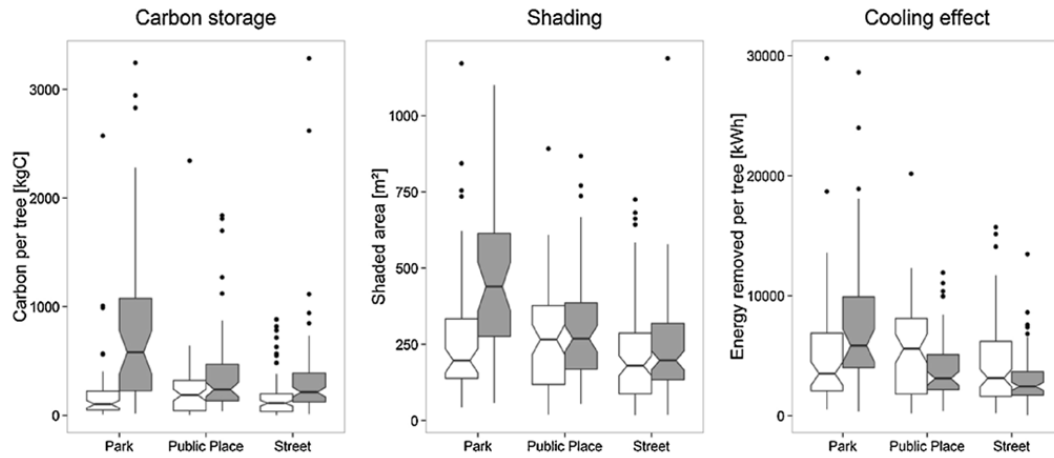
**Fig. 5.** Box plots of the ecosystem services: carbon storage, shading, and cooling effect of *T. cordata* (white) and *R. pseudoacacia* (gray) and for different sites (park, public place, and street).

**Table 10**

*p*-Values of conducted analysis of variance (ANOVA, Tukey's test) for differences regarding the ecosystem services of *T. cordata* (T, right upper part) and *R. pseudoacacia* (R, left lower part) for growing sites park (Pa), public place (Pl), and street (St), significant differences marked bold.

| Carbon storage | | | | Shading | | | | Cooling effect | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| R\T | Pa | Pl | St | R\T | Pa | Pl | St | R\T | Pa | Pl | St |
| Pa | | 0.88 | 0.43 | Pa | | 0.80 | 0.08 | Pa | | 0.97 | 0.24 |
| Pl | **0.01** | | 0.16 | Pl | **0.003** | | 0.28 | Pl | **0.001** | | 0.14 |
| St | **<0.001** | 0.27 | | St | **<0.001** | **0.04** | | St | **<0.001** | **0.04** | |

soil conditions in tree growth modeling will improve knowledge of urban tree growth patterns in association with their services. Moreover, future studies should be based on specific biomass functions for *T. cordata* and *R. pseudoacacia*, such as those generated by terrestrial laser scanning. With scanned images of trees, information about leaf distribution and crown transparency can be obtained. The shaded area was calculated based on crown height and diameter, while crown transparency was not considered. With more precise information about leaf area, leaf density, and leaf distribution within the tree crown, accurate shading values and process-based modeling of urban tree growth can be derived from laser scanning and improved, respectively.

## Conclusions

For an adequate planning of urban green areas and the realization of certain demands (esthetics and spatial functions), modeling the growth patterns of urban trees is crucial. Because empirical growth modeling of urban trees is scarce, we analyzed the growth of the common urban tree species *T. cordata* and *R. pseudoacacia*. With the shown conversion equations, dbh at 1.3 m could be transformed into stem diameter at 1 m. By using the identified relationships tree dimensions of *T. cordata* and *R. pseudoacacia* in urban environments of south German cities could be predicted for any stem diameter and age. The growth patterns of both species were strongly correlated; though *R. pseudoacacia* showed slightly weaker relationships. Because urban trees face challenges, such as less water availability, limited root volume (Grabosky and Gilman, 2007; Sanders et al., 2013), and higher temperatures (Akbari et al., 2001), tree growth for the site categories park, public place, and street canyon were

analyzed. The results revealed similar growth patterns of *T. cordata*, regardless of their site while *R. pseudoacacia* exhibited markedly better growth in parks. Other factors like age and open surface area had influence on the growth patterns of both species. Regression analysis confirmed the importance of the open surface area for the development of urban trees. The estimation of ecosystem services like carbon storage, shading, and cooling by transpiration indicate that *T. cordata* and *R. pseudoacacia* can improve the urban microclimate. Thus, the ecosystem services provided by both species are beneficial for the city climate and increase human thermal comfort (Bolund and Hunhammar, 1999; Georgi and Zafiriadis, 2006). An understanding of the development of single tree dimensions can facilitate benefit assessment that trees provide for urban climates. The results indicate the virtue of urban trees, however some are up to now only rough estimates. The present study facilitates tree selection based on space requirements and their functions for the city climate. Up to now, only limited data are available for urban tree growth in central Europe; more research dealing with allometry and functions of trees in urban areas is required.

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## 2. Paper II

Moser, A., Rötzer, T., Pauleit, S., Pretzsch, H (2016): The Urban Environment Can Modify Drought Stress of Small-Leaved Lime (*Tilia cordata* Mill.) and Black Locust (*Robinia pseudoacacia* L.). Forests 7, 71

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# The Urban Environment Can Modify Drought Stress of Small-Leaved Lime (*Tilia cordata* Mill.) and Black Locust (*Robinia pseudoacacia* L.)

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**Abstract:** The urban environment characterized by various stresses poses challenges to trees. In particular, water deficits and high temperatures can cause immense drought stress to urban trees, resulting in reduced growth and die-off. Drought-tolerant species are expected to be resilient to these conditions and are therefore advantageous over other, more susceptible species. However, the drought tolerance of urban trees in relation to the specific growth conditions in urban areas remains poorly researched. This study aimed to analyze the annual growth and drought tolerance of two common urban tree species, namely small-leaved lime (*Tilia cordata* Mill. (*T. cordata*)) and black locust (*Robinia pseudoacacia* L. (*R. pseudoacacia*)), in two cities in southern Germany in relation to their urban growing conditions. Marked growth reductions during drought periods and subsequent fast recovery were found for *R. pseudoacacia*, whereas *T. cordata* exhibited continued reduced growth after a drought event, although these results were highly specific to the analyzed city. We further show that individual tree characteristics and environmental conditions significantly influence the growth of urban trees. Canopy openness and other aspects of the surrounding environment (water supply and open surface area of the tree pit), tree size, and tree species significantly affect urban tree growth and can modify the ability of trees to tolerate the drought stress in urban areas. Sustainable tree planting of well adapted tree species to their urban environment ensures healthy trees providing ecosystem services for a high quality of life in cities.

**Keywords:** drought tolerance; mixed models; standardized precipitation-evapotranspiration index; superposed epoch analysis; urban trees

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## 1. Introduction

Urban trees are of great value to a city: their performance and esthetics are beneficial to the climate and human population. By providing ecosystem services such as evaporative cooling [1] and shading [2], trees in an urban landscape are able to ameliorate negative effects of urban climates and climate change by reducing irradiances, and surface and air temperatures [3–5]. Furthermore, urban trees store carbon [6], reduce rainwater runoff [7], and filter pollutants [8]. The moderation of microclimates and improvement of environmental conditions by urban trees has been analyzed by several studies [9–13], with results highlighting the importance of trees for cities and the city climate. Therefore, urban forests can also become a key component to the adaptation of cities to climate change [14]. However, healthy and well growing trees provide the greatest benefits.

In addition to the functions and effects of trees on the climate of a city, tree growth and site conditions are closely related to the services that trees can provide. Cities are characterized by varying growth conditions that highly influence the growth, resilience, and mortality of trees [15–17]. Urban environments can be stressful habitats for trees [18], with hindering growth conditions such as restricted water availability [19], restricted soil volume [20], de-icing salt in winter [21], mechanical injury [22], and insect infestation [23]. Climate change with an associated higher frequency of years of extreme weather events [24] will expose urban trees to even more restrictive growth conditions. Drought years can cause high stress levels for trees with excessive evaporative demands [25] and less photosynthesis. Frequent drought years with short time periods in between may result in less time for tree recovery. High water stress can induce a change in tree growth with modified tree allometry [26,27], overall reduced growth [28], and tree die-off [29,30]. Tree species with a higher drought tolerance are expected to be more resistant and are therefore advantageous to altered climates than those with less tolerance. Moreover, the individual tree structure can additionally influence the drought tolerance of a tree, with higher crown volume increasing drought resistance [31] and higher age decreasing growth during drought episodes [32].

The annual growth patterns of urban trees can indicate their growth conditions and provide information regarding the climate influencing individual tree growth [33] and drought tolerance. Dendrochronology describes the study of tree rings with the aim of examining events through time recorded by tree-ring widths [34]; it can be used to interpret urban ecosystem dynamics and the impact of land use on trees as well as to analyze the climate-growth relationships of trees [33]. Tree-ring analysis of forest trees has been conducted in many studies [29,31,35–37], whereas studies on urban trees are relatively scarce [3,15,33]. Gillner *et al.* [38] and He *et al.* [39] pointed out that the effects of past climates on growth can provide valuable information on tree performance, including performance under climate change, using the growth patterns of urban trees based on dendrochronology. Knowledge regarding the drought tolerance and sensitivity of common urban tree species could assist urban space planners in selecting long-living and healthy trees, thereby providing sustainable ecosystem services, such as evaporative cooling, to mitigate the effects of a changing climate [38]. Dendrochronology allows a retrospective study of how tree species perform within urban climates and provides important information regarding the choice of optimal species for planting at a certain location.

The aims of this study were to analyze the annual growth rates and drought response of two common urban tree species, namely the small-leaved lime (*Tilia cordata* Mill. (*T. cordata*)) and the black locust (*Robinia pseudoacacia* L. (*R. pseudoacacia*)), in two cities in southern Germany with differing climates. Furthermore, the influence of individual tree structure and site conditions (canopy openness, distance to neighboring trees and buildings, and open surface area (OSA)) on annual tree-ring development was assessed. In more specific detail, we focused on the following questions:

- Can the average growth rates of the analyzed tree species be quantified in respect to their growing sites?
- How stable and sensitive is the growth of trees in urban environments?
- How does the urban climate (temperature and precipitation) and environment (light, open surface, neighboring trees, and close buildings) influence tree growth?
- What are the responses of urban tree species with varying drought tolerances to drought years?
- Do the urban environment and individual tree structure modify the drought stress of trees?

## 2. Materials and Methods

### 2.1. Site Description and Data Collection

Increment cores were collected in München (48°09′ N, 11°35′ E, 519 m above sea level (a.s.l.)) and Würzburg (49°48′ N, 9°56′ E, 177 m a.s.l.), two major cities in southern Germany. As shown in Figure 1, the climatic characteristics of the two cities differ. The long-term annual precipitation

values of München and Würzburg are 959 and 596 mm, respectively [40], whereas the mean annual temperature (1961–1990) of both cities is 9.4 °C [40].
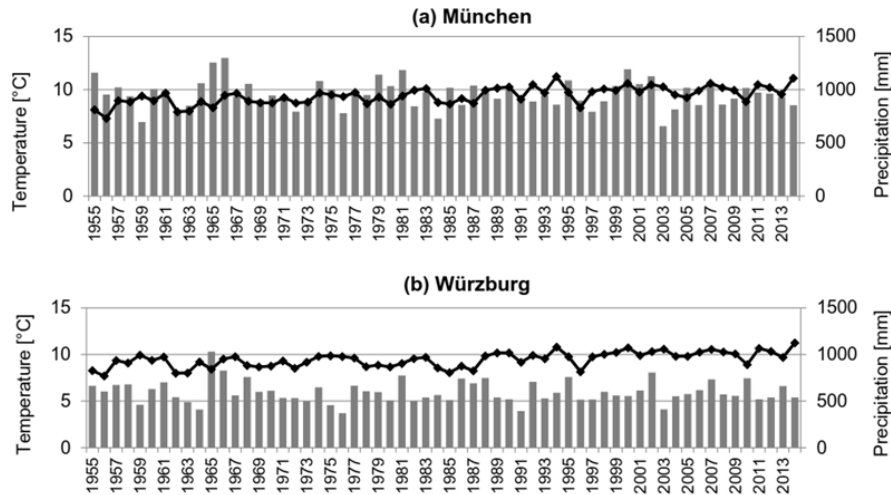


**Figure 1.** Climate graphs of München (**a**) and Würzburg (**b**) from 1955 to 2014 (data source: DWD [40]). Black dots and line represent the mean annual temperature in °C and the gray bars represent the annual precipitation in mm.

For this study, two common urban tree species were selected: *T. cordata* and *R. pseudoacacia*. These two species were selected as they are most relevant common urban tree species in Germany and markedly differ in their ecological features [41]. Whereas *T. cordata* is shade tolerant and moderately drought tolerant [42,43], *R. pseudoacacia* requires a certain amount of light (shade intolerant) and is very tolerant to drought as well as fast growing [43–46].

Tree selection was based on visual impression, that is, damaged, pruned, or low-forking trees were excluded. Increment cores were collected from June 2014 to September 2014. Altogether, increment cores were taken from 68 individual *T. cordata* trees and 62 individual *R. pseudoacacia* trees. All trees were located in street canyons, parks, and public squares and were randomly distributed in both cities. Two cores perpendicular to each other were extracted per tree at a height of 1.3 m, in a northern and eastern direction. Due to the main wind direction (southwest), coring from north and east yields more representative tree ring widths [29].

Moreover, stem diameter (dbh) at a height of 1.3 m, tree height, open surface area of the tree pit (OSA) of all trees, and distance to neighboring trees as well as distance to adjacent buildings were recorded [47]. Hemispheric photographs of the tree crowns were taken using a Nikon Coolpix P5100 camera with a fisheye lens and Mid-OMount. The resulting hemispherical photos were analyzed using WinSCANOPY (Régent Instruments Inc., Ville de Québec, Canada) to derive the canopy openness (percentage of open sky visible, degree of development, equivalent to the sky view factor SVF) of every individual tree [48,49].

*2.2. Quantification of Urban Tree Growth in Relation to Growing Site Based on Tree Ring Analysis*

All cores were mounted on grooved boards with glue and sanded using progressively finer sand papers. The first sanding was applied to flatten the cores, whereas the subsequent sanding episodes polished the cores for better visualization of the cross-sectional area [34]. The annual tree-ring widths

of the cores were measured using a Lintab digital positioning table with a resolution of 1/100 mm [50]. For cross-dating of the time-series, the software package TSAP-Win [51] was used.

Further analyses were carried out in R [52] using package dplR [53]. With dplR, all tree-ring series were indexed using a double detrending process: first, modified negative exponential curves were applied followed by cubic smoothing splines (20 years rigidity, 50% wavelength cutoff). The detrending was conducted to remove low frequency trends, which are age associated [38,54]. The resulting detrended series were averaged using Tukey's biweight robust mean to build chronologies for both species and each city. As a result of detrending, standardized chronologies with a yearly ring width index (RWI) averaging around 1 were obtained. Values smaller than 1 indicate growth below normal, while values greater than 1 indicate growth higher than normal. Further, for chronology building, the autocorrelation of every individual series was removed using autoregressive models with a maximum order of three. This procedure of detrending ensured a removal of all long-term growth trends, thereby obtaining a chronology containing only tree ring variability with climate fluctuations [3,55]. The statistical validity of the chronologies was assessed using the expressed population signal (EPS) for the common period of the time series of all analyzed tree individuals.

### 2.3. Investigated Variables

Linear regressions and linear mixed models were used to assess the influence of structural and environmental variables on the annual growth rates (response variable). Tested explanatory variables were the analyzed species (*R. pseudoacacia* and *T. cordata*), the sampling city (München and Würzburg), the growing site (park, public square, or street canyon), the stability of growth (Equation (1)), the sensitivity of growth in relation to environment (see below), the dbh, the tree height, the estimated age (Equations (2) and (3)), estimation of vitality by Roloff [56], the OSA (Equation (4)), the distance to neighboring trees (Equation (5)), the distance to adjacent buildings (Equation (6)), and the canopy openness. According to Roloff [56] the vitality was rated from very good (0) to very poor (3) conditions regarding the branching structure of the crown. Stability of growth was computed following Jucker *et al.* [57] by

$$\text{Stability} = \text{Average growth rate per tree/standard deviation of growth} \tag{1}$$

with values ranging from 0 to 10, whereby low values indicate less stability and high values high stability.

The average growth rates and sensitivity of each tree were derived using the R package dplR. The mean sensitivity describes the year-to-year variability of tree ring data in relation to the previous year ring width [58]. The ages of *T. cordata* trees were calculated by the formula of Lukaszkiewicz and Kosmala [59]:

$$\text{age} = a + \exp^{\left(b + c \times \frac{\text{dbh}}{100} + d \times \text{tree height}\right)} \tag{2}$$

where $a = 264.073$, $b = 5.5834$, $c = 0.3397$, and $d = 0.0026$; dbh in cm and tree height in m ([47]).

To estimate the age of *R. pseudoacacia* we applied a species-dependent age factor of 0.996, which was computed by the measurements of Dwyer [60] for honey locust (*Gleditsia triacanthos*):

$$\text{age} = 0.996 \times \text{dbh} \tag{3}$$

The open surface area (OSA) of the tree pit is calculated by

$$\text{OSA (m}^2) = (\sqrt{(r_N^2 + r_{NE}^2 + \ldots + r_{NW}^2)/8}) \tag{4}$$

where $r_N$ is the length of the visible open surface in the northern direction and $r_{NW}$ the length of the visible open surface in the northwest direction.

Mean distances to neighboring trees $d_t$ and adjacent buildings $d_b$ were computed as follows:

$$d_t(m) = \sqrt{(((t_{d_N} - r_N)^2 + (t_{d_{NW}} - r_{NW})^2)/8)} \tag{5}$$

$$d_b(m) = \sqrt{(((b_{d_N} - r_N)^2 + (b_{d_{NW}} - r_{NW})^2)/8)} \tag{6}$$

where $t_{d_N}$ is the distance to neighboring trees in the northern direction, $t_{d_{NW}}$ is the distance to neighboring trees in the northwest direction, $b_{d_N}$ is the distance to adjacent buildings in the northern direction, $b_{d_{NW}}$ is the distance to adjacent buildings in the northwest direction, $r_N$ is the maximum crown extension in the northern direction, and $r_{NW}$ the maximum crown extension in the northwest direction.

To analyze the effects of climate in terms of the water supply on tree growth, we calculated a monthly, multiscalar climatic drought index, the SPEI [61]. The SPEI uses precipitation and potential evapotranspiration (PET) as input data, whereby PET was calculated according to the Hargreaves approach [62,63]. The time scale was set to 4 months, with a Gaussian kernel to consider the water supply, and a log-logistic distribution was applied [61]. Using the calculated SPEI, the influence of the identified drought years on the growth patterns of urban trees was investigated. A yearly SPEI for 1955–2013 was computed by averaging the monthly index values of the growing season from April to September. Years with an SPEI smaller than −1 were classified as moderate drought years, whereas years with an SPEI smaller than −2 were interpreted as years with severe droughts [61,64].

### 2.4. Statistical Analyses

The stability of growth was assessed with linear regressions following Pretzsch *et al.* [26]:

$$y = b \times x^a \text{ or } = \ln(a) + b \times \ln(x) \tag{7}$$

where $y$ is the response variable, $x$ is the explanatory variable, $a$ is the intercept, and $b$ is the slope.

Further, using the R package nlme [65], linear mixed models of the following form were developed to estimate the influence of climate, environment, and tree structure (explanatory variables) on the annual tree growth (response variable) derived by increment cores:

$$\text{Growth rate}_{ij} = \beta_1 \times x_{1ij} + \ldots + \beta_n \times x_{nij} + b_{i1} \times z_{1ij} + \ldots + b_{in} \times z_{nij} + \varepsilon_{ij} \tag{8}$$

where the growth rate is the response variable for the $j$th of $n_i$ observations in the $i$th of $M$ groups or clusters, $\beta_1, \ldots, \beta_n$ are the fixed-effect coefficients, which are identical for all groups, $x_{1ij}, \ldots, x_{nij}$ are the fixed-effect regressors for observation $j$ in group $i$; the first regressor is usually for the constant, $x_{1ij-1}, b_{i1}, \ldots, b_{in}$ are the random-effect coefficients for group $i$, $z_{1ij}, \ldots, z_{nij}$ are the random-effect regressors, and $\varepsilon_{ij}$ is the error for observation $j$ in group $i$.

The derived annual growth rates over the past twenty years (1994−2013) for each tree were used as input for the response variable. The individual tree number and the species were set as random effects. The back-calculated dbh, the water supply (measured with SPEI), the distance to neighboring trees, the distance to adjacent buildings, the OSA, the vitality, and the canopy openness were used as fixed effects along with the species (*R. pseudoacacia*, *T. cordata*), city (Würzburg, München), and growing site (park, public square, and street canyon). In addition, all fixed effects were tested with interactions. Nonsignificant terms were gradually removed from the models. The models with significant *p*-values for the fixed effects and the overall lowest Akaike's information criterion (AIC) were chosen as final models.

To investigate the influence of the urban environment and tree structure on growth during low growth episodes and episodes with extraordinarily high growth, another two linear mixed models were calculated. First, we investigated all years with a ring width index of <1 (detrended values

averaged around 1). Second, all years with a ring width index of >1 were analyzed in relation to the tree structure and environment as explained earlier.

Extreme drought years, such as 2003, can have a great impact on the growth of trees [31]. The effects of extreme droughts on tree growth patterns may persist several years after the drought. Therefore, we analyzed the influence of single years on the tree growth using superposed epoch analysis (SEA). SEA investigates the significance of a mean tree growth response to certain events (such as droughts) to pre- and post-drought growth periods [53]. According to Lough and Fritts [66], Orwig and Abrams [36], and Gillner *et al.* [38], deviations from the mean ring width index of each core were calculated for the following three periods. The growth of the 5 years prior to the analyzed drought year (pre-drought), of the drought year and of the 5 years after the drought year (post-drought) was averaged to detect significant departures between those superposed epochs. The SEA was computed using the R package dplR [53], using random sets of 11 years from 1000 bootstrapped sets [38,53].

## 3. Results

### 3.1. Quantification of Urban Tree Growth in Relation to Growing Site Based on Tree Ring Analysis

The average dbh of the analyzed trees was 44.5 cm in München and 44.3 cm in Würzburg for *R. pseudoacacia* and 34.2 cm in München and 33.1 cm in Würzburg for *T. cordata* with a maximum range between 11 cm and 102.2 cm (Table 1). The tree individuals chosen for coring therefore represented a broad size spectrum of urban trees. The annual growth rates of both species were similar in München and Würzburg, with *R. pseudoacacia* displaying a higher overall growth rate compared with *T. cordata* (3.9 and 4.0 mm·year$^{-1}$ *versus* 3.2 and 2.9 mm·year$^{-1}$, respectively). After crossdating and detrending, four tree ring chronologies were derived. The EPS values of all series varied between 0.85 and 0.90 for the common period of all tree ring series, thereby exceeding the required threshold of 0.85 [67].

**Table 1.** Statistical characteristics of the tree ring series of *Tilia cordata* and *Robinia pseudoacacia* in München and Würzburg.

| | *n* | dbh min [cm] | dbh Avg [cm] | dbh Max [cm] | Avg Tree Height [m] | Avg OSA [m²] | Avg Age [a] | Growth Rate ± SD [mm·Year$^{-1}$] | Mean Sensitivity | EPS [1] |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | *Robinia pseudoacacia* | | | | | |
| München | 30 | 14.0 | 44.5 | 101.9 | 15.7 | 102.39 | 44.3 | 3.9 ± 1.7 | 0.33 | 0.90 |
| Würzburg | 32 | 11.0 | 44.3 | 102.2 | 15.1 | 109.55 | 44.1 | 4.0 ± 1.9 | 0.34 | 0.85 |
| | | | | | *Tilia cordata* | | | | | |
| München | 37 | 12.0 | 34.2 | 86.7 | 13.1 | 146.48 | 42.6 | 3.2 ± 1.6 | 0.38 | 0.89 |
| Würzburg | 30 | 14.0 | 33.1 | 71.5 | 12.5 | 123.34 | 44.0 | 2.9 ± 1.6 | 0.39 | 0.89 |

[1] Based on the ring width index of tree ring chronologies for the common period, obtained by double detrending of the tree ring series. dbh = stem diameter; OSA = open surface area; EPS = expressed population signal; SD = standard deviation.

All tree ring series showed a similar mean sensitivity of 0.33 to 0.39 and were within the range given by Speer [34] (Table 1). The slightly higher sensitivity of *T. cordata* indicated a higher susceptibility to climatic variables and poor growth conditions [3,38,68,69].

In total, the chronologies of *R. pseudoacacia* covered the period from 1954 to 2013 in München and 1960 to 2013 in Würzburg, whereas the tree ring data of *T. cordata* ranged from 1941 to 2013 in München and 1910 to 2013 in Würzburg (Figure 2). Of the analyzed species and cities, *T. cordata* in Würzburg showed the longest tree ring series covering 103 years. The chronologies of *R. pseudoacacia* of München and Würzburg were both shorter than that of *T. cordata*, covering 59 years in München and 53 years in Würzburg.
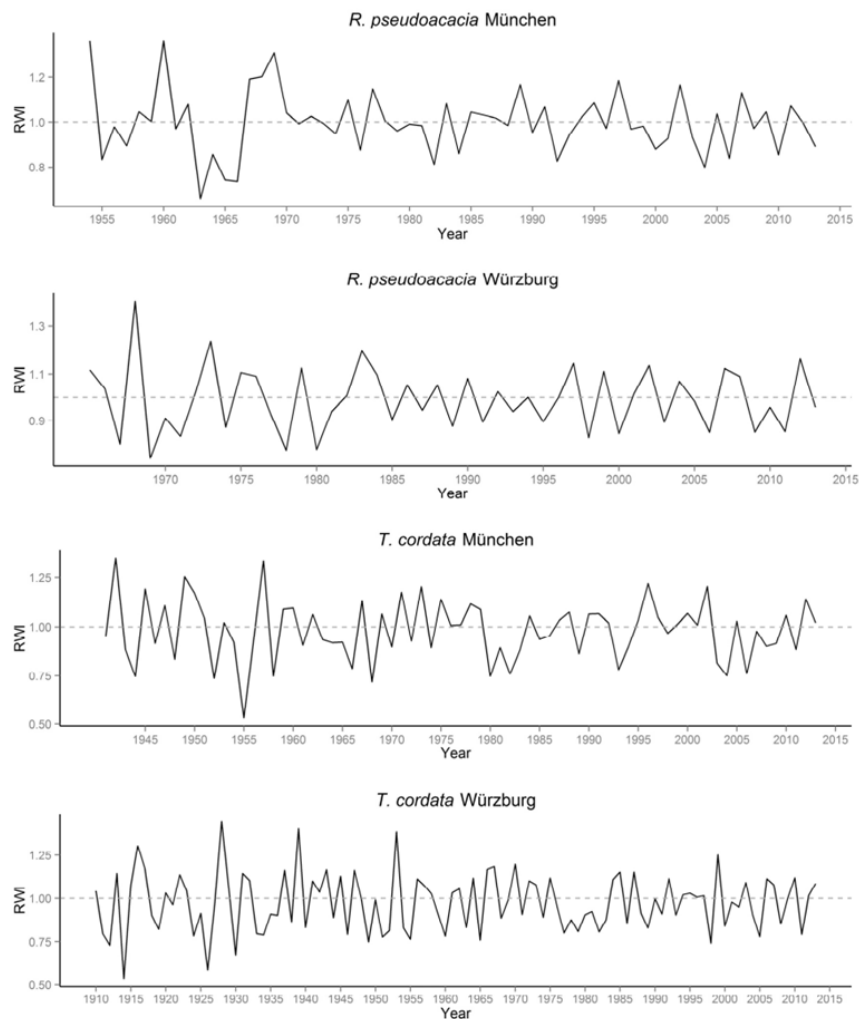
**Figure 2.** Ring width index of *Robinia pseudoacacia* and *Tilia cordata* in München and Würzburg after double detrending (negative exponential function and 2/3 cubic smoothing spline).

*3.2. Stability, Sensitivity, and Modeling of Tree Growth in Relation to Their Environment*

To reveal any species-specific reaction patterns and the effect of the sampled cities on the stability of growth, simple linear regressions were applied (Figure 3). With a *p*-value of 0.08, the relationship of stability with city was not significant, whereas there was a significant relationship of species with stability ($p = 0.001$, $r^2 = 0.1$); however, the low coefficient of determination indicated a very weak fit. The stability value between the species was significantly different, with *R. pseudoacacia* showing a higher stability on average as compared to *T. cordata*.
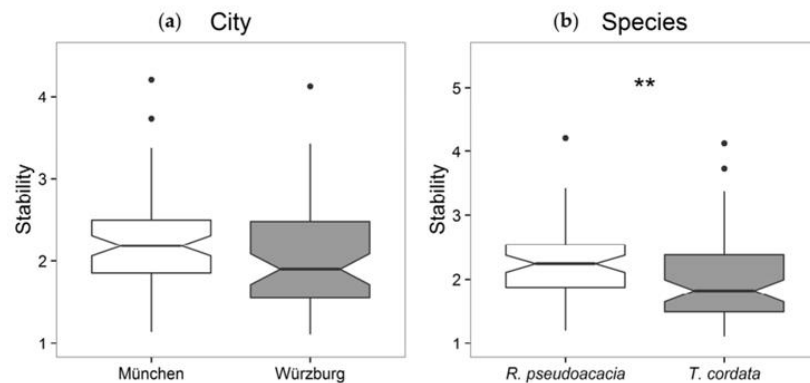
**Figure 3.** Boxplots showing the relationships of growth stability with city (**a**) and species (**b**). Significances are marked by asterisks.

The regressions of the further analyzed variables tree height, vitality, OSA, distance to adjacent buildings, and trees as well as the canopy openness resulted in non-significant *p*-values. Significant relationships were only found for sensitivity, dbh, and age (Table 2).

**Table 2.** Results of linear regressions with stability as response variable and the variables sensitivity, dbh, and age as individually tested explanatory variables for the stability of growth of *Tilia cordata* and *Robinia pseudoacacia* in München and Würzburg (equation: response variable = $a + b \times$ explanatory variable). The table lists the regression coefficients ($a$, $b$), coefficients of determination ($r^2$), RSE for bias correction, *F*-values, *p*-values, and SEs of regression coefficients as well as the sample size ($n$).

| Response Variable | Explanatory Variable | $n$ | $a \pm SE$ | $b \pm SE$ | $r^2$ | RSE | F | p |
|---|---|---|---|---|---|---|---|---|
| ln(Stability) | ln(Sensitivity) | 128 | $0.14 \pm 0.09$ | $-0.54 \pm 0.08$ | 0.25 | 0.28 | 42.83 | <0.001 |
| Stability | Dbh | 128 | $1.68 \pm 0.15$ | $0.01 \pm 0.003$ | 0.09 | 0.77 | 12.85 | <0.001 |
| Stability | Age | 128 | $1.74 \pm 0.16$ | $0.01 \pm 0.003$ | 0.06 | 0.78 | 7.77 | 0.006 |

Dbh = stem diameter; RSE = residual standard error; SE = standard error.

For dbh and age, the $r^2$ values obtained were <0.1 with small *F*-values and therefore not suitable for explaining the variance in the stability of growth. In contrast, sensitivity displayed a higher $r^2$ and *F*-value. The $r^2$ of stability in relation to sensitivity and age strongly increased when adding the terms "species" and "city" as dummy variables to the regression (Figure 4).

As shown in Table 2 and Figure 4, stability decreased with increasing sensitivity. In contrast, stability increased with increasing age. To derive a final model best fitting the data, "city" and "species" were tested as explanatory variables. Therefore, a regression with both "species" and "city" added to explain stability on the basis of sensitivity was obtained. With an $r^2$ of 0.29 ($F = 17.37$, $p < 0.01$), the model resulted in the highest $r^2$ values and could explain most variance in the data of all computed models. The form of the final model was:

$$\text{ln(Stability)} = 0.18 - 0.48 \times \text{"Sensitivity")} + 0.11 \times \text{"Species"} - 0.09 \times \text{"City"} + \varepsilon \qquad (9)$$

The derived regression illustrates, again, that stability and sensitivity were opposing variables. Further, the analyzed individuals of *T. cordata* showed a higher sensitivity than those of *R. pseudoacacia* and the trees in München had a slightly higher stability than the trees in Würzburg.

Appendix
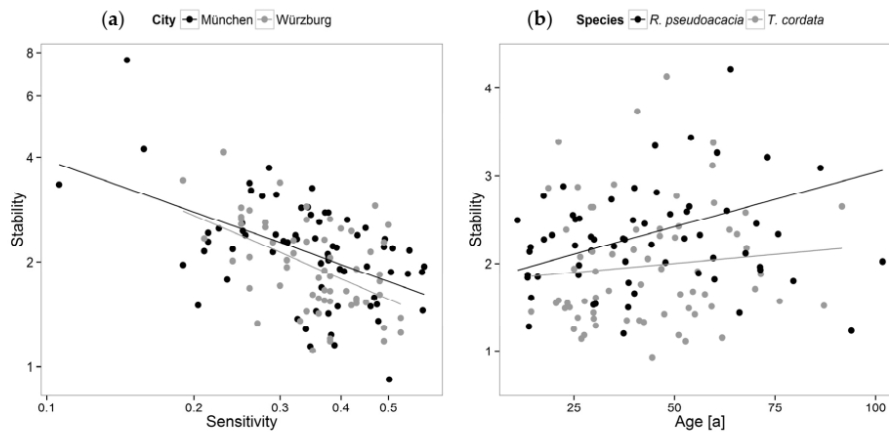
**Figure 4.** Regressions of stability with sensitivity (**a**) and stability with age (**b**). Highlighted are the regression lines of trees in München and Würzburg (a, ln(Stability) = 0.18 − 0.53 × ln(Sensitivity) − 0.08 × "City" + ε), and *Tilia cordata* as well as *Robinia pseudoacacia* (b, Sensitivity = 1.57 + 0.01 × age + 0.36 × "Species" + ε). For the regression of stability with sensitivity (b), both variables were log transformed.

Using the water supply (measured with SPEI), the growth of *R. pseudoacacia* and *T. cordata* before, during, and after certain drought years was investigated (Figure 5). In München, the SPEI of 2003 was −2.0, whereas the indices for 1998 and 1992 were −1.3 and −1.1, respectively. Several years showed an SPEI close to −1: 2004 (−0.96) and 1976 (−0.93). The drought years of Würzburg as indicated by the SPEI were 2003 (−1.6), 1976 (−1.5), and 1947 (−1.2). Moreover, 2012 (−1.1), 1993 (−1.1), and 1964 (−1.1) showed SPEI values smaller than −1. The SPEI additionally identified years with a high positive index, therefore revealing years with extraordinarily positive growth conditions. In München, those being 1979 (1.2), 1966 (1.5), 1965 (2.0), and 1955 (1.5). Positive SPEI values in Würzburg were found for 1968 (1.4), 1966 (1.2), and 1965 (1.65).

In the following step, linear mixed models of tree growth over the past 20 years in relation to site conditions were developed. Table 3 presents the statistical results of the final model including significant fixed effects and lowest AIC.

**(a)**



**Figure 5.** *Cont.*

**(b)**
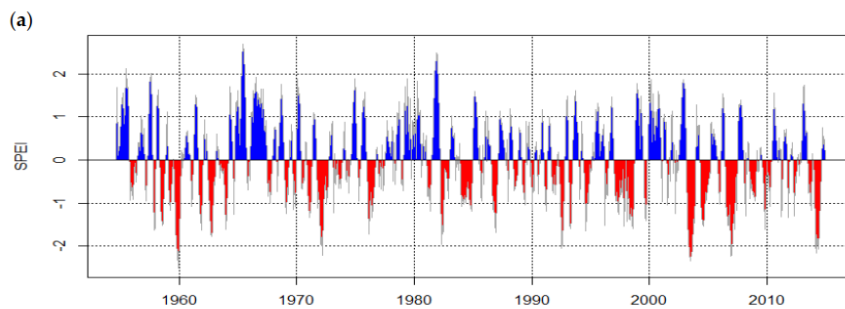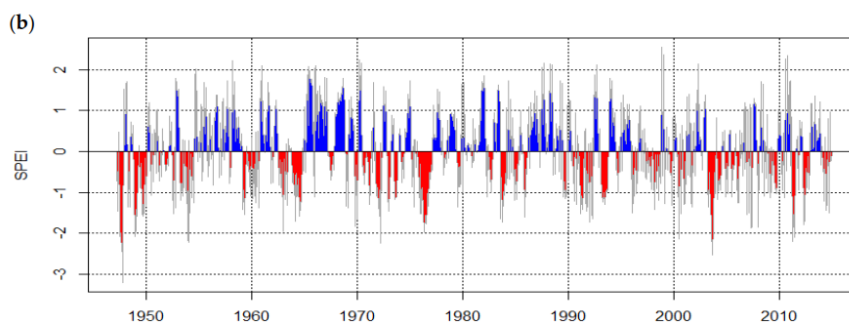


**Figure 5.** Calculated standard precipitation-evapotranspiration index (SPEI) for München (**a**) from 1955 to 2014 and Würzburg (**b**) from 1947 to 2014 with a time scale of four months. Blue-colored series represents positive SPEI values (>0) of years with a positive climatic water balance, and red-colored series represents negative SPEI (<0) values of a negative climatic water balance.

**Table 3.** Linear mixed model on the annual growth rate (mm·year$^{-1}$) during the last 20 years (1994–2013) of all analyzed trees (response variable) with the individual tree as random effect and a random slope added for the effect of dbh, and fixed effects dbh, canopy openness, OSA, species, and water supply (measured with SPEI) of the form: Growth rate$_{ij}$ = $\beta_1 \times x_{1ij} + \ldots + \beta_n \times x_{nij} + b_{i1} \times z_{1ij} + \ldots + b_{in} \times z_{nij} + \varepsilon_{ij}$ where the growth rate is the response variable for the *j*th of $n_i$ observations in the *i*th of *M* groups or clusters, $\beta_1, \ldots, \beta_n$ are the fixed-effect coefficients, which are identical for all groups, $x_{1ij}, \ldots, x_{nij}$ are the fixed-effect regressors for observation *j* in group *i*; the first regressor is usually for the constant, $x_{1ij-1}, b_{i1}, \ldots b_{in}$ are the random-effect coefficients for group *i*, $z_{1ij}, \ldots, z_{nij}$ are the random-effect regressors, and $\varepsilon_{ij}$ is the error for observation *j* in group *i*.

| | Value ± SE | $p$ |
|---|---|---|
| Intercept | $8.46 \pm 1.74$ | <0.001 |
| Dbh | $-0.27 \pm 0.03$ | <0.001 |
| Canopy openness | $0.10 \pm 0.03$ | <0.001 |
| OSA | $-0.02 \pm 0.01$ | 0.03 |
| Species | $-0.33 \pm 1.90$ | 0.86 |
| Water supply (SPEI) | $0.18 \pm 0.04$ | <0.001 |
| Dbh:OSA | $0.001 \pm 2.1 \times 10^{-4}$ | <0.001 |
| Canopy openness:Species | $0.09 \pm 0.04$ | 0.03 |
| SD Intercept | 5.72 | - |
| SD Dbh | 0.19 | - |
| $\varepsilon$ | 1.18 | - |

Dbh = stem diameter; OSA = open surface of the ground; SPEI = standardized precipitation-evapotranspiration index; SD = standard deviation; SE = standard error. Levels of species: 2 (*Robinia pseudoacacia* and *Tilia cordata*).

Of all tested variables, the dbh, the canopy openness, the OSA, and the water supply (measured with SPEI) proved to be highly significant variables, whereas the vitality, the city, the distance to neighboring trees and adjacent buildings, and the growing site (parks, public squares, and street canyons) had no marked effect and were therefore removed from the final model (Table 3). The variable species was significantly influencing tree growth especially in interaction with the canopy openness. The model showed that an increasing dbh had a strong negative influence on the growth rate, highlighting the age trend of tree growth. A higher canopy openness, in contrast, was beneficial for the growth of the analyzed urban trees; however a smaller OSA was more positive for tree growth. Furthermore, a higher water supply (measured with SPEI) increased the annual growth rate of the analyzed trees. The variable species was non-significant in the final model; however, this variable displayed a strong influence on tree growth in interactions with the canopy openness. According to

the model, *R. pseudoacacia* displayed a higher growth with a higher canopy openness and consequently higher light availability. *T. cordata* was less influenced in its growth by the available light. Those results reflect the individual shade tolerance of both species, since *R. pseudoacacia* is characterized as a light-demanding species, whereas *T. cordata* is very shade-tolerant.

Another significant interaction was dbh with OSA; younger trees with a smaller OSA displayed growth reductions, while bigger, older trees with a larger OSA showed increased growth.

The model highlighted how the growth of the analyzed urban tree species was influenced by the local climate (in terms of the water supply (SPEI) calculated with the potential evapotranspiration), the individual tree structure (dbh), and the surrounding environment (OSA), whereas the variable species was highly significant in relation with another explanatory variable (the canopy openness, representing the light availability). Other variables such as the distance to adjacent buildings and the city were of less importance for growth.

### 3.3. Growth of Urban Trees Under Drought Stress in Relation to the Tree Structure and Environment

With the obtained SPEI of the drought years, a drought year analysis (SEA) was performed to investigate the different drought strategies of both tree species (Figures 6 and 7).
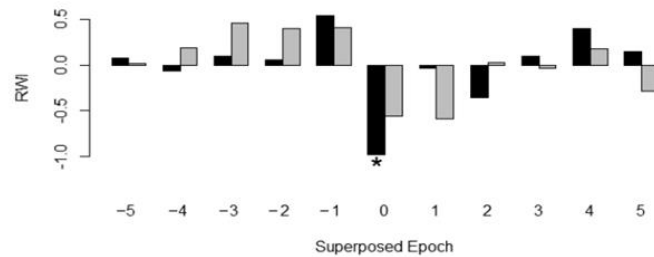


**Figure 6.** Drought year analysis (superposed epoch analysis) of the ring width index (RWI) during drought years (0), pre-drought (−5 to −1) and after drought (1−5) for *Robinia pseudoacacia* (black) and *Tilia cordata* (gray) in München. Input drought years are 2004, 2003, 1998, 1992, 1982, and 1976 for *R. pseudoacacia* and 2004, 2003, 1998, 1992, 1982, and 1976 for *T. cordata*. Marked columns (asterisk) represent a departure that is greater than would have occurred randomly as determined from 1000 bootstrap simulations at $p < 0.05$.
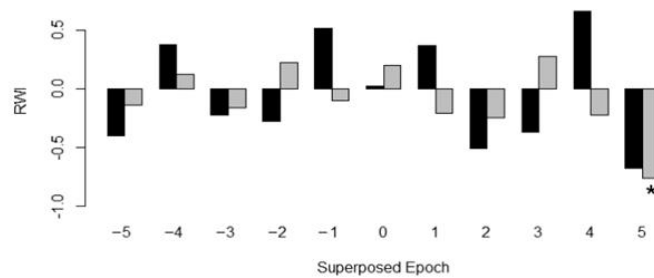


**Figure 7.** Drought year analysis (superposed epoch analysis) of the ring width index (RWI) during drought years (0), pre-drought (−5 to −1) and after drought (1−5) for *Robinia pseudoacacia* (black) and *Tilia cordata* (gray) in Würzburg. Input drought years are 2012, 2003, 1993, 1976, and 1964 for *R. pseudoacacia* and 2012, 2003, 1976, and 1947 for *T. cordata*. Marked columns (asterisk) represent a departure that is greater than would have occurred randomly as determined from 1000 bootstrap simulations at $p < 0.05$.

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Both species showed positive growth rates for most of the five pre-drought years in München and a decrease in growth during the drought year. The decline in growth was significant (asterisk) for *R. pseudoacacia* but not for *T. cordata*, which exhibited the highest growth decrease in the first year after drought. *R. pseudoacacia* could recover very quickly from the drought and reached nearly the former growth during the first year after the drought event. In the third year after a drought, *R. pseudoacacia* again had a positive average ring width index. *T. cordata* had recovered from the drought during the second post-drought year.

The growth patterns of both species in Würzburg were completely different compared with München (Figure 7). Although there were several growth reductions and positive ring width index values during the pre-drought years, during the drought years no decline in growth was observable for both species. *T. cordata* showed a reduction in the average ring width index value during the first and second years following the drought. However, a significant growth decline occurred five years after the analyzed drought years. *R. pseudoacacia* displayed no growth decrease during the first year after the drought but showed a very quick recovery during the fourth year. During the fifth year, *R. pseudoacacia* additionally exhibited a severe growth reduction.

To analyze the influence of the urban environment and tree structure on growth during drought years, we investigated all years with an RWI of <1 (Table 4).

**Table 4.** Linear mixed model on ring width index (RWI) development of all years with an index < 1 (growth lower than normal, drought years) of all analyzed trees as the response variable with the individual tree code as a random effect and a random slope added for the effect of dbh, and fixed effects water supply (measured with SPEI), canopy openness, analyzed species, growing city, and dbh of the form: $RWI_{ij} = \beta_1 \times x_{1ij} + \ldots + \beta_n \times x_{nij} + b_{i1} \times z_{1ij} + \ldots + b_{in} \times z_{nij} + \varepsilon_{ij}$ where RWI is the response variable for the *j*th of $n_i$ observations in the *i*th of *M* groups or clusters, $\beta_1, \ldots, \beta_n$ are the fixed-effect coefficients, which are identical for all groups, $x_{1ij}, \ldots, x_{nij}$ are the fixed-effect regressors for observation *j* in group *i*; the first regressor is usually for the constant, $x_{1ij-1}, b_{i1}, \ldots b_{in}$ are the random-effect coefficients for group *i*, $z_{1ij}, \ldots, z_{nij}$ are the random-effect regressors, and $\varepsilon_{ij}$ is the error for observation *j* in group *i*.

|  | Value $\pm$ SE | *p* |
|---|---|---|
| Intercept | $0.82 \pm 0.04$ | <0.001 |
| Water supply (SPEI) | $0.03 \pm 0.01$ | <0.001 |
| City | $-0.01 \pm 0.01$ | 0.67 |
| Species | $-0.05 \pm 0.01$ | <0.001 |
| Dbh | $-0.001 \pm 7.4 \times 10^{-4}$ | 0.27 |
| Canopy openness | $-0.002 \pm 7.4 \times 10^{-4}$ | 0.03 |
| Water supply (SPEI): City | $-0.03 \pm 0.01$ | 0.009 |
| Canopy openness: Dbh | $0.0001 \pm 1.7 \times 10^{-5}$ | 0.002 |
| SD Intercept | 0.09 | - |
| SD Dbh | 0.001 | - |
| $\varepsilon$ | 0.17 | - |

RWI = ring width index, dbh = stem diameter; SPEI = standardized precipitation-evapotranspiration index; SD = standard deviation; SE = standard error. Levels of species: 2 (*Robinia pseudoacacia* and *Tilia cordata*), levels of city: 2 (Würzburg and München).

The factors driving growth during drought years (Table 4) were partially altered compared with the model for the overall growth (Table 3). In the previous model on the annual growth rates over the past 20 years, dbh, the canopy openness, the OSA, and the water supply (measured with SPEI) as single terms as well as the interaction terms canopy openness with the species and the OSA with dbh were significantly affecting tree growth. During drought years (Table 4), the surrounding climate in terms of the water supply (SPEI) and the species proved to be highly influential for tree growth. Contrary to the overall growth, the dbh was not significantly affecting the growth patterns. The stand climate, especially in interaction with the city, was of most influence. Würzburg, with far less precipitation than

München resulted in more frequent drought years, and was therefore more inhibiting for tree growth than München. This result was compliant with the overall study design, as the trees in Würzburg exhibit an even higher effect of drought years on growth. Further, the model highlighted the better growth of *R. pseudoacacia* during drought, displaying less growth reductions than *T. cordata* in years with poor growth conditions. The canopy openness, representing the available light conditions, once again had a pronounced effect on growth, but reversed to the previous model. The influence of light availability on growth was additionally dependent on size and age; less canopy openness inhibited the growth of younger trees to a stronger degree than those of older, bigger trees.

Since we used detrended values with a removed age trend for the drought model, the variable dbh alone did not significantly affect growth in contrast to the previous overall growth model we ran. Further, the OSA did not affect growth during drought years. The growth in drought years appeared to be influenced by overall environmental factors including the water supply (SPEI), the species, the growing city, and the available light, and was less significantly influenced by the tree structure (age and dbh) and surroundings such as the OSA, neighboring trees, and buildings.

To reveal the differences in the growth patterns during drought years (low growth episodes) and favorable years (high growth periods), we ran an additional model using a ring width index greater than 1 (Table 5). In contrast to the model with a ring width index of smaller than 1 (Table 4), the canopy openness positively affected growth, which was in accordance with the overall linear mixed model on tree growth (Table 3). Moreover, during years with extraordinarily good growing conditions, *T. cordata* could display higher growth than *R. pseudoacacia*. Converse to the model of years with poor growth, younger trees with a higher light availability obtained higher growth rates. Other factors such as the distance to neighboring trees as well as the OSA were of less importance for growth.

**Table 5.** Linear mixed model on ring width index (RWI) development of all years with an index > 1 (growth higher than normal, favorable years) of all analyzed trees as the response variable with the individual tree code as a random effect and a random slope added for the effect of dbh, and fixed effects canopy openness, analyzed species, and dbh of the form: $RWI_{ij} = \beta_1 \times x_{1ij} + \ldots + \beta_n \times x_{nij} + b_{i1} \times z_{1ij} + \ldots + b_{in} \times z_{nij} + \varepsilon_{ij}$ where RWI is the response variable for the *j*th of $n_i$ observations in the *i*th of *M* groups or clusters, $\beta_1, \ldots, \beta_n$ are the fixed-effect coefficients, which are identical for all groups, $x_{1ij}, \ldots, x_{nij}$ are the fixed-effect regressors for observation *j* in group *i*; the first regressor is usually for the constant, $x_{1ij-1}, b_{i1}, \ldots b_{in}$ are the random-effect coefficients for group *i*, $z_{1ij}, \ldots, z_{nij}$ are the random-effect regressors, and $\varepsilon_{ij}$ is the error for observation *j* in group *i*.

|  | Value $\pm$ SE | *p* |
|---|---|---|
| Intercept | $1.24 \pm 0.04$ | <0.001 |
| Canopy openness | $0.002 \pm 8.9 \times 10^{-4}$ | 0.01 |
| Species | $0.05 \pm 0.02$ | 0.001 |
| Dbh | $0.001 \pm 7.5 \times 10^{-4}$ | 0.30 |
| Canopy openness: Dbh | $-0.0001 \pm 1.8 \times 10^{-5}$ | 0.003 |
| SD Intercept | 0.11 | - |
| SD Dbh | 0.001 | - |
| $\varepsilon$ | 0.22 | - |

RWI = ring width index; dbh = stem diameter; SD = standard deviation; SE = standard error. Levels of species: 2 (*Robinia pseudoacacia* and *Tilia cordata*).

## 4. Discussion

### 4.1. Tree Ring Analysis: Quantification of Growth of Urban Trees in Relation to the Growing Site

Although numerous studies regarding the drought tolerance and resilience of forest trees in relation to their site conditions exist [29,31,37,70–72], the behavior of urban trees to drought is rarely researched [33,38,39]. However, urban trees are of great benefit to the climate of a city and the well-being of its citizens. The current study points out a high variability in the growth of urban

trees. Since urban trees experience increased stress situations at their growing sites, for example a stronger limitation of water availability, higher temperatures, limited space, and a high variation in their growing conditions, the variability in growth indicates a high adaptability of urban trees to their environment. As expected from the ecological characteristics of both species, *R. pseudoacacia* showed a higher annual growth rate than *T. cordata* [44,47].

München and Würzburg were chosen as study sites since they differ in regard to climate characteristics; with 300 mm· year$^{-1}$ less precipitation Würzburg is considerably drier than München. Minor growth differences in relation to the differing climates of München and Würzburg were observable for *T. cordata*; this species showed a slightly higher growth in München than in Würzburg. This is not surprising regarding the low drought tolerance of the species, whereas *R. pseudoacacia* had similar growth rates in both cities. *R. pseudoacacia* is considered as a drought tolerant species [43]; therefore, the individuals in Würzburg are expected to show improved adaptation to a drier climate compared with the less drought-tolerant species *T. cordata*. The measured growth based on tree ring data of both analyzed species was similar to those reported in other dendrochronological studies with regard to urban trees. *T. cordata* had an average growth rate of 3.05 mm· year$^{-1}$, which is higher than the growth found by Gillner [73] in Dresden (1.6 and 2.43 mm· year$^{-1}$ for two different streets). Iakovoglou *et al.* [18] found that for the honey locust (*G. triacanthos*), growth rates were approximately 4.8 mm· year$^{-1}$ in the United States Midwest, whereas *R. pseudoacacia* in München and Würzburg exhibited a growth of 3.95 mm· year$^{-1}$.

### 4.2. Stability, Sensitivity, and Modeling of Tree Growth in Relation to Their Environment

The stability of growth was similar in both cities, but a difference regarding species was found. *R. pseudoacacia* had a significantly higher stability than *T. cordata*. This is consistent with the slightly higher sensitivity of *T. cordata* than that of *R. pseudoacacia*, which can be explained by a higher vulnerability to climate and poor sites [3,38,68,69] Therefore, the sensitivity proved to be inversely related to the stability of growth; however, stability increased with age. With higher age an equilibrium between growth and the environmental influences is achieved, which leads to a stabilization of the growth rates over time, reducing the variation in growth [57]. This pattern is similar to the age trend of trees.

Trees growing in an urban landscape are affected by various environmental factors and site conditions, which influence the annual growth rate aside from their structure and species characteristics [18–20]. We could prove using a linear mixed model approach that in particular the available light, dbh, OSA, and water supply (measured with SPEI) influenced the growth of the analyzed tree species. Light is one of the driving forces of growth [74,75], not only in forests but also in urban areas. Trees growing solitarily without neighboring higher trees and shading buildings can reach their growth maximum [76,77]. These findings are strongly dependent on the species, with *T. cordata* being less affected by shade as a shade-tolerant species in comparison with *R. pseudoacacia*, a light-demanding species experiencing growth deficits when growing in shade. Furthermore, the analyzed growth over the last 20 years was not significantly affected by the city although Würzburg is characterized by 300 mm· year$^{-1}$ less precipitation than München. The growing site with its local climate and individual tree characteristics are more important for tree growth than the overall growing city. A negative or positive city effect was therefore not observable.

### 4.3. Growth of Urban Trees under Drought Stress in Relation to the Tree Structure and Environment

The results of the drought year analysis (SEA) for München could not confirm the high drought resistance of *R. pseudoacacia*, which exhibited a significant growth reduction during the drought year, but recovered very quickly from the drought, proving a high drought resilience. When examining the original distribution of *R. pseudoacacia*, those results are not surprising. The original distribution of this species is the Atlantic North America at sites with yearly precipitation rates of 1000–1500 mm [44]. Roloff *et al.* [43] classified *R. pseudoacacia* as well suited for dry sites but not fit for more humid sites.

The former distribution of this species suggests that *R. pseudoacacia* can additionally grow in regions of very high precipitation rates and is not as drought tolerant as expected. However, a fast recovery from the drought events highlights a high resilience to drought. *T. cordata* had the highest growth decrease during the first year after the drought event, followed by a quick recovery. Those results are similar to the results of Roloff *et al.* [43] and Gillner *et al.* [38] for two *Acer* species and confirm the classification of *T. cordata* as a moderate drought tolerant-species.

The drought year analysis (SEA) could prove that the studied species react to drought events with different water management strategies. Anisohydric species such as *Quercus rubra* close their stomata very slowly after a drought event; they show high fluctuations in their water management [78,79]. In contrast, isohydric species such as *Picea abies* react immediately to drought by closing their stomata; therefore, they exhibit fewer disruptions in their water management. *Fagus sylvatica* shows an intermediate water management type, showing similar patterns to *T. cordata* [78,79]. Although *T. cordata* reacted slowly, with growth reductions visible 1–2 years after the drought, *R. pseudoacacia* showed an immediate growth decrease during drought, with a fast recovery in München. Those species traits indicate that *T. cordata* is a more anisohydric species, whereas *R. pseudoacacia* displays characteristics more indicative of an isohydric species. This is in line with the findings of Peters *et al.* [80], who characterized ring-porous species such as *R. pseudoacacia* as having a higher capacity for regulation of their stomata as protection against drought, whereas diffuse-porous species such as *T. cordata* experience higher rates of water loss during summer [81].

Trees sampled in Würzburg showed no clear patterns regarding growth during and after drought. During the drought year, both species displayed no growth decline, whereas *R. pseudoacacia* showed a distinct growth reduction two years after the drought and *T. cordata* during the first- and second-year post-drought year. In particular, *R. pseudoacacia* displayed rapid recovery after the growth reduction; however, both species exhibited the highest drop in growth five years after the analyzed drought events (significant for *T. cordata*), which was presumably caused by an additional drought year. The SPEI of Würzburg revealed more frequent drought years than in München. Furthermore, the frequency of drought years in Würzburg is very high; after approximately five years, an additional year with an SPEI of $<-0.5$ occurs, adding further stress to the trees, which have only a very short time to recover from the previous drought. Moreover, Würzburg is characterized by extreme growth conditions with frequent late frosts and an overall low precipitation rate. Altogether, the factors driving the growth of urban trees in Würzburg are more complex compared to München, and tree growth can be influenced by other factors besides the extreme climate, including different planting methods or soil material and different fertilization methods. In particular, the method and soil used for planting, along with soil compaction, may have a strong effect on water management and growth of urban trees [82,83].

### 4.4. Effect of Urban Environment and Individual Tree Structure on the Drought Stress of Trees

A linear mixed model describing the growth behavior of the analyzed trees during low-growth episodes revealed that under stress, light availability, species traits, and the water supply (measured with SPEI) of the city are the driving forces of growth. The found results are mainly related to the analyzed species characteristics (higher shade tolerance of *T. cordata*), and contrary to the model for the overall growth, the variable "city" (München and Würzburg) in relation to the water supply (SPEI) proved significant, showing that specific city effects like, e.g., less precipitation and smaller planting pit in Würzburg [84,85] can modify the growth and drought stress of urban trees. During years with high growth episodes, the water supply (measured with SPEI) and the city had no influence on growth. All other variables (species, dbh, and canopy openness) significantly affected growth, but with reversed effects. While during drought years, a smaller canopy openness diminished tree growth, during favorable years the effect of higher canopy openness was positive. Both models of growth during drought years and during favorable years revealed the driving forces of urban tree growth reacting opposing on growth dependent on the surrounding environmental conditions.

## 5. Conclusions

The study highlights the growth patterns of two common European urban tree species (*R. pseudoacacia* and *T. cordata*) and how their growth changed with age. Different strategies were found depending on the analyzed city and species, with the ring-porous *R. pseudoacacia* exhibiting immediate growth reductions followed by fast recovery, while the diffuse-porous *T. cordata* showed delayed growth reductions. Further, the influence of the surrounding environment on tree growth was analyzed. Hereby, light conditions, water supply, and the species were most important for tree growth, which proved to be detrimental for tree growth during drought. However, the effects of the urban environment such as higher temperatures, highly sealed sites, late frosts, and de-icing salting in winter on tree growth, in particular within the context of drought years and climate change, have rarely been analyzed up to date. Further studies regarding the driving forces of urban tree growth during drought in relation to their environment, structure, and ecophysiology, as well as in view of climate change are necessary. These further studies would help to gain valuable information on sustainable tree planting and tree performance for healthy, long-living trees, thereby providing ecosystem services such as evaporative cooling, shading, carbon sequestration, and pollutant filtering for increasing the quality of life of human populations in cites.

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## Abbreviations

The following abbreviations are used in this manuscript:

| | |
|---|---|
| AIC | Akaike's information criterion |
| Dbh | diameter at breast height |
| EPS | expressed population signal |
| OSA | open surface area of the tree pit |
| PET | potential evapotranspiration |
| RSE | residual standard error |
| RWI | ring width index |
| SD | standard deviation |
| SE | standard error |
| SEA | superposed epoch analysis |
| SPEI | standardized precipitation-evapotranspiration index |
| SVF | sky view factor |

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## 3. Paper III

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**ORIGINAL PAPER**

# Inter- and intraannual growth patterns of urban small-leaved lime (*Tilia cordata* mill.) at two public squares with contrasting microclimatic conditions

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**Abstract** The effects of urban conditions on tree growth have been investigated in an increasing number of studies over the last decades, emphasizing the harsh environment of cities. Urban trees often grow in highly paved, compacted sites with consequently less soil moisture, higher soil temperatures, and greater vapor pressure deficits. However, there is still a knowledge gap regarding the impact of harsh paved environments on urban tree growth during drought years on the growth patterns of urban trees. The present study investigated the structure and growth of the common urban tree species small-leaved lime (*Tilia cordata*) at a highly paved public square (CPS) compared with a contrasting more open, greener square (OGS). Continuously, measured high precision dendrometer data along with meteorological data of the extreme dry and warm summer 2015 as well as dendrochronological data of the sampled trees were investigated to analyze tree growth during a drought year. The results highlight different tree dimensions and growth patterns of the trees at both sites, influenced by tree age and distinct site conditions. While the trees at OGS grew up to 2.5 mm from July until mid of August, the trees at CPS had only 0.4-mm diameter increment. After the initial expansion at CPS, tree diameter contracted again during summer to the point of shrinkage (up to 0.8 mm) at the end of our investigation. Further drought year analysis confirmed the patterns of significant stem growth reductions in the consecutive two years following the drought. A correlation analysis revealed that transpiration, air temperature, and vapor pressure deficit were negatively correlated with the daily diameter growth, whereas precipitation had a strong positive effect. Due to high transpiration rates associated with anisohydric water use behavior, *T. cordata* was able to provide evaporative cooling even during drought. However, this anisohydric behavior resulted in substantial growth decline afterwards especially at paved sites like CPS. Our results suggest selection of tree species, such as those with isohydric water use behavior, which may achieve a better balance between growth, transpiration, and hence evaporative cooling.

**Keywords** Drought stress · Microclimate · Tree growth · Urban trees

## Introduction

Compared to forest stands, urban trees grow in harsh environments with limited rooting space (Day et al., 1995), high temperatures (Kim, 1992; Day et al., 2010), less access to soil moisture and nutrients (Kjelgren and Clark, 1992; Rhoades and Stipes, 1999), and reduced aeration due to impervious pavement (Morgenroth and Buchan, 2009; Rahman et al., 2013). During the past decades, many studies analyzed the effects of growth conditions on urban tree growth and mortality (Nowak et al., 1990; Kjelgren and Clark, 1992; Hodge and Boswell, 1993; Gregg et al., 2003). For example, Bühler et al. (2007) and Day et al. (1995) reported that the planting methods and pit sizes are of great importance for tree establishment and vitality. Species sensitive to soil compaction with sparse rooting space showed reduced growth and vitality

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(Sjöman et al., 2015). Altogether, the mentioned growth conditions can lead to an enhanced soil moisture stress for urban trees.

Water availability can be the most important determinant of tree growth (Littell et al., 2008; Allen et al., 2010; Zhao and Running, 2010; Williams et al., 2013). Along with less water availability during summer, especially in drought years, strong atmospheric demands might drive more transpiration exceeding the tree's water uptake (Clark and Kjelgren, 1990). This negative water balance can result in an internal water deficit in the tissues. Planting drought tolerant species is a possible adaption strategy to water shortage (Moser et al., 2016). Different drought tolerant species usually illustrate different adaption strategies against drought such as early leaf loss (Tyree et al., 1993; Sandquist and Ehleringer, 1998), closure of stomata (Gallé et al., 2007; Pretzsch et al., 2014), osmotic adjustment (Elmi and West, 1995; Arndt et al., 2000), alteration of leaf orientation (Comstock and Mahall, 1985; Werner et al., 1999), and reduction in leaf area (Liu and Stützel, 2004). Among them, different strategies concerning the water use behavior have been commonly discussed, resulting in two different types of adaptation: isohydric species close their stomata immediately during a drought event versus anisohydric species, which are characterized by a slower closure of stomata during and after drought (Pretzsch et al., 2014; Río et al., 2014). Further, the stomatal regulation of isohydric species maintains a consistent minimum leaf water potential, while the leaf water potential of anisohydric species markedly decreases with changes in evaporative demands (Klein, 2014).

Different tree species and even trees of same species but growing in distinct site conditions with different nutrient and soil moisture availability can show altered growth and vitality (Kjelgren and Clark, 1992). Nonetheless, few studies have investigated the effects of the urban climate on tree water relationships and the water balance of typical species in relation to their ecophysiology and hydric behavior (Bush et al., 2008; Chen et al., 2011).

Apart from harsh growth conditions, urban areas are characterized by huge microclimatic differences within short distance depending on the structure of the sites. Urban trees are commonly planted in street canyons, plazas, pedestrian areas, parking areas, and industrial areas (Ko et al., 2015). Those sites combined are regarded as paved urban area impervious to water infiltration and vapor, providing unique surroundings and microclimates for urban trees concerning radiation, water availability, soil compaction, evaporative demands, physical and rooting space, and light availability (Kjelgren and Clark, 1992). While in parks, these extreme conditions are less pronounced and street trees are exposed to high levels of radiation and water shortage with limited rooting space. Kjelgren and Clark (1992) analyzed the microclimatic characteristics of three typical urban environments, street canyons, parks, and plazas, and the effects on the ecophysiology and

growth of mature *Liquidambar styraciflua* trees. Differences in radiation, air temperature, vapor pressure deficit, and evaporation between the sites were found, with the highest values at plazas resulting in the least tree vitality and growth due to higher evaporative demands and limited soil resources (Dobbertin, 2005).

Consequently, tree species with different drought tolerances and ecophysiological traits can have different growth patterns as influenced by their surrounding above—and belowground environment. Still, there is a knowledge gap regarding the growth of mature trees at contrasting urban conditions based on detailed measurements of tree growth, ecophysiological parameters, and micrometeorological conditions especially in relation to climate change and drought. For example, Kjelgren and Montague (1998) investigated the stomatal response and transpiration of *Pyrus calleryana* trees grown over different surfaces. However, the study was conducted with young trees (height of 2.5 m) in containerized pots. With older and larger trees as well as under changed climatic scenarios with frequent years of low precipitation and high temperatures, those results might be substantially different. Knowledge about tree vitality and tree growth in urban street canyons—especially during drought summers—is of special importance for arborists and urban planners since trees provide numerous benefits for the amelioration of the microclimate and well-being of humans (Dimoudi and Nikolopoulou, 2003; Dobbs et al., 2014). A mechanistic understanding of the relationships between tree growth and environmental parameters can provide input for dynamic modeling of urban trees' carbon sequestration (Nowak and Crane, 2002) as well their capacity for cooling by evapotranspiration and shading (Akbari et al., 2001; Shashua-Bar and Hoffman, 2003). Currently, estimates of carbon sequestration and storage are mostly based on static assumptions (Nowak and Crane, 2002; Yoon et al., 2013). Especially, the cooling ability is of great importance for the city climate and human comfort, which has been intensively researched during the past years (McPherson et al., 1997; Gill et al., 2007; Rahman et al., 2015). Due to the difficulties in measuring, scarce data are available on the evapotranspiration of urban trees in relation to their growth and site conditions (water availability, surface temperature, wind speed) and their vitality (trees with less drought resistance have higher evapotranspirational water loss and provide less cooling) (Rahman et al., 2014, 2015). With impending climate change scenarios (IPCC, 2007), the services of trees for the city climate are getting more and more attention and urban trees will play a key role for climate adapted cities (Tyrväinen et al., 2005). The growing conditions influence the vitality and growth of urban trees (Bühler et al., 2006): Stress caused by water deficits reduces photosynthetic productivity and tree growth (Pretzsch and Dieler, 2011; Rötzer et al., 2012), at worst resulting in early senescence (Clark and Kjelgren,

1990). Vital trees are expected to grow faster and provide more benefits for the urban climate (Gómez-Muñoz et al., 2010; Sjöman et al., 2015). However, how common urban tree species react to changed climatic conditions in relation to their site conditions is still scarcely known.

The present study aims to improve the understanding of the growth patterns of urban trees at different urban sites under present climatic conditions as well as in respect to potential climate change and drought years. Therefore, we investigated the commonly planted urban tree species *Tilia cordata*, located at two contrasting sites within the center of a major city (Munich). To address the mentioned knowledge gaps about the growth of urban trees and the effects of the present growth conditions on tree vitality and benefits, we posed the following questions: (1) Do *T. cordata* trees grow differently when planted at two contrasting sites in terms of the degree of impervious surface (highly paved site compared to open, greener site)? (2) How do the reaction patterns to drought stress differ between the chosen contrasting environments? (3) What are the main driving forces behind the revealed growth reaction patterns of urban trees in relation to the microclimate?

## Material and methods

### Site description and data collection

The study site Munich is the third largest city in Germany, located close to the Alps and characterized by a warm temperate climate. The long-term (1961–1990) annual precipitation mean is 959 mm, whereas the mean annual temperature is 9.4°C (DWD, 2016). Munich shows a strong UHI effect with monthly mean UHI intensity up to 6°C in the city center (Pongracz et al., 2010). Two sites within the city center were chosen for data collection, representing typical urban conditions: Bordeaux Platz is an open, green square with less pavement (OGS), while the Pariser Platz is likely to be a highly compacted, heavily paved square (CPS) (Fig. 1). Our criteria for site selection were as follows: (1) two sites with contrasting characteristics in terms of micrometeorology and surface cover, (2) commonly planted and healthy matured street trees of the same species in a sufficient number, and (3) situated close to the city center with pronounced urban heat island (UHI) effect.

Both sites are situated close together (distance of 180 m), with OGS being far bigger than paved-CPS (17,250 to 1945 m$^2$). CPS is a roundabout connected by six roads with heavy traffic with 10 *T. cordata* trees planted, while green-OGS is an oval-shaped avenue with two roads going in opposite directions with less traffic and 84 *T. cordata* trees in total. In total, ten trees at both sites were chosen for data collection. At OGS, we selected five trees of the most southern side, while at CPS we selected five neighboring trees in S, SW,

W, NW, and N directions. The average size of the uncovered tree pit is 55 m$^2$ at OGS and 5.8 m$^2$ at CPS. The planting pits of OGS are avenue plantations surrounded by lawn while the planting pits at CPS are small cut-out pits. Both sites were contrasting in terms of micrometeorological differences especially regarding wind speed and direction due to the shape of the canyons. Paved-CPS had lower wind speed, showed more channeling effect compared to OGS.

We investigated growth patterns, ecophysiology, and drought response of *T. cordata* trees at each site with high precision dendrometer data and increment core data. *T. cordata* was chosen for investigation since it is a commonly planted urban tree species (Pauleit et al., 2002); it is characterized as a shade-tolerant species (Radoglou et al., 2009). Tree data was collected from June to October 2015. The following tree information was recorded: diameter at breast height (dbh), tree height, height to live crown base, crown diameter, crown projection area (CPA), open surface area of the tree pit (OSA), vitality, coordinates including altitude, as well as the leaf area index LAI. Tree height was measured using a Vertex Forestor. The crown radius and the OSA of each tree were measured in eight intercardinal directions (N, NE, . . ., NW) along the ground surface from the center of the trunk to the tip of the most remote downward-projecting shoot and to the last visible open, non-asphalted surface of the soil, respectively, according to Moser et al. (2015, 2016). LAI was estimated from hemispherical photographs taken during the fully leafed phase (June–August) using a Nikon Coolpix P5100 camera with fisheye lens and Mid-OMount following Moser et al. (2015). The resulting hemispherical photos were processed with the program WinSCANOPY (Régent Instruments Inc., Quebec, Canada). Among several methods, we used the LAI-2000 generalized method based on the work of Miller (1967) and Welles and Norman (1991) for deriving LAI.

### Meteorological data collection

Air temperature, air pressure, relative air humidity, precipitation, wind speed, and direction were measured with two Vaisala Weather Transmitters WXT520 (EcoTech Umwelt-Meßsysteme GmbH, Bonn, Germany) at OGS and CPS. The stations were installed on top of street lamp posts at a height of 3.3 m by cross arms 2 m outward from the lamp to avoid influence of lamp and shade of the nearby trees and buildings (Rahman et al., 2017). At OGS, the weather station was placed in a straight line 16.8 m away from the first tree and at CPS 11 m away right at the center of the site to capture differences in micrometeorology as accurately as possible. All measured data were recorded continuously in a 15-min resolution from July 28th to October 21st 2015 on a enviLog remote data logger (EcoTech Umwelt-Meßsysteme GmbH, Bonn, Germany) at each site (Rahman et al., 2017).
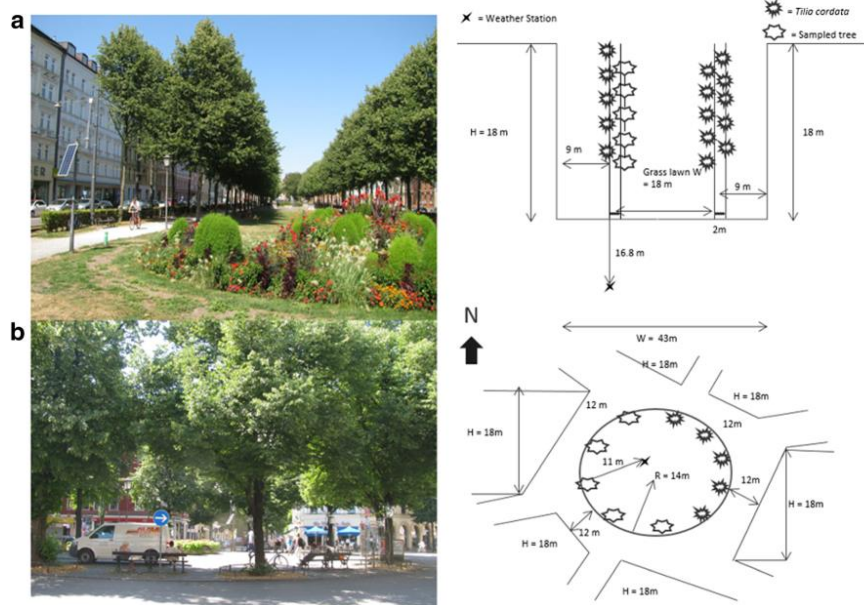
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**Fig. 1** Illustrations of Bordeaux Platz (open green square OGS) (**a**) and Pariser Platz (compacted paved square CPS) (**b**) in Munich

**Soil moisture potential and temperature measurements**

Soil matric potential and temperature were measured using Tensiomark 1 (4244/1, range pF 0-pF 7) (EcoTech Umwelt-Meßsysteme GmbH, Bonn, Germany) installed vertically through the soil profile to a depth of 30 cm. At OGS, the sensors were installed 3.5 m away from the main stem within the edge of the CPA. Three sensors were installed at the first tree, followed by two sensors for the next four to have three replicates for each tree within the grass verges. At CPS, a different setting was necessary due to the small planting pits. Here, two sensors for each tree were installed at the furthest opening point from the main stem. All the sensors were installed where shade was the heaviest to minimize the effect of direct radiation on the soil surface (Rahman et al., 2017).

**Growth data collection in 2015—dendrometer**

For continuous measurements of stem growth, precision dendrometer (Ecomatic, Dachau, Germany) was attached at approximately 4-m height at five trees of each site. The dendrometer measures the linear shifting of a small sensing rod fixed at the bark of the trunk. With stem contraction and expansion, the rod is displaced as well, translating the movement in an electrical signal (Mäkinen et al., 2008; Drew and Downes, 2009). Raw data were recorded every 5 min from

June 23rd to October 21st 2015 and stored in a CR800 data logger (Campbell Scientific, U.K.) equipped with Campbell Logger Multiplexer (AM16/32B). The height was chosen to ensure inaccessibility of installed devices to avoid vandalism.

**Growth over time: dendrochronology**

At both sites, from the five *T. cordata* we studied in detail and of five additional trees each, two cores opposite to each other were extracted at a height of 1.3 m in northern and southern directions. The obtained cores were processed by mounting on grooved boards with glue and sanded using progressively finer sand papers. The first sanding was conducted to flatten the increment cores while the subsequent sandings polished the cores to improve visualization of the cross-sectional area (Speer, 2012). The annual tree ring widths of the cores were measured with a Lintab digital positioning table with a resolution of 1/100 mm (Rinn, 2005). For cross-dating of the time series, the software packages TSAP-Win (Rinn Tech, 2010) were used. All following analyses were carried out with R (R Core Team, 2014), package dplR (Bunn et al., 2015). The tree ring series were detrended with a double detrending process, applying modified negative exponential curves and cubic smoothing splines (20 years rigidity, 50% wavelength cutoff, further averaged with Tukey's biweight robust mean. The autocorrelation of every series was removed using

autoregressive models (maximum order of 3). All further analyses of climate-growth correlations were conducted with the resulting chronologies. The statistical validity of the chronologies was assessed by the Expressed Population Signal (EPS) for the common period of the series. The EPS quantifies the degree to which the final chronology represents the hypothetically perfect chronology (Briffa and Jones, 1990). From the chronologies, the age of the analyzed trees was derived. If the exact age of the tree was not clear (missing tree pith etc.), the age was back-calculated based on the undetrended average growth rate of the last 10 years and the dbh of the tree.

Further, a pointer year analysis was conducted; it computes the individual relative radial growth variation by comparing the ring width of year $t$ to that of year $t-1$ for each tree's chronology. The sign and magnitude of the found variation between the series were calculated (Mérian and Lebourgeois, 2011; Bunn et al., 2015). Positive pointer years display an extraordinary wide ring width, while negative pointer years are characterized by exceptionally narrow rings appearing concurrently within the analyzed trees (Gillner et al., 2014). In this study, pointer years were defined as those years where at least 75% of the series presented an absolute relative radial growth variation higher than 10% (Bunn et al., 2015).

### Climate-growth relationships

To investigate the effects of climate on tree growth, we calculated a monthly drought index, the SPEI (Vicente-Serrano et al., 2010), using data of DWD (2016). The SPEI uses precipitation and potential evapotranspiration (PET) as input data (Vicente-Serrano et al., 2010), whereby a monthly PET was calculated according to the Penman-Monteith equation (Allen et al., 1998) including monthly mean daily minimum and maximum air temperature in °C instead of ambient vapor pressure, monthly mean wind speed in km/h and monthly mean sunshine hours in h. Using the calculated SPEI, the influence of the identified drought years on the growth patterns of urban trees was investigated. A yearly SPEI for 1985–2015 was computed by averaging the monthly index values of the growing season from April to September. Years with an SPEI smaller than −1 were classified as moderate drought years, whereas years with an SPEI smaller than −2 were interpreted as years with severe droughts (Lough and Fritts, 1987; Vicente-Serrano et al., 2010).

Moreover, we analyzed the influence of drought years on tree growth by using a superposed epoch analysis (SEA). SEA assesses the significance of events like droughts on tree growth during drought and post-drought growth periods (Bunn et al., 2015). Following Lough and Fritts (1987), Orwig and Abrams (1997), and Gillner et al. (2014), deviations from the mean ring width index of each core were calculated for the following two periods. The growth of each tree during the drought year derived by the SPEI and the growth of each tree during the 5 years after the drought year (post-drought) were averaged to detect significant departures between those superposed epochs.

### Statistical analysis

All data analysis was conducted with R (R Core Team, 2014). Two-sample $t$ tests were applied to check for statistical differences regarding the tree dimensions of the selected trees at both sites. Assumptions on normal distribution of data and homogeneity of variances were given.

Data derived by dendrometer were cleared of error values (e.g., touching the dendrometer during installation). Dendrometer displacement was converted in micrometer, and daily diameter growth curves were developed for each tree. To assess the influence of measured meteorological data on tree growth in 2015, daily growth values of dendrometer were analyzed in respect to meteorological values via Spearman's rank correlation tests. Further, using the R package mgcv a generalized additive mixed model (GAMM) with daily tree growth over time related to the sites and the meteorological data was developed.

The pointer year analysis and the SEA were computed using the R package dplR (Bunn et al., 2015). For the SEA, random sets of 11 years from 10,000 bootstrapped sets (Gillner et al., 2014; Bunn et al., 2015) were applied.

## Results

### Structure of *T. cordata* at Bordeaux Platz and Pariser Platz

Tree dimensions age, dbh, tree height, crown radius, crown length, CPA, and crown volume of the analyzed trees were significantly higher for trees grown at CPS compared with OGS along with significantly higher age (Table 1). However, open surface area (OSA) was significantly lower at CPS than at OGS. Although LAI of trees at CPS was higher than those of the trees at OGS, they were not significantly different.

### Interannual growth patterns and climate–growth relationships

Due to the younger age, the growth rate of the trees at OGS was 5.0 mm year$^{-1}$ and markedly higher than that of CPS (2.6 mm year$^{-1}$) (Table 2). By contrast, basal area increment (BAI) of trees at CPS was higher compared to those at OGS (206.7 to 80.5 mm$^2$ year$^{-1}$) (Table 3). Moreover, the investigated trees at OGS showed a slightly higher mean growth sensitivity than CPS. Both detrended chronologies exceed

**Table 1** Minimum, average and maximum age, stem diameter (dbh), and tree height (a), crown radius, crown length, and crown projection area (CPA) (b), crown volume, open surface area (OSA), and leaf area index (LAI) (c) of *Tilia cordata* at Bordeaux Platz (open green square OGS) and Pariser Platz (completely paved square CPS), n = sample size

(a)

| Site | n | Age [year] | | | Dbh [cm] | | | Tree height [m] | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | Min | Mean[1] | Max | Min | Mean[1] | Max | Min | Mean[1] | Max |
| Green-OGS | 10 | 35 | 43[a] | 49 | 25.8 | 28.7[a] | 30.3 | 13.7 | 14.9[a] | 15.6 |
| Paved-CPS[2] | 10 | 67 | 95[b] | 132 | 37.7 | 44.1[b] | 50.6 | 15.9 | 17.3[b] | 19.2 |

(b)

| Site | n | Crown radius [m] | | | Crown length [m] | | | CPA [m$^2$] | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | Min | Mean[1] | Max | Min | Mean[1] | Max | Min | Mean[1] | Max |
| Green-OGS | 10 | 4.0 | 4.6[a] | 5.0 | 9.0 | 10.5[a] | 11.9 | 50.9 | 61.8[a] | 79.3 |
| Paved-CPS | 10 | 4.8 | 5.1[b] | 5.5 | 12.2 | 13.8[b] | 14.8 | 72.6 | 82.5[b] | 95.8 |

(c)

| Site | n | Crown volume [m$^3$] | | | OSA [m$^2$] | | | LAI [m$^2$ m$^{-2}$] | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | Min | Mean[1] | Max | Min | Mean[1] | Max | Min | Mean[1] | Max |
| Green-OGS | 10 | 463.1 | 655.9[a] | 895.8 | 30.2 | 55.2[a] | 48.7 | 1.8 | 2.4[a] | 3.9 |
| Paved-CPS | 10 | 965.1 | 1135.5[b] | 1341.9 | 6.2 | 5.8[b] | 8.7 | 2.0 | 2.6[a] | 3.9 |

[1] Mean values in the same column differ significantly when followed by different letters (Two-sample t-test, $p < 0.05$)

[2] Age of CPS based on extrapolated values

the commonly applied threshold of 0.85 for the EPS (Wigley et al., 1984).

The time series of trees at CPS covered the period of 1926 to 2015 while the time series of trees at OGS ranged from 1987 to 2015 (Fig. 2). In general, the trees at both sites displayed similar growth patterns with low growth episodes e.g., in 2005 and high growth episodes e.g., in 2007.

To statistically validate the years with extreme low and high growth, a pointer year analysis was conducted (Table 3). Pointer years at both sites were found for 2005, 2009, and 2011. Other years like 2003 are negative pointer years at only one of the investigated sites. All in all, eight negative pointer years were found for green-OGS and nine for paved-CPS. In the common time period of both sites, less positive pointer years (8) were found at CPS compared to OGS (9). Similar patterns were found regarding the negative pointer years of both sites (6 at CPS and 8 at OGS).

Using the SPEI, the growth of *T. cordata* at both sites before, during, and after drought years was investigated (Fig. 3). In Munich, the SPEI of the worst drought year (2003) was −1.8, other drought years were 2015 (−0.75), 2014 (−0.80), 2004 (−0.66), 1998 (−0.82), and 1997 (−0.76).

The superposed epoch analysis—performed with the derived drought years by the SPEI—revealed similar drought responses of the trees at OGS and CPS (Fig. 4). The trees at both sites showed decreased growth values during drought years (0). In the first (1) and second (2) years following drought, growth was still significantly lower than the average growth of the time series. In the third year (3) after a drought event, the trees recovered sufficiently to resume tree ring growth. There were no significant differences in the growth during and after drought between green-OGS and paved-CPS. The younger trees at OGS exhibited a smaller growth decrease following the drought years than the older trees at CPS. The trees of paved-CPS had a stronger growth decline during the drought year. However, in the third and fourth years after a drought event, the older trees at CPS showed a better growth compared to the average growth of the time series and a greater recovery than the trees at OGS.

Analysis of the long-term growth of trees at both sites highlighted similar reaction patterns during drought regardless of differences in age and surface properties. The highly significant growth reductions in the second post-drought year at both sites might also be influenced by the drought year occurrence; the drought years 2014, 2003, and 1997 might have subsequent aftereffects of water stress and reduced tree growth, which increased the drought stress on the trees strongly, leading to the greatest growth drop afterwards.

**Table 2** Calculated age, growth rate, basal area increment (BAI), mean sensitivity of growth, and expressed population signal (EPS) of *Tilia cordata* at Bordeaux Platz (open green square OGS) and Pariser Platz (completely paved square CPS) with standard deviation (SD) based on tree ring data

| Site | Age [a] | Growth rate ± SD [mm year$^{-1}$] | BAI ± SD [mm$^2$ year$^{-1}$] | Mean sensitivity | EPS[a] |
|---|---|---|---|---|---|
| Green-OGS | 43 | 5.0 ± 0.6 | 80.5 ± 57.4 | 0.5 | 0.9 |
| Paved-CPS | 95 | 2.6 ± 0.8 | 206.7 ± 149.5 | 0.4 | 0.9 |

[a] Based on the ring width index of tree ring chronologies for the common period, obtained by double detrending of the tree ring series

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**Table 3** Positive and negative pointer years detected by the year ring width of *Tilia cordata* at both sites and at Bordeaux Platz (open green square OGS) as well as Pariser Platz (completely paved square CPS) separately (italic years of CPS outside of the common time period of both sites)

| Site | Positive | n | Negative | n |
|------|----------|---|----------|---|
| Both | 2012, 2010, 2006, 2000, 1995 | 5 | 2014, 2011, 2009, 2005 | 4 |
| Green-OGS | 2002, 1992, 1990, 1987 | 4 | 1999, 1998, 1994, 1989 | 4 |
| Paved-CPS | 2013, 2004, 1989, *1965, 1950, 1947* | 6 | 2015, 2003, 1997, *1941, 1933* | 5 |

### Intraannual growth patterns (2015)

All in all, the daily growth increment of *T. cordata* at two contrasting sites decreased over the summer. At OGS, the trees showed a steady increase in a daily diameter until mid-August. Then, growth of most trees started to decline, with tree no. 2 decreasing the most by 1.0 mm in diameter. Other trees such as nos. 1, 3, and 4 illustrated minor growth reduction. Only tree no. 5 grew slightly during summer. Compared to green-OGS, the trees at CPS exhibited significantly reduced growth. While the trees at OGS grew up to 2.5 mm until mid of August, the trees at CPS had only 0.4-mm diameter increment. Moreover, the growth patterns during the observed period are different at CPS, where all investigated trees had a distinct reduction in growth until mid (only tree no. 5) or end of September (rest of the trees). After initial diameter expansion, tree diameter contracted again during the summer up to 0.8 mm at the end of our investigation (Fig. 5).

Trees at both sites reacted similarly to microclimate (Fig. 6). Only soil matric potential, soil temperature, transpiration in September, and wind speed in August and September displayed contrary patterns in their relationships with growth at both sites. The vapor pressure deficit and the air temperature were negatively correlated with growth at both sites, with a very strong negative relationship throughout August, September, and October at OGS. However, the observed negative relationship was visible only in October at CPS. The amount of precipitation on the other hand was beneficial for tree growth, especially for the trees at OGS. Transpiration mostly had a negative correlation with tree growth, though in July and September both illustrated a positive relationship with growth at CPS, while in July transpiration showed a positive correlation with the tree growth at OGS. Additionally, the soil matric potential changed in its relationship with tree growth: In August, a less negative soil matric potential was beneficial for growth at both sites, while in September a positive relationship with growth was observed at OGS. In October, the soil matric potential was negatively correlated with the growth at both sites. Finally, higher soil temperature had a negative effect on tree growth during all 3 months at OGS, while soil temperature had a positive effect on growth at CPS in August. The following months also showed a negative correlation. In summary, all measured meteorological data except precipitation had a negative correlation with growth, with vapor pressure deficit, air temperature, transpiration, and soil matric potential illustrating the greatest influence ($r < -0.5$, $r^2 > 25$) on tree growth (Fig. 6 (i)).

A performed GAMM revealed a significantly lower interannual growth rate of the trees at CPS compared to OGS (Table 4). Moreover, there was a significant influence of the soil matric potential, precipitation, and transpiration on tree growth. The soil temperature, vapor pressure deficit, air temperature, and wind speed had no significant influence on growth in relation to the site differences and were consequently removed from the final model. The strongest negative influence had the site and the transpiration decreasing growth, while precipitation and the soil matric potential proved to be of positive influence on tree growth.



**Fig. 2** Ring width index (RWI) of *Tilia cordata* at Bordeaux Platz (OGS, *solid line*) and Pariser Platz (CPS, *dashed line*) after double detrending (negative exponential function and 2/3 cubic smoothing spline)
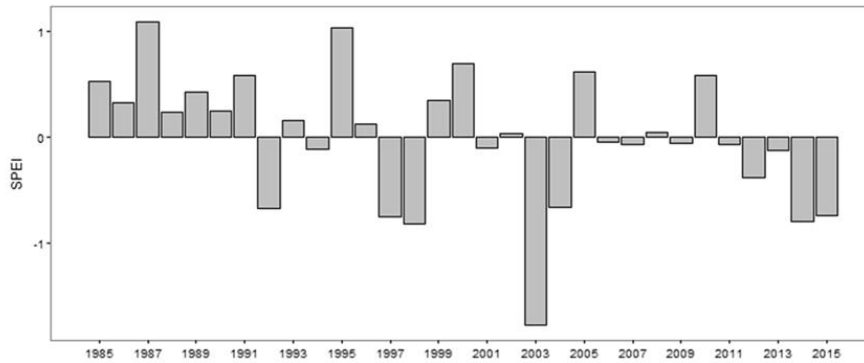
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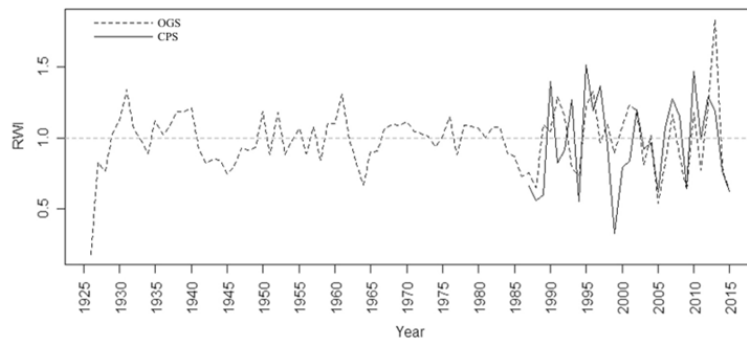
**Fig. 3** Calculated standard precipitation-evapotranspiration index (SPEI) for Munich from 1985 to 2015 with a time scale of 4 months. *Bars* > 0 represent positive SPEI values of years with a positive climatic water balance, and *bars* < 0 represent negative SPEI values of years with a negative climatic water balance

## Discussion

### Structure of *T. cordata* at Bordeaux Platz and Pariser Platz

This study illustrated that growth patterns of *T. cordata* were highly influenced by the surrounding environment. The average tree growth during the severe drought year 2015 at the highly paved site Pariser Platz CPS was significantly lower than the growth of the trees at the more open, greener Bordeaux Platz OGS and confirmed the original hypothesis of reduced growth due to extreme environmental conditions associated with paving. This pattern of an overall greater growth of younger trees might also be influenced by the CPS trees being older (they were on average 52 years older), since age can induce greater DBH and crown dimensions. As a study of Ryan et al. (1997) showed, younger trees grow faster than older trees ("age-trend"), but age-related bias in ring increment was removed by detrending. Thus, the results

of the drought year analysis SEA were not influenced by differing age between the two sites.

As shown in Table 1, the investigated trees at CPS varied in their age and size development, even though they were planted in short distance of each other, which may be due to heterogeneous soils and microclimates, e.g., shading by buildings. Bühler et al. (2006) found similar patterns of varying size of established *T. cordata* trees in Copenhagen. However, the measured structural dimensions of *T. cordata* at both sites are in line with other studies about the growth of this species. For example, Moser et al. (2015) measured the growth and selected ecosystem services of *T. cordata* trees in Munich, especially analyzing the allometry based on dbh. The trees at CPS were similar to those measured by Moser et al. (2015). Due to being older, they represented the upper range of urban tree growth and tree structures. By contrast, younger trees at OGS had more uniform and slightly faster growth compared to the results of Moser et al. (2015). A comparison of the *T. cordata* structural dimensions with a study of Larsen and Kristoffersen
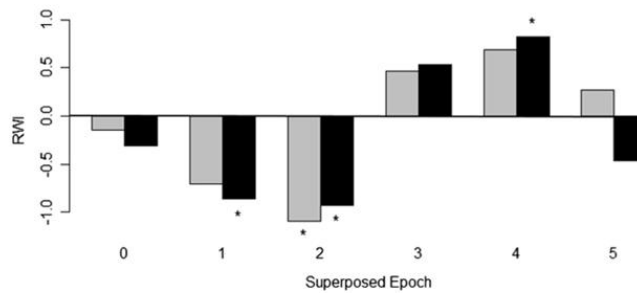


**Fig. 4** Superposed epoch analysis of the ring width index (RWI) during drought years (0) and after drought (1–5) for *Tilia cordata* at Bordeaux Platz (OGS, *gray*) and Pariser Platz (CPS, *black*). Input drought years are 2015, 2014, 2004, 2003, 1998, 1997, and 1992. Marked columns (*asterisk*) are significantly different at $p < 0.05$
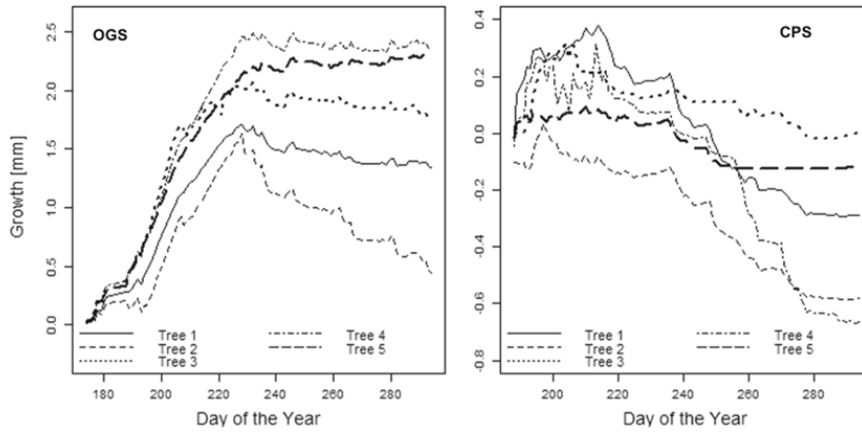
**Fig. 5** Daily growth of the *Tilia cordata* trees at Bordeaux Platz (OGS, *left*) and Pariser Platz (CPS, *right*) during summer, 2015 (July, 1st to October, 21st, 2015)
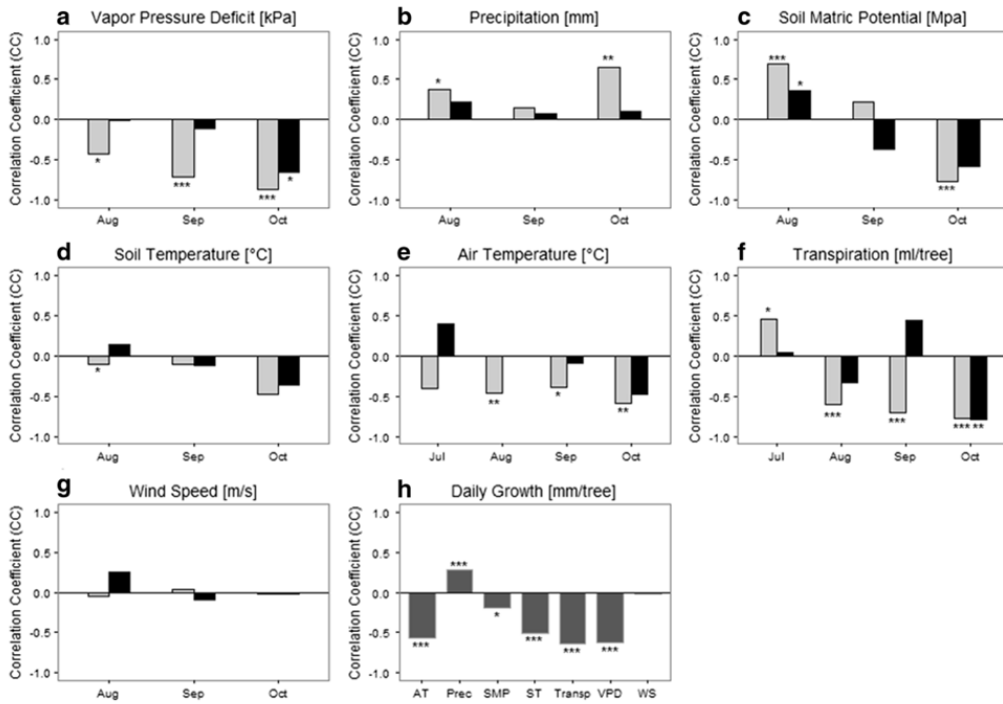


**Fig. 6** Correlations between minimum daily growth and **a** vapor pressure deficit (*VPD*), **b** precipitation (*Prec*), **c** soil matric potential (*SMP*), **d** soil temperature (*ST*), **e** air temperature (*AT*), **f** transpiration (*Transp*), and **g** wind speed (*WS*) of *Tilia cordata* at Bordeaux Platz (OGS, *gray bars*) and Pariser Platz (CPS, *black bars*) in July, August, September, and October 2015, and **h** the correlations of measured meteorological data with growth at both sites together. *Asterisks* indicate a significant correlation (*at the $\alpha < 0.05$, **at the $\alpha < 0.01$, ***at an $\alpha$-level $< 0.001$)

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**Table 4** Generalized additive mixed model (GAMM) on the mean daily growth rates of *Tilia cordata* at Bordeaux Platz (open green square OGS) and Pariser Platz (completely paved square CPS) from July to October 2015 of the form: daily growth [mm] = a + s(day, by site) + b × soil matric potential + c × precipitation + d × Transpiration

|  | Value ± SE | p |
| --- | --- | --- |
| Intercept (a) | −0.03 ± 0.01 | 0.02 |
| Site (s) | −0.01 ± 0.003 | < 0.001 |
| Soil matric potential (b) | 0.01 ± 0.003 | 0.003 |
| Precipitation (c) | 0.002 ± 0.0002 | < 0.001 |
| Transpiration (d) | $-2.2 \times 10^{-7} \pm 1.0 \times 10^{-7}$ | 0.03 |
| SD intercept (e) | 0.003 | – |
| $\varepsilon$ | 0.02 | – |

Levels of site: 2 (CPS and OGS)

*SD* standard deviation, *SE* standard error

(2002) in Copenhagen revealed that the measured trees at OGS had a similar dbh with a faster height growth, while the trees at CPS were smaller than those of the same age in Copenhagen. Less growth and structural dimensions of the CPS trees was likely induced by greater heat and drought stress due to surrounding paving, building proximity, small planting pits, and compacted soil that would limit rooting volume and uptake of soil water, reducing vitality and ultimately leading to reduced growth as shown by Nielsen et al. (2007).

**Interannual and intraannual growth patterns, climate-growth relationships**

However, Nielsen et al. (2007) did not analyze the growth patterns during drought, which may limit growth even further. In our study, we measured the diameter increment of ten *T. cordata* trees at two contrasting sites in high temporal resolution (every 5 min) during the hot and dry summer of 2015. Overall, the measured trees showed increased diameter growth until the beginning of August, with the trees at the highly paved site CPS exhibiting a growth reduction several days earlier than those at the more open and greener OGS. When comparing both sites, the effect of shrinkage due to extensive drought was observed for CPS later in the season, though not at OGS. Reduction in diameter occurs when tree transpiration exceeds availability of stored intercellular and soil water uptake (Sjöman et al., 2015). If this imbalance continues, tree diameter decrease and further decline in growth may occur as a combination of influences like limited water supply, loss of hydraulic conductance, reduced photosynthesis, restricted soil and rooting volume, disruption in soil hydrological processes, and climate. Further, impermeable surfaces and compacted soil can accelerate water deficits (Sieghardt et al., 2005; Morgenroth and Visser, 2011; Sjöman et al., 2015). Physiological differences in response

to drought may be due to different site conditions. The dendrochronological analysis showed that more negative pointer years were found at the paved site (9 at CPS compared to 8 at OGS), likely as a result of their greater age (Ryan et al., 1997). However, more positive pointer years were found for OGS in the common time period of both sites (8 at CPS to 9 at OGS). SEA analysis revealed similar patterns in growth during and after drought at CPS and OGS. Significant increment growth reductions were found in the years after a drought event. Slight differences were revealed in the severity of the growth reductions and the following recovery, with the highly paved-CPS exhibiting greater growth decreases after drought though better recovery. This might be due to the higher tree age at CPS, since older trees can recover better from drought events (Lloret et al., 2011).

The patterns of growth reductions after drought events can be explained with the evident anisohydric water stress behavior of *T. cordata*. Anisohydric species allow their leaf water potential to decrease during drought by sustaining a relatively high stomatal conductance and thus carbon assimilation (Klein, 2014; Roman et al., 2015). Therefore, growth may be reduced as a consequence in the years following drought events (Pretzsch et al., 2014; Río et al., 2014). While during the drought event, no immediate growth reductions would occur, minimal stomatal closure means high transpiration rates despite soil drying, leading to a greater risk of cavitation and disruptions in xylem water management (McDowell et al., 2008; Roman et al., 2015), decreasing growth and vitality in subsequent years. High atmospheric demand along with high transpiration rates due to a limited soil moisture availability might also cause the observed shrinkage of trees at CPS as a consequence of the drought year 2015 and previous drought years like 2014.

Differences in 2015 growth patterns found by dendrometer were likely caused by site differences in terms of microclimate. Most importantly the study illustrated the negative influence of high temperatures, less water availability and transpiration on tree growth, especially during drought years. All measured meteorological data except precipitation had a negative influence on tree growth, while higher air and soil temperatures, a greater vapor pressure deficit as well as higher transpiration resulted in reduced growth during summer. These findings are in line with the mixed model results, a reduced amount of precipitation results in less water availability in the soil, leading to a reduced soil matric potential. Increased air and soil temperatures induce higher vapor pressure deficit (Kirschbaum, 2004) resulting in higher transpiration and hence evapotranspirational cooling (Rahman et al., 2014), depleting the intracellular stored water. The relationship between growth and meteorological variables, in particular transpiration, was also likely due to the anisohydric character of *T. cordata*. High atmospheric demand translates to high transpiration rates in anisohydric species, resulting in

greater soil water depletion, water stress, and stem shrinkage that ultimately reduce increment growth (Marsal et al., 2002; Scoffoni et al., 2014). Albeit, trees with a higher transpiration rate should also have a higher girth and are expected to have greater cooling abilities, since trees with higher girth usually have a greater sapwood area and more sap flow to support more leaf area (Rahman et al., 2014; Moser et al., 2015).

Aggressive anisohydric water use behavior might provide greater overall evaporate cooling and thermal comfort to mitigate the effects of climate change. However, our study suggests that added cooling during a hot, dry year may come at the cost of reduced growth in subsequent years. This makes *T. cordata* a not very suitable urban tree species under hotter, drier conditions, especially at paved, impervious sites like CPS, where irrigation might be needed to meet transpiration needs and sustain evaporative cooling of anisohydric species such as *T. cordata*. At the limits of extreme drought, such water use behavior could prove unsuitable, increasing the risk of damage to tree health and loss of canopy (Kolb and Stone, 2000; Klein, 2014). Thus, in the context of a hotter, drier climate, species with an isohydric water stress response might be a more suitable choice for harsh sites with reduced water availability.

## Conclusions

In conclusion, the conducted study could display how the growth of *T. cordata* at two contrasting sites was influenced by the surrounding environment in terms of the site conditions soil, size of planting pits, and meteorological variables like precipitation and temperature. Since the urban environment is very heterogeneous, even sites in short distance (like green-OGS and paved-CPS) can vary tremendously. Those site differences and the severe drought year 2015 caused a decrease in the growth of the measured trees at both sites; however, the *T. cordata* trees at CPS suffered significantly more to the degree of the shrinkage of tree girth in summer. This was very likely induced by a negative water balance due to insufficient water supply consequently high negative soil matric potential and following high water loss through transpiration. Due to the anisohydric water stress response of this species, *T. cordata* can provide high rates of cooling even during drought years, however, which can cause high growth decreases afterwards. In the future, tree species which achieve a better balance between growth, transpiration, and hence evaporative cooling should be preferred. The observed patterns of tree behavior during and after drought events were mainly caused by species traits and also influenced by the environment. Though the present study showed the negative influence of highly paved sites with related problems like less water availability and rooting space on tree growth, especially during drought years, there are still knowledge gaps concerning other issues affecting growth, e.g., effects of the growing media and planting methods, irrigation, as well as the specific microclimate environment at the tree site. There is an increasing demand for future studies on these influences to gain more expertise for sustainable tree planting, particularly at difficult sites to ensure healthy, well-grown trees, providing ecosystem services such as evaporative cooling also under future climate change conditions.

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## 4. Paper IV

Moser, A., Rötzer, T., Pauleit, S., Pretzsch, H. (2016): Stadtbäume: Wachstum, Funktionen und Leistungen – Risiken und Forschungsperspektiven. Allgemeine Forst- und Jagdzeitung AFJZ, in Begutachtung (under review)

# Stadtbäume: Wachstum, Funktionen und Leistungen –
# Risiken und Forschungsperspektiven

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## 1. Einleitung

Als Folge der Urbanisierung steigt die Zahl der in Städten lebenden Menschen kontinuierlich an. Laut Prognosen der UNO wird bis 2030 die Stadtbevölkerung weltweit mehr als 60 % der Weltbevölkerung ausmachen und bis 2050 circa 70 % erreichen (United Nations, 2008). Für München hat 2006 die Bertelsmann-Stiftung einen Bevölkerungsanstieg von 7,8 % auf 1,42 Millionen Menschen bis 2020 prognostiziert (Bertelsmann-Stiftung, 2006). Diese Prognose muss jedoch bereits angepasst werden, da in München im Jahr 2015 schon 1,44 Millionen Einwohner gemeldet waren (Bayerischen Landesamtes für Statistik und Datenverarbeitung, 2016). Auch Nowak & Dwyer (2007) haben in den USA diesen Trend der Urbanisierung beschrieben, danach ist der Anteil an urbanen Flächen der USA von 2.5 % in 1990 auf 3,1 % in 2000 angestiegen. Folgen der Urbanisierung sind u.a. ein hohes Verkehrsaufkommen mit steigendem Lärmpegel und Schadstoffbelastungen der Luft. Ebenso bewirken die hohen Bebauungsdichten und Versiegelungsgrade höhere Temperaturen im Vergleich zum Umland sowie höhere Maximaltemperaturen und eine hohe Wärmespeicherung, sowohl am Tag als auch in der Nacht. Dieser Effekt wird als städtische Wärmeinsel („urban heat island" UHI) bezeichnet (Oke, 1982). Gründe dafür sind die hohen Anteile versiegelter Flächen wie Asphalt oder Gebäude, welche

die einfallende Sonnenstrahlung absorbieren und als Wärmeenergie wieder abgeben (Santamouris et al., 2001). Daneben sind in der Stadt oft nur geringe Grünanteile vorhanden, welche durch Transpiration und Beschattung eine Abkühlungswirkung erzielen könnten (Collier, 2006; Kuttler, 2004; Oke, 1982). Abbildung 1 stellt das spezifische Stadtklima und die urbane Hitzeinsel dar: Versiegelte Flächen und Gebäude absorbieren die einfallende Strahlung und verminderte Windbewegungen erhalten die warme Luft in der Stadt, was zu der urbanen Grenzschicht führt (Oßenbrügge & Bechtel, 2010). Folgen der städtischen Wärmeinsel sind erhöhte Kosten für Klimaanlagen und mehr Bedarfsspitzen, höhere Umweltbelastungen und negative Folgen für die Gesundheit der Stadtbewohner (Hassid et al., 2000; Santamouris et al., 2001; Santamouris et al., 2007; Santamouris et al., 2011). Denn insbesondere hohe Nachttemperaturen führen zu Schlafproblemen und verminderter Erholung (Santamouris, 2014).
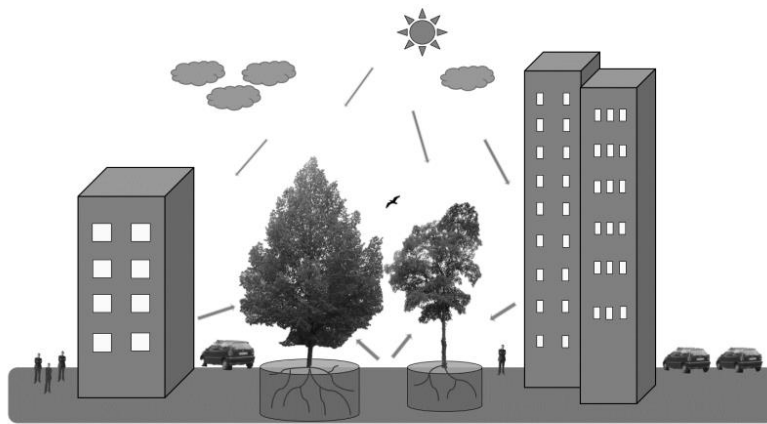


*Abbildung 1: Hohe Bebauungsdichten und Asphaltierungsgrade in der Stadt begünstigen den urbanen Wärmeinseleffekt*
*Figure 1: High building densities and highly sealed sites intensify the urban heat island effect*

Derzeitige Prognosen prophezeien eine Verstärkung der angesprochenen Probleme in der Stadt durch den Klimawandel. Extremwetterereignisse wie Starkregen, Unwetter und extreme Hitze werden vermehrt auftreten. Sinnflutartige Niederschläge werden häufiger vorkommen, wie zum Beispiel im Juni 2016, als in Bayern bis zu 150 Liter pro Quadratmeter innerhalb von wenigen Tagen fielen. Diese großen Mengen an Wasser führen zu einer Überlastung der Kanalisation, Überschwemmungen und volllaufender Keller. Auch werden die Temperaturen deutlich steigen, nach dem IPCC Klimaszenarien wird es bis in 60 Jahren zu einer Erwärmung von 2 bis 4,5°C der Jahresmitteltemperatur Deutschlands kommen, wobei erneut insbesondere nachts hohe Temperaturen auftreten können. Tropennächte (Minimum nicht unter 20°C), welche bis vor wenigen Jahren noch kaum in Deutschland vorkamen, werden gehäuft auftreten. Auch die

Anzahl der Sommertage (Tagesmaximum über 25°C) und der heißen Tage (Tagesmaximum über 30°C) werden voraussichtlich deutlich zunehmen.

Um diesen Trends entgegenzuwirken und ein für den Menschen weniger belastendes Stadtklima zu fördern, wird vermehrt im Bereich klimaangepasster Städte geforscht. Einrichtungen wie das „Zentrum für Stadtnatur und Klimaanpassung" ZSK der TU München (www.zsk.tum.de) und das Projekt „Future Cities" (Lippeverband, 2013) beschäftigen sich mit der Leistungsfähigkeit von grüner Infrastruktur für das Stadtklima und den thermischen Komfort. Neben Dach- und Fassadenbegrünung werden insbesondere die Wirkungen und Leistungen von Stadtbäumen für das Klima und den Menschen näher untersucht.

Per Definition des BMUB (2015) werden mit Stadtgrün oder urbanem Grün alle Formen grüner Freiräume und begrünter Gebäude zusammengefasst, darunter Bäume, Sträucher, Hecken und Wiesen in Parkanlagen, Friedhöfen, Kleingärten, Brachflächen, Spielbereiche und Spielplätze, Sportflächen und Straßen. Zu Grünflächen zählen Straßenbäume, Siedlungsgrün, Grünflächen an öffentlichen Gebäuden, Naturschutzflächen, Wald und weitere Freiräume, die zur Gliederung und Gestaltung der Stadt entwickelt, erhalten und gepflegt werden müssen. Auch private Gärten und landwirtschaftliche Nutzflächen sind ein wesentlicher Teil des Grüns in den Städten, ebenso wie Bauwerksgrün mit Fassaden- und Dachbegrünung, Innenraumbegrünung sowie Pflanzen an und auf Infrastruktureinrichtungen. Durchschnittlich kommen in Großstädten auf jeden Einwohner 46 m² Grünflächen (BMUB, 2015). Alle diese Formen des städtischen Grüns können – als „Grüne Infrastruktur" geplant und entwickelt – unverzichtbare wirtschaftliche, soziale und ökologische Leistungen für das Wohlergehen der Stadtbewohner erbringen.

Aufgrund ihres prägnanten Erscheinungsbildes und ihrer Größe werden insbesondere Stadtbäume mit dem Begriff Stadtgrün in Verbindung gebracht und auch die Leistungen und Funktionen des Stadtgrüns beziehen sich zumeist auf die der Stadtbäume (McPherson et al., 1997; Tyrväinen et al., 2005). Jedoch hat eine intensive Erforschung der Stadtbäume erst vor wenigen Jahrzehnten begonnen und steckt im Vergleich mit der jahrhundertealten Forstwissenschaft noch in den Kinderschuhen. Es fehlt an akkuraten Datensätzen und Wissen, allometrische Beziehungen speziell für Stadtbäume liegen kaum vor und Angaben zu ihrer Leistungsfähigkeit wie Kohlenstoffspeicherung, Sauerstofffreisetzung, Abkühlungswirkung und Beschattung beruhen allenfalls auf groben Schätzungen (Peper et al., 2014; Rust, 2014; Semenzato et al., 2011). Pflanzungen erfolgen anhand von Pflanzlisten (z. B. GALK Straßenbaumliste) und orientieren sich nicht an den Leistungen des Baumes sondern ausschließlich an der gärtnerischen und ökologischen Eignung für städtische Pflanzungen.

Eine genaue Kenntnis der Ansprüche einzelner Baumarten und das Wissen ihrer Leistungen ist im Hinblick auf den Klimawandel und der Problematik vermehrter Schädlinge im Stadtbaumbereich dringend notwendig (Tubby & Webber, 2010). Aufgrund fehlender Datensätzen wird für die Modellierung von Wachstum und Umweltleistungen oftmals auf bestehendes Wissen aus Waldbeständen zurückgegriffen (Aguaron & McPherson, 2012; Peper et al., 2014; Pretzsch et al., 2015a). Stadtbäume unterscheiden sich jedoch von Waldbäumen deutlich (McHale et al., 2009; Nowak & Crane, 2002). Unterschiedlich ist insbesondere der Wuchsraum. Im Gegensatz zu Waldbäumen oder Bäumen in Parkanlagen ist der Lebensraum von Bäumen in Straßen und auf gepflasterten Stadtplätzen oft geprägt durch begrenzte Wuchsräume, verminderte Wasser- und Nährstoffversorgung und Gebäudenähe (Clark & Kjelgren, 1990; Grabosky & Bassuk, 1995). Auch stehen bei Stadtbäumen der ästhetische Gedanke und ihre Klimafunktionen im Vordergrund (Bolund & Hunhammar, 1999; Pauleit et al., 2011), während der Nutzen von Forstbeständen eher auf dem Holzbedarf ausgerichtet ist.

Mit dieser Literaturstudie wird der Wissensstand zur Struktur und Dynamik von Stadtbäumen dargestellt sowie die Funktionen und Leistungen des Stadtbaumbestandes erläutert. Es werden die positiven und negativen Aspekte der Stadtbäume aufgezeigt sowie deren Wirkungen auf den Menschen und das Stadtklima, aber auch die Effekte des Standortklimas auf den Baum werden diskutiert. Weiter werden die Folgen des Klimawandels für den Baumbestand der Stadt dargestellt sowie moderne Konzepte und zukünftige Forschungsperspektiven analysiert. Zu diesem Zweck wurden in dieser Literaturstudie weltweit Schlüsselstudien der Stadtbaumforschung herangezogen sowie neue Forschungsansätze und Studien vorgestellt. Die Ziele der vorliegenden Studie sind:

- Herausarbeitung der Unterschiede im Wachstum, des Wuchsraumes, der Artenzusammensetzung und Wahrnehmung von Stadtbäumen im Vergleich zu Forstbeständen
- Darlegung und Diskussion der wichtigsten Leistungen von Stadtbäumen und der durch Stadtbäume entstehenden Risiken für Mensch und Umwelt
- Darstellung möglicher Auswirkungen des Klimawandels auf das Wachstum und die Funktionen von Stadtbäumen
- Analyse zukünftiger Forschungsperspektiven und Wissenslücken im Bereich der Stadtbaumforschung

Mit Hilfe der dargelegten Ergebnisse dieser Studie kann das Management der Stadtbaumbestände verbessert werden, um den Zustand, die Entwicklung und zukünftige Methoden besser abschätzen zu können. Ebenso können die Leistungen und Funktionen der städtischen Bäume und deren Veränderungen ermittelt werden.

## 2. Struktur und Dynamik von Stadtbäumen

Stadtbäume können als Bäume definiert werden, die an typischen urbanen Standorten wie Straßenzügen, an Parkplätzen, vor öffentlichen Gebäuden, in Stadtparks, in Industriegebieten und an Gleisen oder ähnlichem zu finden sind. In Nordamerika wird die Gesamtheit der städtischen Gehölzbestände auch als „Urban Forest" bezeichnet (Moll, 1989). Der Wuchsraum der Stadtbäume ist im Gegensatz zum typischen Waldstandort deutlich verändert und häufig geprägt von wachstumshindernden Faktoren wie stark eingeschränktem Wuchsraum (Day et al., 1995; Grabosky & Bassuk, 1995), geringer Wasser- und Nährstoffverfügbarkeit (Beatty & Heckman, 1981; Whitlow & Bassuk, 1986), hohen Temperaturen (Akbari et al., 2001; Kjelgren & Clark, 1992), hoher Bodenverdichtung und Asphaltierung (Beatty & Heckman, 1981; Day et al., 1995), Schadstoffeintrag wie Hundeurin und Streusalz (Petersen & Eckstein, 1988; Whitlow & Bassuk, 1986). Da Stadtbäume jedoch oft ohne direkte Konkurrenz zu anderen Bäumen wachsen, können sie größere Dimensionen erreichen als Forstbäume (Hasenauer, 1997).

Die Artenzusammensetzung der Bäume in mitteleuropäischen Städten weicht stark von denen mitteleuropäischer Forstbestände ab, ein großer Teil der Stadtbäume sind fremdländische Arten. So sind in Deutschland 171 Arten nach der GALK Straßenbaumliste eingestuft (GALK e. V., 2016). Diese Liste soll wissenschaftliche Daten über das Wachstum, die Resistenz, Größe und Verwendbarkeit von Bäumen in Städten bereitstellen und wurde 1975 auf der Sitzung der Gartenamtsleiter ins Leben gerufen (GALK e. V., 2016). Die Anzahl an gepflanzten Stadtbäumen variiert je nach Größe der Stadt, so umfasst der Straßenbaumbestand von Berlin beispielsweise 438.000 Bäume, wobei 14 % des Stadtgebiets öffentliche Grünflächen ausmachen (Stadt Berlin, 2016). Im Stadtgebiet München bedeckten 1982 die Kronen der Stadtbäume und Sträucher einen Flächenanteil von 17 %, was sogar einem ähnlichen Anteil an Hausflächen entspricht (Pauleit & Duhme, 2000).

### 2.1 Strukturmerkmale - Übersicht über funktionelle Gruppen

Trotz der vielen in Städten vorkommenden Baumarten ist der Stadtbaumbestand meist von wenigen Arten dominiert. In Mitteleuropa zählen Linden *Tilia* zu den am häufigsten gepflanzten Arten (bis zu 50 % des Straßenbaumbestandes), insbesondere die Winterlinde (*Tilia cordata*) (Pauleit & Duhme, 2000). Weitere häufige Gattungen sind Eichen *Quercus*, Ahorn *Acer*, Hainbuchen *Carpinus*, Platanen *Platanus*, ferner Kastanien *Aesculus*, Birken *Betulus* und in manchen Städten Robinien *Robinia*. Als heimische Stadtbaumarten zählen Linde (Winterlinde, Sommerlinde *Tilia platyphyllos*), Ahorn (Spitzblättriger Ahorn *Acer platanoides*, Bergahorn *Acer pseudoplatanus*, Feldahorn *Acer campestre*), Buche *Fagus sylvatica*, Eiche (Stieleiche

*Quercus robur*, Traubeneiche *Quercus petraea*, Flaumeiche *Quercus pubescens*), Rosskastanie (*Aesculus hippocastanum*), Edelkastanie (*Castanea sativa*), Pappel *Populus nigra* 'Italica', Birke *Betula pendula,* Esche *Fraxinus excelsior* und Hainbuche *Carpinus betulus*. Einige dieser Arten haben eine weit zurückreichende Bedeutung. So war z. B. die Linde früher ein Symbol des Dorfmittelpunktes, während Kastanien in Süddeutschland typische Bäume für Biergärten sind.

Dagegen gelten die Robinie *Robinia pseudoacacia*, Platane *Platanus ×hispanica*, Zerreiche *Quercus cerris,* Ginko *Ginko biloba* und Schnurbaum *Styphnolobium japonicum* als Beispiele für fremdländische, eingebrachte Arten. Einige Arten gelten inzwischen als Problemarten, insbesondere wegen ihrer starken ungewollten Vermehrung, wie der Götterbaum *Ailanthus altissima* oder die Robinie.

Darüber hinaus zählen Obstbäume (Gattung *Prunus* und weitere), die vermehrt in Privatgärten vorkommen, ebenfalls als Stadtbäume.

In anderen Teilen der Welt ist die Baumartenzusammensetzung in Städten natürlich anders als in Mitteleuropa. So sind z. B. in Chicago Ahorn, Eschen und Linden dominierende Stadtbäume gleicher Gattungen wie in Europa, wobei jedoch viele andere Arten vorkommen McPherson et al. (1997).

*Tabelle 1: Artenzusammensetzung von Straßenbäumen in Chicago verändert nach McPherson et al. (1997)*

*Table 1: Composition of tree species on streets and all lands in Chicago adapted from McPherson et al. (1997)*

| | Straßenbaumart (street tree species) | | Anteil in % (percentage tree cover) |
|---|---|---|---|
| 1 | *A. saccharinum* | Silberahorn | 21,5 |
| 2 | *F. pennsylvanica/americana* | Grün-Esche/Weiß-Esche | 19,4 |
| 3 | *A. platanoides* | Spitzahorn | 17,9 |
| 4 | *G. triacanthos* | Gleditschie | 9,0 |
| 5 | *Prunus* spp. | Kirschen, Pflaumen, Pfirsiche | 5,4 |
| 6 | *A. saccharum* | Zuckerahorn | 4,1 |
| 7 | *Tilia* spp. | Linden | 4,0 |
| 8 | *U. americana* | Amerikanische Ulme | 3,8 |
| 9 | *U. pumila* | Zwerg-Ulme | 1,8 |
| 10 | *Q. rubra/velutina* | Roteiche/Schwarzeiche | 1,4 |

*2.2 Zuwachsgänge. Übersicht über funktionelle Gruppen*

Hinsichtlich des Wachstums können sowohl bei Forst- als auch bei Stadtbäumen zwei Gruppen unterschieden werden, langsam wüchsige, schattentolerante Arten und schnellwüchsige, lichtbedürftige Arten. Zu der erstgenannten Gruppe zählen die Eichen, Kastanien, Buchen und Linden. Dagegen sind Arten wie Robinie, Platane und Götterbaum schnellwüchsige Pionierarten,

welche sich leicht an neuen Standorten etablieren können und in kurzer Zeit viel Biomasse bilden.

*2.3 Modellhafte Abbildung von Struktur und Dynamik*

Während für viele Waldbaumarten wie Fichte und Buche zahlreiche Studien zu allometrischen Beziehungen des Wachstums existieren, sind Studien zu den Wachstumsreaktionen von Stadtbaumarten erst seit wenigen Jahren für einzelne Arten verfügbar. Zu nennen sind hier die Arbeiten von Peper et al. (2001) über mediterrane Arten in Kalifornien, Peper et al. (2014) zur Esche, Semenzato et al. (2011) zu fünf südeuropäischen Arten, Rust (2014) zu sechs mitteleuropäischen Arten sowie Larsen & Kristoffersen (2002) und Moser et al. (2015) zu Linden bzw. Winterlinde und Robinie die alle das Wachstum und Strukturentwicklung von Stadtbäumen untersuchen. Tabelle 2 und Abbildung 2 zeigen beispielhaft die Ergebnisse zweier Studien zu den Wachstumsreaktionen verschiedener Stadtbaumarten. Diese Informationen sind insbesondere für Stadtplaner für Bedeutung, um für einen Standort geeignete Baumarten zu finden, die den vorhandenen Wuchsraum optimal besetzen.

*Tabelle 2: Entwicklung des Stammdurchmessers (DBH), der Höhe, der Kronengröße und des Kronendurchmessers der Weiß-Esche (F. americana) und Grün-Esche (F. pennsylvanica) über das Alter verändert nach Peper et al. (2014)*

*Table 2: Development of diameter at breast height, tree height, crown height and crown diameter over age of Fraxinus americana and Fraxinus pennsylvanica adapted from Peper et al. (2014)*

| | BHD (cm) | | | Höhe (m) | | | Kronengröße (m) | | | Kronendurchmesser (m) | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | *(dbh (cm))* | | | *(height (m))* | | | *(crown length (m))* | | | *(crown diameter (m))* | | |
| Alter *(age)* | 20 | 30 | 40 | 20 | 30 | 40 | 20 | 30 | 40 | 20 | 30 | 40 |
| *F. americana* | 21,9 | 36,1 | 47,9 | 8,2 | 12,1 | 15,3 | 6,3 | 8,8 | 10,3 | 4,2 | 8,6 | 13,5 |
| *F. pennsylvanica* | 22,0 | 35,3 | 48,3 | 8,3 | 12,1 | 15,8 | 5,6 | 8,9 | 12,0 | 5,4 | 8,5 | 11,6 |
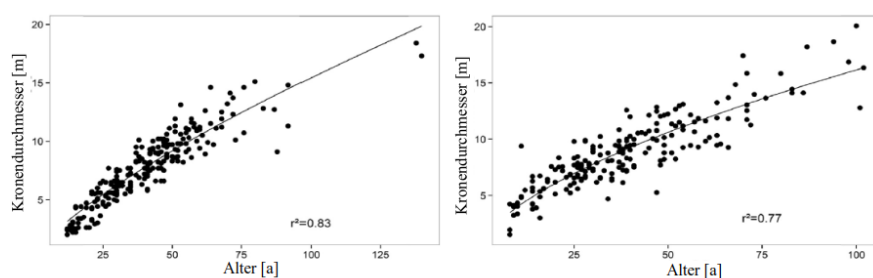


*Abbildung 2: Entwicklung des Kronendurchmessers über das Alter von Winterlinde (links) und Robinie (rechts) aus Moser et al. (2015)*

*Figure 2: Development of crown diameter over age of Tilia cordata (left) and Robinia pseudoacacia (right) from Moser et al. (2015)*

**3 Funktionen und Leistungen von Stadtbäumen**

Funktionen und Leistungen Stadtbäumen sind in den letzten Jahren stark in den Fokus der Forschung und der Öffentlichkeit gerückt. Sie lassen sich in mehrere Kategorien unterteilen: Bereitstellung von Holz, Biomasse, Wasser und Lebensräumen, regulierende Leistungen des Klimas wie z. B. Abkühlungswirkung oder Beschattung, Verbesserung der Luftqualität und Hochwasserschutz sowie kulturelle Leistungen für die Erholung und Lebensqualität und unterstützende Leistungen wie Nährstoffkreisläufe und Photosynthese (Millennium Ecosystem Assessment, 2005) (Abbildung 3).
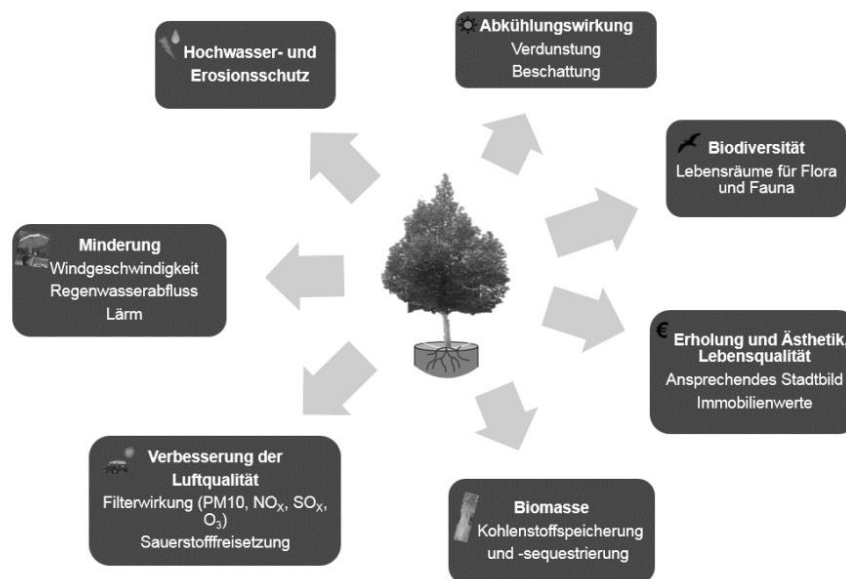


*Abbildung 3: Ökosystemleistungen von Stadtbäumen und Stadtgrün*

*Figure 3: Ecosystem services of urban trees and urban green*

Insbesondere die Reduzierung von thermischem Stress des Menschen werden unter dem Aspekt der Wärmeinsel der Stadt und des Klimawandels intensiv untersucht (Luber & McGeehin, 2008; Sakka et al., 2012). Schon seit einigen Jahren wird mit Pionierprojekten wie dem „Chicago Urban Forest Climate Project" und auch der Software i-Trees (www.itreetools.org) versucht, die sog. Ökosystemleistungen von Stadtbäumen zu quantifizieren und monetär zu bewerten (McPherson et al., 1997; Nowak & Crane, 2002; Nowak et al., 1994). Insgesamt wurde für das untersuchte Grün für das Jahr 2010 ein Wert von $51,2 Milliarden berechnet (Tabelle 3), basierend auf den Ökosystemleistungen der Feinstaubausfilterung, Kohlenstoffspeicherung und

Sequestrierung, Reduzierung des Energiebedarfs von Gebäuden und die Verminderung des $CO_2$-Ausstoßes (Nowak et al., 2013b).

*Tabelle 3: Darstellung der Ergebnisse des Chicago Urban Forest Climate Projects verändert nach Nowak et al.*
*(2013b)*

*Table 3: Results of the Chicago Urban Forest Climate Project adapted from Nowak et al. (2013b)*

| Merkmal *(feature)* | Wert *(value)* |
|---|---|
| Anzahl von Bäumen *(number of trees)* | 157.142.00 |
| Bedeckungsgrad durch Bäume *(tree cover)* | 15,5% |
| Dominante Arten anhand: *(most dominant species by:)* | |
| - Anzahl der Bäume *(number of trees)* | Kreuzdorn (European buckthorn), Grün-Esche (green ash), Eschen-Ahorn (boxelder), Schwarzkirsche (black cherry), Amerikanische Ulme (American Elm) |
| - Blattfläche *(leaf surface area)* | Silberahorn (silver maple), Eschen-Ahorn (boxelder), Grün-Esche (green ash), Kreuzdorn (European buckthorn), Schwarznuss (black walnut) |
| Bäume mit Durchmesser kleiner als 6 Inch (%) *(trees < 6 inches diameter %)* | 73,3% |
| Schadstofffilterung *(pollution removal)* | |
| - Bäume *(trees)* | 18.080 Tonnen/Jahr ($137 Millionen/Jahr) |
| - Bäume und Sträucher *(trees and shrubs)* | 24.170 Tonnen/Jahr ($183 Millionen/Jahr) |
| VOC Ausstoß *(VOC emissions)* | 11.976 Tonnen/Jahr |
| Kohlenstoffspeicherung *(C-storage)* | 16,9 Millionen Tonnen ($349 Millionen) |
| Kohlenstoffsequestrierung *(C-sequestration)* | 677.00 Tonnen/Jahr ($14,0 Millionen/Jahr) |
| Reduzierter Energieverbrauch von Gebäuden *(building energy reduction)* | $44,0 Millionen/Jahr |
| Reduzierte $CO_2$-Freisetzung *(reduced carbon emissions)* | $1,3 Millionen/Jahr |
| Ausgleichswert *(compensatory value)* | $51,2 Milliarden |

*3.1 Kohlenstoffbindung*

Bäume bilden in ihrem Lebenszyklus Biomasse, indem sie aus $CO_2$ und $H_2O$ Zuckerbausteine herstellen, mit denen Blatt-, Wurzel- und Stammmasse aufgebaut werden können (Aguaron & McPherson, 2012). Stadtbäume stellen zwar nur einen kleinen Teil der weltweiten Baumbestände dar, jedoch können auch sie einen Beitrag zur Bindung von $CO_2$ aus der Atmosphäre leisten (Strohbach & Haase, 2012). Einige Forschergruppen haben deshalb versucht, die Kohlenstoffspeicherkapazität der gesamten Vegetation einzelner Städte zu bestimmen, so wurde beispielsweise für Lissabon in Portugal ein Wert von $0.33 pro Baum im Jahr 2003 für die $CO_2$-Reduktion errechnet (Maco et al., 2003; Soares et al., 2011). Auch Davies et al. (2011) haben für die Stadt Leicester, UK geschätzt, dass in der oberirdischen Vegetation innerhalb der Stadt 231.521 Tonnen Kohlenstoff gespeichert ist, was 3,16 kg Kohlenstoff pro Quadratmeter Stadtfläche ausmacht. Dabei sind Bäume mit 97,3 % die Hauptspeicher für Kohlenstoff. Andere Studien wie von Zheng et al. (2013) und Nowak et al. (2013a) haben sich mit dem regionalen Kohlenstoffspeicher der Vegetation beschäftigt. So hat Nowak et al. (2013a) die Kohlenstoffspeicherung- und Sequestrierung von Stadtbäumen in diversen Städten und Staaten der

USA analysiert (Tabelle 4). Auffällig war insbesondere die schwankende Anzahl kronenüber-schirmter Flächen (Baumdichte) einiger Städte, während Atlanta und Gainesville über 50 % Baumdichte aufwiesen, wurden für Casper weniger als 10 % Baumdichte verzeichnet. Aller-dings schlug sich diese Baumdichte nur bedingt in der Kohlenstoffspeicherung und –sequest-rierung nieder (Nowak et al., 2013a).

*Tabelle 4: Standardisierte Kohlenstoffspeicherung und –sequestrierung bezogen auf die Baumdichte in verschie-denen Städten und Staaten der USA verändert nach Nowak et al. (2013a)*

*Table 4: Standardized carbon storage and sequestration estimates per unit of tree cover and percent tree cover in various cities and states in the USA adapted from Nowak et al. (2013a)*

| Stadt/Staat (state) | Speicherung (storage) | | Sequestrierung (brutto) (gross sequestration) | | Sequestrierung (netto) (net sequestration) | | Baumdichte (tree cover) | |
|---|---|---|---|---|---|---|---|---|
| | kg C m$^{-2}$ | SE | kg C m$^{-2}$ year$^{-1}$ | SE | kg C m$^{-2}$ year$^{-1}$ | SE | % | SE |
| Arlington, TX | 6,37 | 0,73 | 0,29 | 0,03 | 0,26 | 0,03 | 22,5 | 0,3 |
| Atlanta, GA | 6,63 | 0,54 | 0,23 | 0,02 | 0,18 | 0,03 | 53,9 | 1,6 |
| Baltimore, MD | 8,76 | 1,09 | 0,28 | 0,04 | 0,17 | 0,03 | 28,5 | 1,0 |
| Boston, MA | 7,02 | 0,96 | 0,23 | 0,03 | 0,17 | 0,02 | 28,9 | 1,5 |
| Casper, WY | 6,97 | 1,50 | 0,22 | 0,04 | 0,12 | 0,04 | 8,9 | 1,0 |
| Chicago, IL | 6,03 | 0,64 | 0,21 | 0,02 | 0,15 | 0,02 | 18 | 1,2 |
| Gainesville, FL | 6,33 | 0,99 | 0,22 | 0,03 | 0,16 | 0,03 | 50,6 | 3,1 |
| Golden, CO | 5,88 | 1,33 | 0,23 | 0,05 | 0,18 | 0,04 | 11,4 | 1,5 |
| Hartford, CT | 10,89 | 1,62 | 0,33 | 0,05 | 0,19 | 0,05 | 26,2 | 2,0 |
| Jersey City, NJ | 4,37 | 0,88 | 0,18 | 0,03 | 0,13 | 0,04 | 11,5 | 1,7 |
| Lincoln, NE | 10,64 | 1,74 | 0,41 | 0,06 | 0,35 | 0,06 | 14,4 | 1,6 |
| Los Angeles, CA | 4,59 | 0,51 | 0,18 | 0,02 | 0,18 | 0,02 | 20,6 | 1,3 |
| Milwaukee, WI | 7,26 | 1,18 | 0,26 | 0,03 | 0,18 | 0,03 | 21,6 | 1,6 |
| Minneapolis, MN | 4,41 | 0,74 | 0,16 | 0,02 | 0,08 | 0,05 | 34,1 | 1,6 |
| New York, NY | 7,33 | 1,01 | 0,23 | 0,03 | 0,12 | 0,03 | 20,9 | 1,3 |
| Omaha, NE | 14,14 | 2,29 | 0,51 | 0,08 | 0,40 | 0,07 | 14,8 | 1,6 |
| Philadelphia, PA | 6,77 | 0,90 | 0,21 | 0,08 | 0,15 | 0,02 | 20,8 | 1,8 |
| Sacramento, CA | 7,82 | 1,57 | 0,38 | 0,06 | 0,33 | 0,06 | 13,2 | 1,7 |
| San Francisco, CA | 9,18 | 2,25 | 0,24 | 0,05 | 0,22 | 0,05 | 16 | 2,6 |
| Syracuse, NY | 8,59 | 1,04 | 0,29 | 0,03 | 0,20 | 0,04 | 26,9 | 1,3 |
| Washington, DC | 8,52 | 1,04 | 0,26 | 0,03 | 0,21 | 0,03 | 35 | 2,0 |
| Indiana | 8,80 | 2,68 | 0,29 | 0,08 | 0,27 | 0,07 | 20,1 | 3,2 |
| Kansas | 7,42 | 1,30 | 0,28 | 0,05 | 0,22 | 0,04 | 14 | 1,6 |
| Nebraska | 6,67 | 1,86 | 0,27 | 0,07 | 0,23 | 0,06 | 15 | 3,6 |
| North Dakota | 7,78 | 2,47 | 0,28 | 0,08 | 0,13 | 0,08 | 2,7 | 0,6 |
| South Dakota | 3,14 | 0,66 | 0,13 | 0,03 | 0,11 | 0,02 | 16,5 | 2,2 |
| Tennessee | 6,47 | 0,50 | 0,34 | 0,02 | 0,30 | 0,02 | 37,7 | 0,8 |

Die verwendeten Methoden zur Bestimmung des gespeicherten Kohlenstoffs sind dabei sehr unterschiedlich und oftmals ungenau. Zum einen kann die gespeicherte Menge an Kohlenstoff über die Biomasse der Vegetation mit Hilfe von allometrischen Beziehungen berechnet werden (Aguaron & McPherson, 2012; Moser et al., 2015; Yoon et al., 2013). Allometrische Gleichun-gen für die Stadtvegetation sind jedoch kaum vorhanden, weshalb oftmals auf Gleichungen für Forstbestände zurückgegriffen werden muss (Aguaron & McPherson, 2012). Zum anderen kön-nen Softwaretools wie i-Tree verwendet werden, das auf Basis von vermessenen Plots die Koh-lenstoffspeicherung der gesamten Stadtvegetation berechnet oder dazu Baumkatasterdaten ver-

wendet (Nowak & Crane, 2002). Andere nicht regional aufgelöste Studien bestimmen die Kohlenstoffspeicherung einzelner Bäume oder Baumkompartimente, siehe beispielsweise Yoon et al. (2013), Moser et al. (2015), Timilsina et al. (2014) und McPherson (1998). Solche Studien auf Baumebene sind jedoch lokal und in ihrem Umfang begrenzt.

*3.2 Kühlung und Beschattung*

Angesichts des Klimawandels und der städtischen Wärmeinsel ist die Abkühlungswirkung und Beschattung durch Stadtbäume eine der wichtigsten und derzeit am meisten erforschten Umweltleistungen. Bäume kühlen ihre Umgebung durch Evapotranspiration und reduzieren die Umgebungstemperatur durch Beschattung (Akbari et al., 2001; Rahman et al., 2011). Durch die Belaubung wird Sonnenlicht und Strahlung abgehalten, die einfallende Sonnenenergie wird von den Blättern zur Photosynthese und Transpiration genutzt (Armson et al., 2013a). Folglich wird im Vergleich zu bebauten Bereichen die umgebende Luft stärker abgekühlt (Leuzinger et al., 2010; Pauleit & Duhme, 2000). Studien zeigen, dass die Blatttemperatur 11-30°C kühler sein kann als umgebende bebaute Flächen (Armson et al., 2013a; Leuzinger et al., 2010; Pauleit & Duhme, 2000). Dabei werden die größten Unterschiede zur Mittagszeit während heißen, sonnigen Tagen erreicht, wobei insbesondere Bäume verglichen mit Gras und begrünten Dächern die größten Effekte erzielen (Armson et al., 2013a; Leuzinger et al., 2010; Pauleit & Duhme, 2000). Die einfallende UV-Strahlung kann von großen, dichten Kronen bis zu 90 % reduziert werden (Armson et al., 2013a). Trotz der großen Relevanz ist die Quantifizierung der Transpirationsleistung und Abkühlungswirkung durch Bäume äußerst schwierig (Ennos et al., 2014). In den letzten Jahren wurden vermehrt Blattporometer (Rahman et al., 2011) und Saftflussmessungen (Pataki et al., 2011; Peters et al., 2011) verwendet. Rahman et al. (2017) haben beispielsweise die Abkühlungsleistung bzw. den Energieentzug aus der Atmosphäre von Winterlinden an zwei unterschiedlichen Plätzen in München untersucht (Abbildung 4) und herausgefunden, dass Winterlinden aufgrund ihrer Wasserstressstrategie eine gute Abkühlungswirkung auch bei Trockenheit aufweisen, dieser Wasserverlust jedoch das Wachstum reduziert (Moser et al. in Begutachtung). Deshalb sind Winterlinden bei Trockenperioden nur bedingt als Stadtbäume zu empfehlen, denn um langfristig Wachstums- und Vitalitätseinbußen zu verhindern, kann Bewässerung notwendig werden.
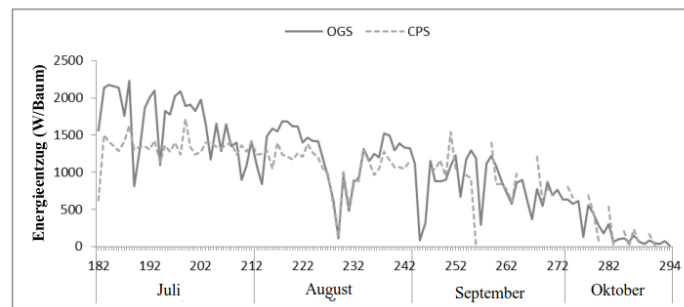
*Abbildung 4: Energieentzug aus der Atmosphäre für die Verdunstungsleistung von Winterlinden in München,*
*verändert nach Rahman et al. (2017)*

*Figure 4: Energy loss for the cooling ability of Tilia cordata in Munich, adapted from Rahman et al. (2017)*

Mit diesen Messungen erzielte Ergebnisse haben gezeigt, dass die Abkühlungswirkung in Parks mit knapp 1°C im Jahresdurchschnitt im Vergleich zur bebauten Referenzfläche relativ gering ist (Bowler et al., 2010). Dies ist jedoch vor allem mit der Durchmischung der Lufttemperatur durch Wind zu begründen (Ennos et al., 2014), denn die tatsächliche Abkühlungswirkung ist um ein vielfaches höher.

Die Kühlungsleistung der Stadtvegetation und die Effekte auf das Stadtklima sind besonders wichtig für den thermischen Komfort von Menschen. So hat die Beschattungs- und Abkühlungswirkung Einflüsse auf die PET (physiologically equivalent temperature) (Matzarakis et al., 2007). Für die Berechnung der PET wird das „Munich energy balance model for individuals" Höppe (1993) und die Lufttemperatur eines standardisierten Innenraums für eine standardisierte Person verwendet (Matzarakis & Amelung, 2008). Menschen im Schatten von Bäumen haben einen geringeren Hitzestress, sie fühlen sich besser und kühler als im Vergleich zu sonnigen Standorten.

### 3.3 Feinstaubausfilterung, Lärmminderung und Windschutz

Neben der Kohlenstoffbindung und der Abkühlungswirkung erbringen Stadtbäume noch weitere Umweltleistungen, wie die Ausfilterung von Feinstaub und Lärm oder Windschutz. Feinstaubbelastung und Lärm haben gemeinsame Ursachen und gesundheitliche Folgen wie Herz-Kreislauferkrankungen, Herzinfarkt, hoher Blutdruck (Gan et al., 2010; Jarup et al., 2008; Ross et al., 2011; Tonne et al., 2007) zur Folge. Feinstaub- und Lärmbelastungen entstehen insbesondere durch anthropogene Prozesse wie Industrieanlagen und Autoabgase, speziell durch die Verbrennung von Dieselkraftstoffen. Feinstaub besteht aus einer Mischung von

Schwermetallen, Kohlenstoffen, aromatischen Kohlenwasserstoffen und weiteren in Luft ge-
lösten Substanzen (Honold, 2016). Diese können weitere negative Folgen nach sich ziehen wie
Atemwegserkrankungen. Deshalb ist besonders für Risikogruppen (Kleinkinder, Senioren) die
Minimierung des Reizes, d.h. die Ausfilterung von Feinstaub von großer Bedeutung (Honold,
2016). Auch Lärm kann gesundheitliche Folgen haben, zu nennen sind vor allem psychologi-
sche Belastungen, die zu einer beeinträchtigen Leistungsfähigkeit, Konzentrationsstörungen
und Schlafstörungen führen aber auch körperliche Folgen wie Bluthochdruck und hormonelle
Störungen nach sich ziehen können. Einige Länder haben Lärm als gesundheitliche Belastung
erkannt und Strategien zur Minimierung entwickelt. So hat Schweden als Langzeitziel die Re-
duzierung der Lärmbelastung auf maximal 55 dB(A) in Außenbereichen und 30 dB (A) in Ge-
bäuden festgelegt (Bluhm & Eriksson, 2011; Bolund & Hunhammar, 1999; Naturvârdsverket,
1996). Auch die Erhöhung des Anteils unbefestigter Oberflächen und Vegetation können das
Lärmniveau verringern. Dabei vermindert und dämpft Vegetation auch die visuellen Reize von
Verkehr, wodurch sie als weniger störend empfunden werden. Hierbei haben immergrüne
Bäume aufgrund ihrer durchgehenden Belaubung Vorteile (Bolund & Hunhammar, 1999).

Ross et al. (2011) haben in einer Studie den Zusammenhang zwischen Luftverschmutzung,
Lärm und Windgeschwindigkeit untersucht. Hierbei zeigten sich eine Zunahme von Lärm mit
mehr Wind und eine Abnahme von Stickstoffoxiden sowie Kohlenwasserstoffen mit höheren
Windgeschwindigkeiten. Auch wurden die täglichen Verläufe von Feinstaub und Lärm analy-
siert, wobei Lärm von morgens bis Abend unterschiedliche Verläufe aufweisen kann, während
die Feinstaubbelastung zumeist bis zum späten Vormittag den höchsten Stand erreichte und
danach absank.

Die Bedeutung von Stadtbäumen für die Feinstaubausfilterung wird von Forschern jedoch kont-
rovers diskutiert (McPherson et al., 1998; Nowak et al., 2006; Sæbø et al., 2012), insbesondere
im Hinblick auf Kosteneffizienz. In Shanghai, China wurde eine Abnahme von Feinstaub,
Stickstoffoxiden und Schwefeldioxiden in der Nähe eines Waldstandortes gemessen (Yin et al.,
2011). In England konnte eine Abnahme der Feinstaubbelastung um bis zu 4 % gefunden
(Ennos et al., 2014; McDonald et al., 2007). Nowak et al. (1994) hat in einer Studie zu den
Umweltleistungen der Stadtbäume Chicagos die monatliche Feinstaubausfilterung untersucht
und dabei insbesondere in den Sommermonaten Juni, Juli und August die höchsten Filterwir-
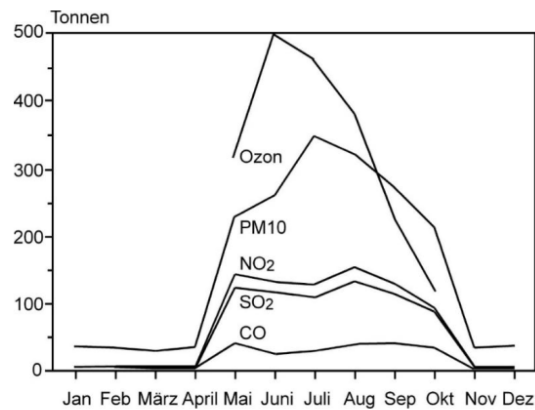kungen gefunden (Abbildung 5).

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*Abbildung 5: Monatliche Feinstaubausfilterung der untersuchten Bäume in Chicago in 1991 verändert nach*

*Nowak et al. (1994)*

*Figure 5: Monthly particulate matter filtering of urban trees in Chicago in 1991 adapted from Nowak et al. (1994)*

Insgesamt zeigen Nadelbäume aufgrund ihrer höheren Blattfläche und ihrer dauerhaften Belaubung (Stolt, 1982) eine bessere Filterwirkung im Vergleich zu Laubbäumen. Jedoch sind Nadelbäume anfälliger gegenüber Schadstoffbelastungen der Luft, während Laubbäume Gase besser ausfiltern können, weshalb eine Mischung aus Laub- und Nadelgehölzen am sinnvollsten wäre (Bolund & Hunhammar, 1999; Stolt, 1982).

Zur Lärmreduzierung wird empfohlen, Bäume und Sträucher nahe an der Lärmquelle und nicht am Lärmempfänger zu pflanzen, da Laub und Stamm den Lärm zerstreuen, während Boden Lärm absorbiert (Aylor, 1972; Cook, 1978; Nowak & Dwyer, 2007). Auf diese Weise können bis zu 50% des Lärms reduziert werden, was 6 bis10 dB (A) entspricht (Cook & Van Haverbeke, 1971; Nowak & Dwyer, 2007).

*3.4 Abflussminderung*

In stark bebauten Gebieten mit asphaltierten Straßen und Wegen ist der Wasserfluss deutlich verändert verglichen mit ländlichen Gegenden. Insbesondere Starkregen verursacht große Mengen an abfließenden Oberflächenwasser, welches die Gefahr von Überflutungen (überflutete Straßen, vollgelaufene Keller usw.) deutlich erhöht. Ebenso sinkt die Wasserqualität durch aufgenommene Schmutzpartikel, Abflusssysteme können aufgrund größerer Verunreinigungen leicht verstopfen, was die Gefahr von Überflutungen weiter steigert. Zudem bedingt die hohe Flächenversiegelung ein Absinken des Grundwasserspiegels (Bolund & Hunhammar, 1999). Vegetation im Stadtbereichen kann dieses Problem deutlich verringern: Zum einen können Bäume, Dachbegrünungen und Grasflächen durch Versickerung Regenwasser auffangen. Daneben kann Vegetation das vorhandene Wasser auch aufnehmen und für Stoffwechselprozesse

wie z. B. Transpiration verwenden. Diese Prozesse reduzieren somit den Abfluss und Minimie-ren die Gefahr von Überflutungen (Bolund & Hunhammar, 1999; Ennos et al., 2014). Die Re-duzierung des Abflusses durch Vegetation wurde bisher aus Daten extrapoliert, die von klein-räumigen Experimenten stammen (Armson et al., 2013b; McPherson et al., 1994). Die Ergeb-nisse dieser Studien haben gezeigt, dass 60 % des Regenwassers auf asphaltierten Flächen di-rekt in die Kanalisation abfloss, während der Abfluss von Grasflächen fast vollständig aufge-nommen wurde (Abbildung 6). Damit kann durch Bäume und offene Flächen der Regenwas-serabfluss im Vergleich zu asphaltierten Flächen deutlich reduziert werden (Armson et al., 2013a; Ennos et al., 2014).
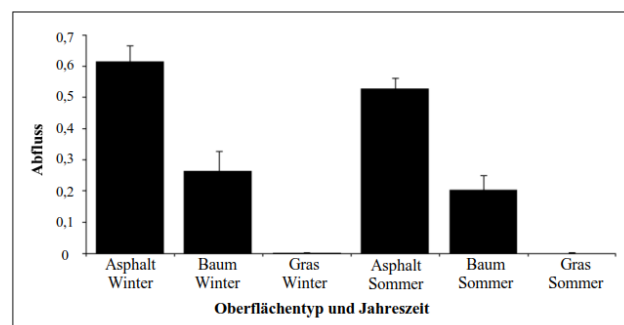


*Abbildung 6: Einfluss von Oberflächentyp und Jahreszeit auf den Abfluss der Experimentalflächen verändert nach Armson et al. (2013a) aus Ennos et al. (2014)*

*Figure 6: Effect of surface type and season on the runoff coefficients of the experimental plots adapted from Armson et al. (2013) of Ennos et al. (2014)*

*3.5 Biodiversität*

Die urbane Umgebung stellt aufgrund vielfältiger anthropogener Einflüsse, Schutz- und Nah-rungsmangel einen extremen Lebensraum für Tier- und Pflanzenarten dar. Viele Arten sind aufgrund ihrer Anpassungsfähigkeiten inzwischen häufige Stadtbewohner. Bekannte Beispiele ländlicherer Gegenden sind Turmfalken, die in Kirchtürmen brüten, nistende Mauersegler an Hauswänden oder Sperlinge an öffentlichen Plätzen. In größeren Städten sind Arten wie Krä-hen, Tauben, Amseln, Meisenarten und Kleinsäuger wie Marder und Eichhörnchen häufig an-zutreffen. Studien wie von Blair (1996), Melles et al. (2003) oder Sandström et al. (2006) haben sich mit dem Vorkommen von Vogelarten im städtischen Bereich beschäftigt. So hat Sandström et al. (2006) beschrieben, dass im Vergleich zu ländlicheren Habitaten in Städten mehr exoti-sche Arten zu finden sind. Ebenso war ein Trend zu erkennen, dass mit steigendem Vorkommen von Bäumen mehr Vogelarten vorhanden sind, und daher mit abnehmender Urbanität und lo-

cker bebauten Wohngebieten mit größeren Baumbeständen die Zahl der vorhandenen Vogelarten deutlich anstieg. Insgesamt ist in diesem Bereich der Stadtbaumforschung jedoch nur wenig Wissen vorhanden. Aktuelle Studien der TU München zur Biodiversität (Vogelvorkommen in urbanen Bereichen) und der Landesanstalt für Wein- und Gartenbau Veitshöchheim (Insektendiversität an Stadtbäumen) versuchen vorhandene Wissenslücken zu schließen (www.lwg.bayern.de).

Projekte wie „Animal Aided Design" der TU München versuchen darüber hinaus zu erforschen, wie sich städtebauliche Anforderungen und die Bedürfnisse von Tieren zusammenführen lassen (Hauck & Weisser, 2015). So wird beispielsweise erprobt, ob durch Fassadendämmungen Nistplätze für Sperlinge geschaffen werden können. Anhand der Ergebnisse solcher Projekte lassen sich Handlungsanweisungen für die Städteplanung generieren, um die Biodiversität in Städten erhalten und fördern zu können.

*3.6 Ästhetik, Lebensqualität*

Vegetation kann ein Stadtzentrum oder einen öffentlichen Platz attraktiver gestalten (Ennos et al., 2014). Neben den bisher beschriebenen Leistungen der Stadtvegetation bieten insbesondere Parks Räume für ästhetisches Erleben und die Möglichkeit, dem hektischen Leben der Stadt kurzzeitig zu entkommen. Dieser Aspekt der Vegetation in Städten wird wahrscheinlich von der Bevölkerung mit am wichtigsten eingestuft (Bolund & Hunhammar, 1999). Dabei spielen insbesondere das Vorkommen von Fauna wie beispielsweise Vögel und Fische eine große Bedeutung. Auch das Vorkommen von Flechten wird positiv bewertet, da dieses ein Indikator für eine gute Luftqualität ist (Miller, 1994). So ist zum Beispiel für die Einwohner der Stadt Stockholm ein hoher Anteil von Grünflächen besonders wichtig. 90 % der Einwohner besuchen Stadtparks mindestens einmal im Jahr Stadtparks, 45 % besuchen wöchentlich einen Park und 17 % sogar mehr als dreimal pro Woche. Zudem und sind die Einwohner Stockholms auch bereit, für Parks in unmittelbarer Nähe höhere Mieten zu bezahlen (Bolund & Hunhammar, 1999; Stadbyggnadskontoret, 1994; Transek, 1993).

In Bezug auf positive Aspekte von Stadtbäumen auf die Lebensqualität haben Donovan et al. (2011) den Zusammenhang zwischen der Anzahl von Bäumen in unmittelbarer Nähe und dem Ausgang von Schwangerschaften untersucht. Dabei wurde eine positive Relation gefunden, eine Erhöhung der Baumdichte um 10 % in einer Distanz von 50 m zu einem Wohnhaus reduziert die Anzahl von Frühgeburten um 1,42 bei 1000 Geburten (Donovan et al., 2011). Die in dieser Studie gezeigten Zusammenhänge waren jedoch vergleichsweise niedrig, Studien von

Maas et al. (2009) oder Townshend & Lake (2009) haben jedoch gezeigt, dass die Verfügbarkeit von großen Grünflächen zu einer besseren sozialen Einbettung, vermehrter körperlicher Aktivität und somit weniger Frühgeburten führt (Donovan et al., 2011)

*3.7 Ansätze zur monetären Bewertung der Funktionen und Leistungen von Stadtbäumen*

Um den Wert von Stadtbäumen besser bewerten zu können, wurden Ansätze zur monetären Bewertung entwickelt. In dem „Chicago Urban Forest Climate Project" wurde eine Kosten-Nutzen-Rechnung für eine typische Stadtbaumart, die Grün-Esche *Fraxinus pennsylvanica* aufgestellt. Dabei wurden 95.000 Bäume an fünf verschiedene Pflanzstandorten verglichen. Nach Abzug von Pflanzungs- und Unterhaltskosten wurde ein Wert von $38 Millionen insgesamt oder $402 pro Baum berechnet (McPherson et al., 1994).

Andere Forscher haben einzelne Aspekte der Umweltleistungen in geldwerte Beträge umgerechnet. So hat Rosenfeld et al. (1998) monetäre Ersparnisse durch die Smogreduktion, Feinstaubfilterung und Abkühlungswirkung von gepflanzten Bäumen auf $273 Millionen pro Jahr geschätzt und die jährliche Wirtschaftsleistung der Stadtbäume in Los Angeles auf $211 pro Baum beziffert (Akbari et al., 2001). McPherson et al. (1997) hat die Kostenersparnis durch die Kühlung eines hohen Baumes auf verschiedene Gebäudetypen in Chicago untersucht (Abbildung 7). Die meisten Ersparnisse wurden dabei von einem Baum vor einem dreistöckigen Mauerhaus erzielt (McPherson et al., 1997).
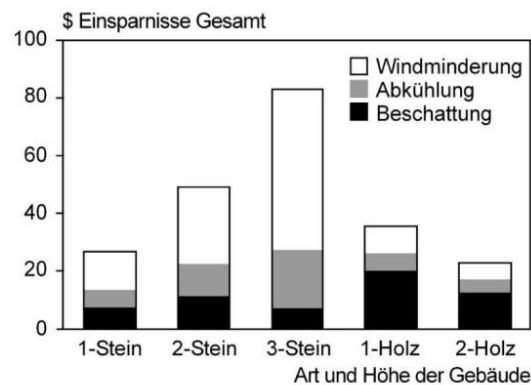


*Abbildung 7: Heizungs- und Klimaanlagenersparnisse sowie Ersparnisse durch Windverringerung pro Baum für ein-, zwei- und dreistöckige Holz- und Mauersteinhäuser in Chicago. Die Ersparnisse wurden anhand eines 25 Fuß großen Laubbaum in Westrichtung der Häuser berechnet, verändert nach McPherson et al. (1997)*

*Figure 7: Simulated annual heating and cooling energy savings per tree for one-, two-, and three-story brick and woodframe buildings. Shading savings are from a 25-foot tall deciduous tree opposite the west wall of each building adapted from McPherson et al. (1997)*

Der Wert durch die Kohlenstoffspeicherung von Stadtbäumen (Punkt 3.1) beläuft sich nach Nowak et al. (1994) für Chicago auf insgesamt $51,2 Milliarden.

## 4 Risiken durch Stadtbäume und gegensteuernde Maßnahmen zu ihrer Vermeidung

Neben den genannten positiven Leistungen entstehen durch Stadtbäume auch Risiken und Schäden. Sie sind jedoch ebenso wie die positiven Umweltleistungen abhängig von der gepflanzten Art und des Zustandes bzw. der Vitalität eines Baumes. An vorderster Stelle müssen Sach- und Personenschäden durch abbrechende und herabfallende Pflanzenteile, vor allem Äste genannt werden. Zudem können Bäume Krankheiten und Schadinsekten beherbergen (Carinanos & Casares-Porcel, 2011; Dobbs et al., 2011). Auch Frucht- und Laubfall, herunterfallende Pollen und Pflanzensäfte führen oft zu Ärgernissen und teils hohen Kosten für Reinigung u. ä. Falsch gepflanzte Arten können zu Schäden an Asphalt und Wegen führen sowie Lichteinfall in Wohnungen verhindern. Daneben wird durch die Transpiration der Bäume die Luftfeuchtigkeit erhöht. Seit einigen Jahren ist auch die Emission von sog. biogenen flüchtigen organischen Verbindungen (biogenic volatile organic compounds = BVOCs) von Stadtbäumen in den Fokus der Forschung gerückt (Calfapietra et al., 2013; Carinanos & Casares-Porcel, 2011; Gregg et al., 2003; Grote, 2009; Grote et al., 2013). Bei Stress erzeugen Bäume BVOCs, was wiederum die Ozonbildung begünstigen und somit Smogprobleme verstärken kann (Carinanos & Casares-Porcel, 2011; Domm et al., 2008). Weiterhin können bestimmte Baumarten allergische Reaktion wie Asthma und Heuschnupfen auslösen (Dobbs et al., 2011; Escobedo et al., 2011; Lyytimäki & Sipilä, 2009; Nowak & Dwyer, 2000).

### 4.1 Allergene

Die Freisetzung von Blütenstaub während der Blütezeit ist die mit am häufigsten genannten negativen Eigenschaften von Stadtvegetation: 30 - 40 % der Weltbevölkerung reagiert allergisch auf Blütenstaub (Carinanos et al., 2014; Pawankar et al., 2011). Auch ist die Wahrscheinlichkeit an einer Pollenallergie zu leiden bei Stadtbewohnern um 20 % höher als von Menschen, die auf dem Land wohnen (Carinanos & Casares-Porcel, 2011). Dies hat enorme direkte Auswirkungen wie Kosten für Ärzte, Medizin und Untersuchungen, indirekte Auswirkungen wie verpasste Arbeits- und Schulzeit, frühzeitiger Ruhestand und immaterielle Auswirkungen wie ein Verlust der Lebensqualität (Carinanos et al., 2014; Green & Davis, 2005; Reed et al., 2004). In den USA wurden die ausschließlich aus den direkten Auswirkungen resultierenden Kosten auf $2 bis $5 Milliarden pro Jahr geschätzt (Reed et al., 2004). Carinanos et al. (2014) haben

einen Index zur Bestimmung der Allergenität von Stadtbäumen entwickelt, welche das allergische Potential, die Bestäubungsstrategie und die Dauer des Bestäubungszeitraums, Baumgröße und Anzahl der Individuen pro Art berücksichtigt. Auch das allergische Potential verschiedener Arten wurde bewertet, wobei die höchsten Risiken bei mehreren *Cupressus*-Arten, *Olea europaea* und *Platanus hispanica* gefunden wurden (Carinanos et al., 2014; Hruska, 2003). Insbesondere bei Platanen stellt dies große Probleme dar, da diese eine weltweit häufig vorkommende Stadtbaumart ist. Die Platane ist windbestäubt und kann 13 x $10^6$ Pollen pro Infloreszenz produzieren (Carinanos & Casares-Porcel, 2011; Tormo Molina et al., 1996). Deshalb wird die Platane als einer Hauptgründe für Pollenallergien im städtischen Bereich betrachtet (Carinanos & Casares-Porcel, 2011; Sabariego-Ruiz et al., 2008).

*4.2 Herabfallende Baumteile und Verschmutzung durch Laub*

Von der Bevölkerung werden Verschmutzungen durch herabfallende Baumteile wie Laub, Blütenstaub- und Blätter, Samen und Zweige negativ gesehen. Neben der Erzeugung von Abfall durch herabfallende Baumteile sind herabtropfende, klebende Pflanzensäfte, insbesondere auf Asphalt und Autos, negative Eigenschaften von Stadtvegetation (Lohr et al., 2004). Darüber hinaus können größere herabfallende Äste, durch Sturm oder falsche Pflanz- und Schnitttechniken umfallende Bäume zu Sach- und Personenschäden führen (Dobbs et al., 2014; Escobedo et al., 2011).

*4.3 Emission von flüchtigen organischen Verbindungen*

Biogene flüchtige organische Verbindungen wie Isoprenoide, Monoterpene, Alkohole, Aldehyde, Ketone und Säuren werden von Bäumen in die Atmosphäre abgegeben (Grote, 2009). Sie bestehen aus Fetten, Harzen und anderen pflanzlichen Produkten. Die Emission von BVOCs variiert je nach Art, Lufttemperatur und anderen Umwelteinflüssen. Funktionen von BVOCs für Pflanzen sind Hitzeschutz und Schutz vor Fressfeinden bzw. Anziehung von Bestäubern (Grote et al., 2013; Guenther et al., 1994; Kramer & Kozlowski, 1979; Nowak & Dwyer, 2007; Sharkey & Singsaas, 1995). Problematisch sind BVOCs, da sie zu der Bildung von Ozon und Kohlenmonoxid beitragen können. Es wird jedoch angenommen, dass eine höhere Anzahl von Bäumen die Emission von BVOCs und damit Ozon verringert. Dies wird mit der Temperaturabhängigkeit der BVOCs und der Abkühlungswirkung von Bäumen begründet (Brasseur & Chatfield, 1991; Nowak & Dwyer, 2000).

*4.4 Kosten von Stadtbäumen*

Neben den genannten Nachteilen von Stadtbäumen entstehen für Pflanzung und Pflege auch Kosten. Dazu zählen Kosten für die Bewässerung, den Schnitt und die Entfernung von schadhaften oder toten Individuen. Hinzu kommen Aufwendungen für Neupflanzungen für die Verwaltung und für die Instandsetzung von beschädigter Infrastruktur wie Gehsteigen oder des Asphaltbelags (Roy et al., 2012). Soares et al. (2011) hat die Kosten für die Baumpflege, Administration u. ä. auf insgesamt $1,9 Millionen beziffert, wobei sich die Gesamtkosten pro Baum auf $45,6 belaufen (Tabelle 5).

*Tabelle 5: Jährliche Kosten für die Baumpflege, Verwaltung und sonstige Kosten in Lissabon, Portugal verändert nach Soares et al. (2011)*

*Table 5: Yearly costs for tree management, administration and other costs in Lisbon, Portugal adapted from Soares et al. (2011)*

| Kosten *(costs)* | $ Gesamt *($ total)* | $ pro Baum *($ per tree)* |
|---|---|---|
| Baumpflege *(tree management)* | 1.121.723 | 29,50 |
| Administration *(administration)* | 409.600 | 2,93 |
| Weitere Kosten *(other costs)* | 256.000 | 6,20 |
| **Gesamtkosten** *(total)* | **1.882.323** | **45,64** |

## 5. Klimawandel

Klimawandel ist ein weltweit auftretendes, international anerkanntes Phänomen, das sich vor allem in dem Auftreten von hohen Temperaturen mit veränderter Niederschlagverteilung widerspiegelt. Auch können extreme Wetterereignisse wie Trockenperioden, Überschwemmungen und Stürme können mit höherer Frequenz und Intensität auftreten (Della-Marta et al., 2007; IPCC, 2007). Die Auswirkungen des Klimawandels können den Wärmeinseleffekt in der Stadt zusätzlich verstärken und die Lebensbedingungen für Stadtbewohner sowie die Wachstumsbedingungen für Stadtbäume deutlich verschlechtern (Defra, 2012; Gill et al., 2013). Diese Zusammenhänge sind jedoch unzureichend untersucht. Gegensteuernde Maßnahmen, um die negativen Effekte des Klimawandels auf das Stadtklima abzumildern, werden dringend benötigt.

*5.1 Versiegelung und Wärmeinseleffekt - Reaktionen von Stadtbäumen auf Klimawandel*

Während die Zuwachsreaktionen von Waldbäumen und -beständen auf den Klimawandel bereits relativ gut untersucht wurden, ist über die Reaktionen von Stadtbäumen, die sowohl dem Klimawandel als auch der städtischen Wärmeinsel ausgesetzt sind, viel weniger bekannt. Dahlhausen et al. (2016) und Pretzsch et al. (2015a) führten Bohrkernentnahmen und -analysen an etwa 1600 Bäumen in 10 Metropolen in borealen bis subtropischen Klimazonen durch. Die

Bohrkerne wurde jeweils von einer Baumart je Stadt gewonnen, von Bäumen im Zentrum bis in den ruralen Bereich (Pretzsch et al., 2015a). Weil die Jahrringchronologien z.T. mehr als 100 Jahre zurückreichen, spiegeln sie sowohl die Wirkung der Klimaänderungen als auch des städtischen Wärmeinseleffekts auf das Baumwachstum wider. Dabei zeigte sich, dass in allen Städten die Bäume gegenwärtig schneller wachsen und früher altern als in der Vergangenheit. Im Stadtzentrum ist dieser Effekt noch stärker ausgeprägt als in den peripheren Zonen. Der Beschleunigungseffekt ist in eher kühl-feuchten und humiden Städten (Sapporo, Price George, Hanoi) stärker ausgeprägt als in mediterranen und temperierten Städten (Kapstadt, Santiago de Chile). Offenbar werden haben die für das Baumwachstum positiven Effekte (Temperaturanstieg, Länge der Vegetationszeit, $CO_2$-Konzentration der Luft) gegenüber den negativen (Reduktion der Photosynthese durch BVOCs und Feinstaub, Zuwachsminderung durch Trockenstress) in den meisten Städten noch die Oberhand (Dahlhausen et al. in Begutachtung). Diese Beschleunigung hat weitreichende Konsequenzen für alle Aspekte der Bewirtschaftung von Stadtbäumen (Pflanzung, Ersatzbeschaffung, räumliche Ausdehnung) und ihre Ökosystemdienstleistungen (C-Bindung, Ästhetik, Luftbefeuchtung).

*5.2 Anpassungsmaßnahmen*

Gegensteuernde Maßnahmen sind nötig, um die Belastung der Stadtbäume durch den Klimawandel und die städtische Wärmeinsel zu verringern, um das Wachstum und die Leistungen von Stadtbäumen zu erhalten.

5.2.1 Pflanzmethode

Sanders et al. (2013) und Sanders & Grabosky (2014) konnten belegen, dass die Größe der Pflanzgrube einen signifikanten Effekt auf das Baumwachstum hat. Bäume mit einer größeren Pflanzgrube wiesen einen größeren Stammdurchmesser und Kronenradius auf. In Deutschland haben die Städte teilweise Verordnungen für die Größe einer Pflanzgrube festgesetzt. So beträgt in Würzburg das Mindestvolumen einer Pflanzgrube für Straßenbäume 12 m³ (Stadt Würzburg, 2014), in München werden je nach Baumgröße 12 m³ bis 36 m³ für die Pflanzgrube vorgeschrieben (Landeshauptstadt München, 2016). Eine größere Pflanzgrube ermöglicht den Bäumen eine bessere Durchwurzelung, bessere Wasser-, Sauerstoff- und Nährstoffverfügbarkeit. Daneben hat auch das verwendete Substrat einen großen Einfluss auf das Wachstum und die Vitalität von Stadtbäumen. Bühler et al. (2007) haben verschiedene Pflanzgruben und Materialien für die Pflanzung von Stadtbäumen in Kopenhagen getestet und dabei eine höhere Vitalität bei sogenannten „super planting pits" gefunden. Diese wiesen im Wurzelbereich keine lasttragenden Materialien und eine große Pflanzgrube von einer Fläche größer als 12 m² auf. Es wurde

lockeres Material verwendet, welches mindestens 15 m³ durchwurzelbaren Bereich für die Bäume aufwies. Insgesamt wiesen die Bäume in Pflanzgruben im lasttragenden Substraten mit durchwurzelbaren Bereichen ein gutes Wurzelwachstum auf (Bühler et al., 2007).

5.2.2 Artenwahl

Die Auswahl der Baumart an einem Standort in der Stadt erfolgt zumeist aus ästhetischen oder räumlichen Gründen. In Deutschland erfolgt die Artenwahl zumeist nach der GALK-Straßen-baumliste (GALK e. V., 2016), welche die Anforderungen, Eigenschaften und Eignungen aller häufigen Straßenbaumarten enthält. Jedoch zeigen viele der verwendeten Arten aufgrund des spezifischen Stadtklimas (hohe Temperaturen, Streusalz, Hundeurin), des lokalen Klimas (Nie-derschlagsmengen, Spätfröste) oder eingeführter Schädlinge (Massaria, Asiatischer Laubholz-bockkäfer) Vitalitäts- und Wachstumseinbußen. Auch reagieren Stadtbaumarten unterschied-lich sensitiv auf Umwelt- und Schadstoffbelastungen (Mertens et al., 2007). Nach Czerniawska-Kusza et al. (2014) kann Streusalz eine Reduktion der Biomasse, Chlorose und Nekrose oder sogar das Absterben des gesamten Baumes bedingen. Pflanzen nehmen das im Streusalz be-findliche $Na^+$ und $Cl^-$ über die Wurzeln bis in die Blätter auf, wobei schnell schädliche Mengen erreicht werden können (Cekstere et al., 2008). Sowohl $Na^+$ als auch $Cl^-$ wirken antagonistisch auf die Nährstoffaufnahme (Marschner, 1995; Neuman et al., 1996), was zu einer verminderten Nährstoffaufnahme bis zu Störungen in physiologischen Prozessen und verminderten Toleranz von anderen wachstumshinderlichen Faktoren führen kann (Cekstere et al., 2010). Die Effekte des Klimawandels können diese Probleme verstärken, was einen kompletten Ausfall typischer Stadtbaumarten nach sich ziehen kann (Tubby & Webber, 2010). Klimatisch schwierige Stand-orte wie Würzburg (hohe Temperaturen, niedrige Niederschlagsmengen, Spätfröste) haben auf-grund dieser Problematik Schwierigkeiten, langfristig geeignete Baumarten für urbane Stand-orte auszuwählen.

An diesem Punkt setzten Projekte wie „Stadtbäume im Klimawandel" der TU München, ein gleichnamiges Projekt der Stadt Hamburg und das Projekt „Stadtgrün 2021" der Landesanstalt für Wein- und Gartenbau an. Während das Projekt der TU München das Wachstum und die Leistungen typischer Stadtbaumarten unter derzeitigen und zukünftigen Klimabedingungen un-tersucht (Pretzsch et al., 2015b), beschäftigt sich die Stadt Hamburg mit dem Wachstum der Stadtbäume im Klimawandel und versucht Anpassungsstrategien zu finden (Behörde für Umwelt und Energie). Das Projekt „Stadtgrün 2021 - Selektion, Anzucht und Verwendung von Gehölzen unter sich ändernden klimatischen Bedingungen" testet dagegen 20 Baumarten auf ihre Eignung als Stadtbaum. Dabei sollen diese Baumarten aufgrund ihrer Eigenschaften in der Lage sein, den prognostizierten Klimabedingungen der Städte zu trotzen (Böll et al., 2014).

4.2.3 Nachhaltigkeitskonzept

Im Forstbereich reguliert die klassische Nachhaltigkeitsplanung auf der Basis des Normalwald-modells den stehenden Vorrat, die Entnahmen, Mortalität und Erneuerung in einer Weise, dass ein kontinuierlicher Holzertrag gewährleistet ist (Speidel, 1972). Das Konzept wurde in den zurückliegenden Jahrzehnten dahingehend erweitert, dass auch die Ökosystemdienstleistungen des Waldes nachhaltig gegeben bleiben (MCPFE, 1993; Millennium Ecosystem Assessment, 2005). Für das Management resultieren daraus klare Regeln für die Ernte, Neubegründung und Pflege von Wäldern.

Für den Baumbestand einer Stadt existieren zwar manchmal Karten, Kataster, Bestandsstatisti-ken oder z.T. auch Modelle (z. B. „i-Tree") für die Quantifizierung von Wirkungen und Leis-tungen der Bäume. Es fehlen bisher aber Modelle für die Nachhaltigkeitsplanung im Sinne einer Gewährleistung nachhaltiger Leistungen und Wirkungen von Stadtbäumen. Wichtigste Kom-ponenten dafür sind der aktuelle Bestand, die Raten des Zuwachses und Ausscheidens von Bäu-men, sowie die Erneuerung durch Ersatz oder Neuanpflanzung. Analog zur Planung im Forst-betrieb könnten solche Modellansätze zeigen, wie ein Stadtbaumbestand aufgebaut und gesteu-ert werden muss (Altersaufbau, Größenverteilung, Baumartenzusammensetzung), um ein ge-wünschtes Spektrum an Ökosystemdienstleistungen kontinuierlich sicherzustellen.

Derartige Modellansätze könnten dazu beitragen, Defizite am Baumbestand zu vermeiden, Be-darf an Pflanzmaterial und Personal zu prognostizieren und insbesondere eine kontinuierliche Ausstattung einer Stadt mit Habitaten, Biodiversität, Beschattung, Kühlung, und erholungs-wirksamem Grün zu verstetigen.

**6. Forschungsperspektiven**

*6.1 Weiterentwicklung von Modellen*

Prozessorientierte Modelle, wie sie für Wald- und Ackerbestände vorliegen, die Wachstum, Ertrag und Umweltdienstleistungen in Abhängigkeit von klimatischen Bedingungen und ande-ren Umweltfaktoren simulieren, liegen für Stadtbäume nicht vor. Bisherige Modelle wie z.B. das i-Tree-Modell basieren auf empirischen Funktionen, die nur bedingt veränderte Verhält-nisse wie etwa die Auswirkungen des Klimawandels berücksichtigen können. Jedoch wären solche physiologischen Modelle vor allem für die Stadtvegetation sehr wichtig, um die Verän-derungen des Wachstums und der Dienstleistungen von Stadtbäumen abschätzen zu können, die neben dem Klimawandel auch von Stadteffekten beeinflusst werden. In einer ersten Version zeigt beispielsweise das neu entwickelte prozessorientierte Modell „CityTree" (Rötzer et al. in

Bearbeitung) den Einfluss von Klima und kleinräumiger Umgebung auf Wachstum und Umweltleistungen (Kohlenstoffspeicherung, Wasserhaushalt, Abkühlung, Beschattung) von Stadtbäumen wie Linden, Robinien oder Rosskastanien (Vargas in Bearbeitung, Mack (2016)). In Zukunft werden solche Modelle wichtige Instrumente für eine vorausschauende Planung von Stadtgrün und deren Management sein.

Für ein nachhaltiges Grünraummanagement ist die Betrachtung auf Einzelbaumebene jedoch nicht ausreichend. Eine exakte und umfangreiche Darstellung des Wachstums und der Leistungen des Baumbestandes ist auch für ganze Straßenzüge bzw. Stadtteile und letztendlich die gesamte Stadt erforderlich. Hierbei gibt es unterschiedliche Ansätze; während die Software i-Trees verschiedene Aspekte des Baumbestandes beleuchtet (Struktur, Leistungen), kann das Modell ENVI-met (http://www.envi-met.com/) den Einfluss grüner Infrastruktur, Bebauungsdichte u. ä. auf das Mikroklima eines Standortes simulieren.

*6.2 Interaktion zwischen Waldbaum- und Stadtbaumforschung*

6.2.1 Nachweis von Existenz und Wachstumspotential

In Ländern, deren Wald aus heimischen Arten stark übernutzt, bzw. in Plantagen oder landwirtschaftliche Flächen umgewandelt wurde (z. B. Vietnam, Südafrika, China) wurden Stadtbäume heimischer Arten mitunter verschont (z. B. *Dipterocarpus*, *Podocarpus*, *Metasequoia*). Belassene alte, große Stadtbäume bilden dann wichtige Anschauungsbeispiele, dass diese Arten dort überhaupt wachsen können, welche Dimensionen sie erreichen, und dass sie als Alternative zu fremdländischen Arten in Frage kommen.

Angesichts des Klimawandels werden im urbanen wie forstwirtschaftlichen Sektor neue Arten auf ihre Trockenheitsresistenz und -resilienz erprobt (z. B. *Pinus nigra*, *Castanea sativa*, *Fraxinus ornus*, *Zelkova serrata*). Beide Seiten können hier von einen Austausch der Erfahrungen profitieren.

6.2.2 Messung und Bestandesüberwachung

Die Messung von Waldbäumen folgt einem internationalen Standard, der seit mehreren Jahrhunderten immer weiter verbessert wurde (Avery & Burkhardt, 1975; Prodan, 1951). Gegenwärtig werden in Wäldern große Fortschritte in innovativen terrestrischer und flugzeuggetragener Fernerkundung erzielt (Radar, LiDAR, stereometrische Digitalbilder), die sich auch für Stadtbäume eignen dürften (Bayer et al., 2013). Aufgrund ihres Freistandes dürften solitäre Bäume, Straßen- und Parkbäume noch leichter in ihrer Struktur, Vitalität, und Entwicklung über klassische und innovative Verfahren zu erfassen sein.

6.2.3 Wuchsgesetzmäßigkeiten

Stadtbäume repräsentieren Wuchsbedingungen, wie sie im Wald selten auftreten. Sie stehen z. B. oft solitär und konkurrenzfrei (Dahlhausen et al., 2016; Pretzsch et al., 2015a), haben wegen des urbanen Wärmeinseleffekts häufig eine verlängerte Vegetationszeit (Rötzer et al., 2000) und leiden wegen begrenztem Wurzelraum häufig unter Trockenstress (Moser et al., 2016). Daraus resultierende Reaktionsmuster, z. B. die maximale laterale Ausdehnung im Freistand, die Wuchsbeschleunigung durch früheres Austreiben und längeres Wachstum im Herbst, oder die Zuwachsreaktion auf Trockenheit ist von größtem Interesse für die Forstwirtschaft und Wissenschaft. Denn damit repräsentieren Stadtbäume ein Wachstum das unter künftiger Bewirtschaftung oder unter künftigen Wuchsbedingungen auch in Wäldern relevant werden dürfte. Potentielle Entwicklungen, wie sie in Wäldern allenfalls auf Solitär- oder Nelderversuchen vorkommen, sind im urbanen Raum die Regel und besonders wertvoll für die Parametrisierung von Wuchsmodellen für Wald- und Stadtbäume, weil sie die Grenzwerte von Wachstum und Struktur repräsentieren.

6.2.4 Modelle

Im Unterschied zur Stadtbaumforschung hat die Forstwissenschaft eine lange Tradition in der biometrischen Modellierung von Baumentwicklungen und Modellierung und Simulation der räumlich-zeitlichen Dynamik auf Baum-, Bestandes- und Regionalebene. Bewährte biometrische Modelle (z. B. für die Höhenentwicklung, Kronenallometrie, Wurzelausdehnung) dürften mit großem Nutzen auf die Quantifizierung, Modellierung und Prognose von Stadtbäumen übertragbar sein. Prozessorientierte Modelle können zudem den Einfluss von Umweltveränderungen (z. B. Klima, kleinräumige Umgebung) aus das Wachstum und die Leistungen von Bäumen aufzeigen. Solche für Waldbestvorhandene Modelle lassen sich einfach für Stadtbäume weiterentwickeln.

6.2.5 Management

Zur Nachhaltigen Versorgung mit Holz und anderen Wirkungen und Leistungen des Waldes wurden das Normalwaldmodell und die Nachhaltigkeitsplanung entwickelt, algorithmisch gefasst und der Praxis in Form von Modellen bereitgestellt (Speidel, 1972). Analoge Algorithmen, Modelle und Leitlinien zur Gewährleistung eines nachhaltigen Baumbestandes in einer Stadt und Kontinuität in den Ökosystemdienstleistungen bedürfen noch der Entwicklung; hier könnten Erfahrungen und Methoden aus der Waldforschung und Forstwissenschaft einfließen.

### 7. Zusammenfassung

Stadtbäume und grüne Infrastrukturen sind ein wesentlicher Bestandteil des Stadtbildes, sie stellen den ästhetischen und erholungswirksamen Mittelpunkt der Stadt dar. Doch nicht nur durch ihre Ästhetik prägen Bäume eine Stadt, sondern auch durch ihre Struktur und Erscheinung sowie die Funktionen, die sie für das Stadtklima erbringen. Neben positiven Eigenschaften wie Kohlenstoffbindung, Abkühlungswirkung, Beschattung, Biodiversität, sowie Abflussminimierung (Abbildung 4) dürfen jedoch auch die negative Eigenschaften des Stadtgrüns wie Kosten für die Pflanzung und Pflege, Allergene und Schäden durch herabfallende Pflanzenteile bzw. Verschmutzungen nicht ignoriert werden. Darüber hinaus ist der Wuchsraum der Stadt durch vielfältige Faktoren gekennzeichnet, die sich negativ auf die Vitalität, das Wachstum und die Leistungen der Bäume auswirken und zu erhöhten Kosten für Pflege und Neupflanzungen führen können. Zu nennen sind hierbei hohen Temperaturen, geringe Wuchsraum-, Nährstoff,- und Wasserverfügbarkeit, Schadstoffbelastungen, Beschattung durch Gebäude und Verletzungen (Abbildung 1). Auch stellen der urbane Wärmeinseleffekt und die Auswirkungen des Klimawandels für Stadtbäume langfristig wachstumshinderliche Faktoren dar, die in der Grünraumplanung immer mehr im Vordergrund stehen. Städte und Forschung versuchen deshalb, nachhaltige Konzepte zu entwickeln, um weiterhin einen gesunden, vielfältigen und attraktiven Baumbestand in der Stadt zu erhalten. Diese Studie beschäftigt sich mit dem aktuellen Stand des Wissens und stellt die positiven und negativen Aspekte der Stadtbäume dar. Die Wirkungen von Stadtbäumen auf den Menschen und das Stadtklima, aber auch die Effekte des Standortklimas auf den Baum werden diskutiert. Dabei werden weiterhin die Folgen des Klimawandels aufgezeigt, sowie moderne Konzepte und zukünftige Forschungsperspektiven dargestellt und analysiert.

### 8. Summary

Title of the paper: *City trees: Growth, functions and services – risks and research prospects*
Urban trees and green infrastructures are a main element of the cityscape; they are the aesthetic and recreational heart of the city. However, not only their aesthetics influence the urban environment but rather their structure and appearance as well as their services for the urban climate. While science focuses mainly on the positive effects of urban trees like carbon storage, cooling, shading, biodiversity and runoff mitigation (Figure 4); negative aspects of urban green such as costs for planting and maintenance, allergenic potential and damages through falling branches further debris need to be mentioned altogether. Moreover, the typical growing site of trees in the city is characterized by numerous factors detrimental for tree growth, vitality and services

leading to increased costs for maintenance and replacement. Those are e.g. high temperatures, limited rooting space, water availability and nutrients as well as pollutants, shading by high buildings and injuries (Figure 1). In addition, the urban heat island effect and climate change can further amplify the negative effects of the urban environment on the growth of urban trees. Therefore these interactions are gaining increased attention of urban planners. City councils and science try to develop sustainable planting concepts and management plans to preserve a vital, diverse, attractive urban tree forests. This review discusses the current state of knowledge and highlights the positive and negative aspects of urban trees. The impacts of urban trees on city dwellers and the urban climate as well as the effects of the growing site on tree vitality are presented. The effects of climate change on tree growth are analyzed and modern concepts and research prospects discussed.

## 9. Résumé

Titre de l'article: *Les arbres urbains : la croissance, les fonctions et les caractéristiques – Les risques et les perspectives de recherche*

Les arbres urbains et les espaces verts sont un élément essentiel du paysage citadin, ils représentent le point central de l´esthétique en donnant un effet de repos à la ville. Cependant, ce n´est pas seulement par leur esthétique que les arbres influencent la qualité d´une ville, mais aussi par leur structure et leur apparence, tout comme par les fonctions qu´ils apportent pour tempérer le climat urbain. À côté des propriétés caractéristiques positives comme le stockage du carbone, les effets du refroidissement, l´ombrage, la biodiversité, ainsi que la diminution d´écoulement, les caractéristiques négatives des espaces verts urbains, comme les coûts de plantation et d´entretien, les allergènes et les dommages causés par la chute de parties végétales ou par la pollution ne doivent pas être ignorées. En outre, l'espace vital urbain est marqué par des facteurs variés, qui retentissent négativement sur la vitalité, la croissance et les caractéristiques des arbres, et qui peuvent mener à l´augmentation des coûts pour l´entretien et les nouvelles plantations. À ce sujet, il y a lieu de citer les températures élevées, les petits emplacements réservés à l´espace vital, les faibles taux de substances nutritives, les pénuries d´eau, les polluants, l´ombrage des bâtiments et les blessures. Aussi, l´effet d´îlot de chaleur urbain et les effets du changement climatique qui prennent de plus en plus d´importance dans la planification des espaces verts représentent à long terme des facteurs gênants pour la croissance des arbres urbains. Ainsi, les villes et la recherche essaient de développer des concepts efficaces pour préserver un peuplement d´arbres sain, varié et attrayant dans la ville. Cette étude s'occupe de l'état actuel des connaissances et présente les aspects positifs et négatifs des arbres en milieu

urbain. Les conséquences attendues des arbres de ville sur les citadins et sur le climat urbain, mais aussi les effets du lieu d´implantation sur l'arbre sont discutés. Avec cela, les conséquences du changement climatique continuent d´être évaluées, ainsi que des concepts modernes et de futures perspectives de recherche sont présentées et analysées.

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