# Phytodiversity and regulating ecosystem services of urban forests in the Ruhr Metropolitan Region (Northrhine-Westphalia, Germany)

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# List of abbreviations and parameters

| | |
|---|---|
| A.p.a.w. | Plant-available water measured for h = 100 cm in the A-horizon |
| A.coarse.soil | Proportion of coarse soil (> 2 mm) in the A-horizon [%] |
| AL | Zeche Alma |
| $A_{n,m}$ | Number of species gained by the transition from group 1 to 2 |
| ANOVA | Analysis of variance |
| B.coarse.soil | Proportion of coarse soil (> 2 mm) in the B-horizon [%] |
| $B_{n,m}$ | Number of species lost by the transition from group 1 to 2 |
| B.p.a.w. | Plant-available water measured for h = 100 cm in the B-horizon |
| BlueHa | Mixed deciduous urban greening forest at the Blücher Halde (monitoring site) |
| BoHo | Hornbeam urban greening forest in Bochum-Hordel (monitoring site) |
| bw-w-ratio | Brushwood-to-wood-ratio |
| C | Carbon / Competitor (competitive type by GRIME 1979) / Chamaephyte |
| $C_{LB}$ | Annual carbon stock in leaves [kg $a^{-1}$ $m^{-2}$] |
| $C_{mic}$ | Carbon from microbial biomass [µg $g^{-1}$] |
| $C_{Resp}$ | Annual sum of soil respiration [kg $a^{-1}$ $m^{-2}$] |
| $C_{seq}$ | Annual forest carbon sequestration [kg $a^{-1}$ $m^{-2}$] |
| $\Delta C_{WB}$ | Annual change in carbon stock of woody biomass [kg $a^{-1}$ $m^{-2}$] |
| $C_{WBx}$ | Annual carbon stock in woody biomass |
| $\Delta CO_2$ | Annual differences in $CO_2$ assimilation [kg $a^{-1}$ $m^{-2}$] |
| CR | Competitor-ruderal (competitive type by GRIME 1979) |
| CS | Competitor-stress tolerator (competitive type by GRIME 1979) |
| CSR | Intermediary competitive type (GRIME 1979) |
| DBH | Diameter at breast height [cm] |
| DLM | Digital landscape model |
| DOY | Day of the year |

| | |
|---|---|
| E | Evenness |
| EC | Eddy Covariance |
| ES | Ecosystem services |
| F | ELLENBERG's indicator for soil moisture |
| FC | Field capacity |
| G | Geophyte |
| H | Hemicryptophyte |
| $\Delta H$ | Diurnal air humidity deviation [%] |
| H' | Shannon-Index |
| h | Pressure head [cm] |
| he | Heliomorphic leaf anatomy |
| Hert-Bu | Beech forest in Herten (monitoring site) |
| Hert-Ei | Oak-hornbeam forest in Herten (monitoring site) |
| hg | Hygromorphic leaf anatomy |
| $K_s$ | Saturated hydraulic conductivity [cm s-1] |
| L | ELLENBERG's indicator for light |
| LAD | Leaf area density [-] |
| LAI | Leaf area index [-] |
| $LAI_{corr}$ | Leaf area index after correction [-] |
| LMA | Leaf mass area [mg cm$^{-2}$] |
| $LMA_a$ | Mean LMA of the stand [mg cm$^{-2}$] |
| $LMA_{top}$ | LMA of the topmost leaves in the stand [mg cm$^{-2}$] |
| $LMA_{bottom}$ | LMA of the lowest leaves in the stand [mg cm$^{-2}$] |
| LSA | Land surface temperature [°C] |
| m | Number of species in group 2 |
| me | Mesomorphic leaf anatomy |
| mphil | Moderate urbanophilic |

| | |
|---|---|
| mphob | Moderate urbanophobic |
| n | Number of species in group 1 / neutral urbanity |
| N | Nitrogen / ELLENBERG'S indicator for nitrogen content / Nanophanerophyte |
| $N_{mic}$ | Nitrogen from microbial biomass [µg $g^{-1}$] |
| $NO_x$ | Nitrogen oxides |
| PAI | Plant Area Index [-] |
| PCA | Principal component analysis |
| pF | Permanent wilting point |
| pnV | Potential natural vegetation |
| R | ELLENBERG's indicator for soil acidity / Ruderal (competitive type by GRIME 1979) |
| RL BRG | Regional red list of vascular plants for the Ruhr Metropolitan Region |
| RL NRW | Red list of vascular plants for Northrhine-Westphalia |
| RL WB/WT | Regional red list of vascular plants for the Westphalian Lowland and Westphalian Bay |
| RE | Halde Rheinelbe |
| RecPa | Rechener Park |
| ReHa-Ah | Sycamore forest at the Halde Rheinelbe (monitoring site) |
| ReHa-Bi | Birch forest at the Halde Rheinelbe (monitoring site) |
| ReHa-Ei | Mixed deciduous urban greening forest at the Halde Rheinelbe (monitoring site) |
| RePa-Bu | Beech forest at the Rheinelbe Park (monitoring site) |
| RePa-Ah | Sycamore forest at the Rheinelbe Park (monitoring site) |
| RP | Rheinelbe Park |
| P | Phanerophyte |
| PET | Physiological equivalent temperature |
| phil | urbanophilic |
| phob | urbanophobic |
| S | Stress tolerator (competitive type by GRIME 1979) |

| | |
|---|---|
| sk | Skleromorphic leaf anatomy |
| SR | Stress tolerator-ruderal (competitive type by GRIME 1979) |
| SOC | Soil organic carbon |
| su | Succulence leaf anatomy |
| T | ELLENBERG's indicator for temperature / Therophyte |
| ΔT | Mean diurnal air temperature deviation [K] / Mean diurnal cooling [K] |
| Temp | Mean soil temperature [°C] in 5 cm depth taken during respiration measurements |
| TD | Tree density [trees 100 $m^{-2}$] |
| TH | Tree height [m] |
| TN | Terrain height over NN [m] |
| TR | Turnover rate |
| theta | Median of soil volumentric water content during vegetation period (April - September) measured in 5 cm depth |
| UCI | Urban cool island |
| UHI | Urban heat island |
| WePa | Birch forest at the Westpark (monitoring site) |
| ZeHa | Birch forest at the Zeche Hannover (monitoring site) |
| ZZ | Zeche Zollverein |

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# Introduction

## 1 Research objective

With a population of 5.7 million inhabitants (2,949 people per km²), the polycentric Ruhr Metropolitan Region in Northrhine-Westphalia is the largest agglomeration in Germany and one of the biggest agglomerations in Europe (BRINKHOFF 2021). Due to the industrial past and the history of urban development, the region is characterized by a variety of different urban forest types. On one hand, old forest remnants of the former natural and cultural landscape still exist in form of beech, oak-hornbeam, and birch-oak forests (HETZEL 2013), which are nowadays fragmentally embedded in the urban agglomeration. On the other hand, so-called urban-industrial forests (GAUSMANN 2012, KEIL & LOOS 2005), consisting of birch, poplar, or willow species, spontaneously colonized fallow land and brownfields from former coal mining and steel industry since the 1950s, after the industrial sites had been successively closed. Additionally, forests consisting of different deciduous tree species were planted in urban parks or on the iconic slag and slate heaps as remnants of the former coal mining and steel industries.

Those urban forests are important green spaces in urban agglomerations and provide numerous ecosystem services (ES) (CONSTANZA et al. 1997) for urban dwellers (KOWARIK et al. 2017). Large amounts of carbon are stored in the forest soil and in the living biomass. By the process of photosynthesis, forests actively sequester $CO_2$ in their biomass, and therefore, help to mitigate the greenhouse gas effect. The canopy mitigates air pollution and noise and provides a cool shelter during hot days by shading and the process of transpiration. The forest biomass and the soil retain rainwater and lower the surface run-off during rain events. As green spaces, urban dwellers use urban forests for recreation and the forests offer habitats for various plants and animals (BOLUND & HUNHAMMAR 1999, TYRVÄIN et al. 2005).

KOWARIK et al. (2017) postulate, that all these different urban forest types provide ES for urban dwellers in a similar amount. However, studies validating the ES provision for different urban forest types are missing so far. ES are often assessed for urban forests globally and a differentiation between forest types or tree species are not considered (e.g., JIM & CHEN 2009, DOBBS et al. 2014). Therefore, it is not proven if the urban-industrial forests provide ES in the same amount as old forest remnants. Scientists and practitioners currently working on implementing the concept of ES in urban planning and decision-making (GRUNEWALD et al. 2021). When it comes to decision-making, the assessment of urban forest ES must be very precise and the factors which are influencing ES provision in different urban forest types must be known.

So far, it is also unclear, how the phytodiversity of the urban forests influences the ES provision. In scientific and policy literature, it is often postulated that there is a strong correlation between biodiversity and the provision of ES, and that a consideration of green infrastructure in planning leads to an improvement in both, biodiversity and ES, simultaneously (DG Environment 2012). However, empirical evidence of those relationships is often lacking (Schwarz et al. 2017, Knapp et al. 2018, Kabisch et al. 2016, Ziter 2016).

In the Ruhr Metropolitan Region, descriptive studies were published in the past years describing and classifying the species composition of (urban) forests and urban nature in the Ruhr Metropolitan Region (Dettmar 1992, Reidl 1993, Hetzel 2005, Gausmann 2012, Fuchs 2013), but none of them considered the ES provision so far. Vice versa, studies were published assessing the ES provision for urban forests in Essen (Jay et al. 2015) and for street trees in Duisburg (Scholz et al. 2018a). However, these studies assessed the ES globally and did not differentiate between forest types or tree species.

*Table 1: Research questions addressed in the dissertation.*

| **Framework** | |
|---|---|
| • Is the framework suitable for ES assessments?<br>• What are the advantages and disadvantages of this framework compared to those using indicators, literature reviews, and models? | |
| **ES provision** | **Phytodiversity** |
| • How does the ES provision differ between the forest types?<br>• Do the urban-industrial forests provide ES in a similar amount than old-grown forests of the former natural and cultural landscape?<br>• Which factors influence the ES provision in urban forests?<br>• How does the dry and warm conditions of the drought period 2018-2020 influence the ES provision of urban forests? | • How does the forest phytodiversity differ between the forest types?<br>• Which factors influence the phytodiversity in urban forests? |
| **Phytodiversity-ES relationship** | |
| • How does the forest phytodiversity influence the ES provision?<br>• Do forests with high ES have also a high phytodiversity? | |

The present work tackles these different aspects described above and Table 1 gives an overview of the research questions addressed in this dissertation. Selected ES were measured and modelled for the different forest types of the Ruhr Metropolitan Region in the field on monitoring sites and factors influencing the provision were analyzed. The results from the field investigations were than used to estimate the ES provision of a larger sample of forest biotopes. The phytodiversity of those forest biotopes were mapped and the relationship between phytodiversity and ES was analyzed. The investigations of this dissertation took place in the years 2017, 2018, and 2019, whereby the years between 2018 and 2020 were characterized as drought years in the region. Therefore, the results can show how the ES provision of urban forests in the study area will be impacted by warm temperatures and water shortage.

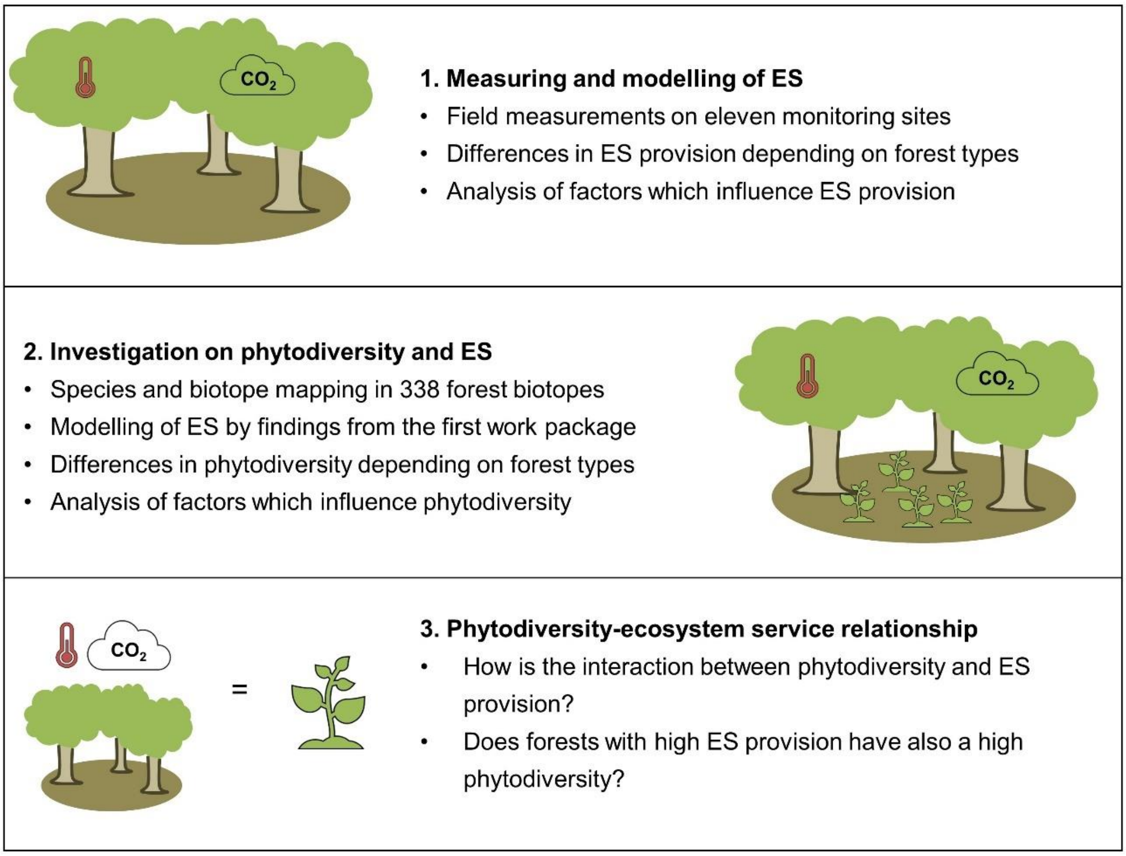
ES provision is typically estimated by a comprehensive literature review, by indicators, or models (e.g., NOWAK et al. 2008, DOBBS et al. 2011, ESCOBEDO et al. 2011, ANDERSSON-SKÖLD et al. 2018), while studies measuring the ES provision precisely in the field are sparsely found (RÖTZER et al. 2021). The present work follows a different approach, where ES provision is measured in the field. Based on the results, this framework is evaluated in terms of suitability for ES assessments and which advantages and disadvantages are imbedded compared to the classical approaches.

The provision of ES quantified in this case study and the factors influencing the ES can be used as a reference for other urban forest studies in the region. Urban forestry, planners, and city administrations can use the findings to calculate ES of urban forests and create measures for the improvement of the ES provision. As the investigations cover a very dry and warm period, the findings of this study show how the provision of ES by urban forests can change under global warming and which factors are crucial to adapt forests to the upcoming climatic changes.

# 2 Research design

## 2.1 Framework and organization of the work

To address the different aspects and research questions, this dissertation is structured in different work packages shown in Figure 1. In Figure 2, the arrangement of the work packages is shown. In the first work package, selected ES were precisely measured and modelled for the years 2018 and 2019 on eleven monitoring sites, representing the different forest types of the region. As ES, carbon storage, $CO_2$ sequestration, and the microclimatic cooling were considered (compare chapter 2.3). The main goal of this work package was to quantify the differences in ES provision between the forest types. Furthermore, the field studies allowed to precisely investigate the factors influencing ES provision in urban forests.



**1. Measuring and modelling of ES**

- Field measurements on eleven monitoring sites
- Differences in ES provision depending on forest types
- Analysis of factors which influence ES provision

**2. Investigation on phytodiversity and ES**

- Species and biotope mapping in 338 forest biotopes
- Modelling of ES by findings from the first work package
- Differences in phytodiversity depending on forest types
- Analysis of factors which influence phytodiversity

**3. Phytodiversity-ecosystem service relationship**

- How is the interaction between phytodiversity and ES provision?
- Does forests with high ES provision have also a high phytodiversity?

*Figure 1: Work packages of the dissertation.*

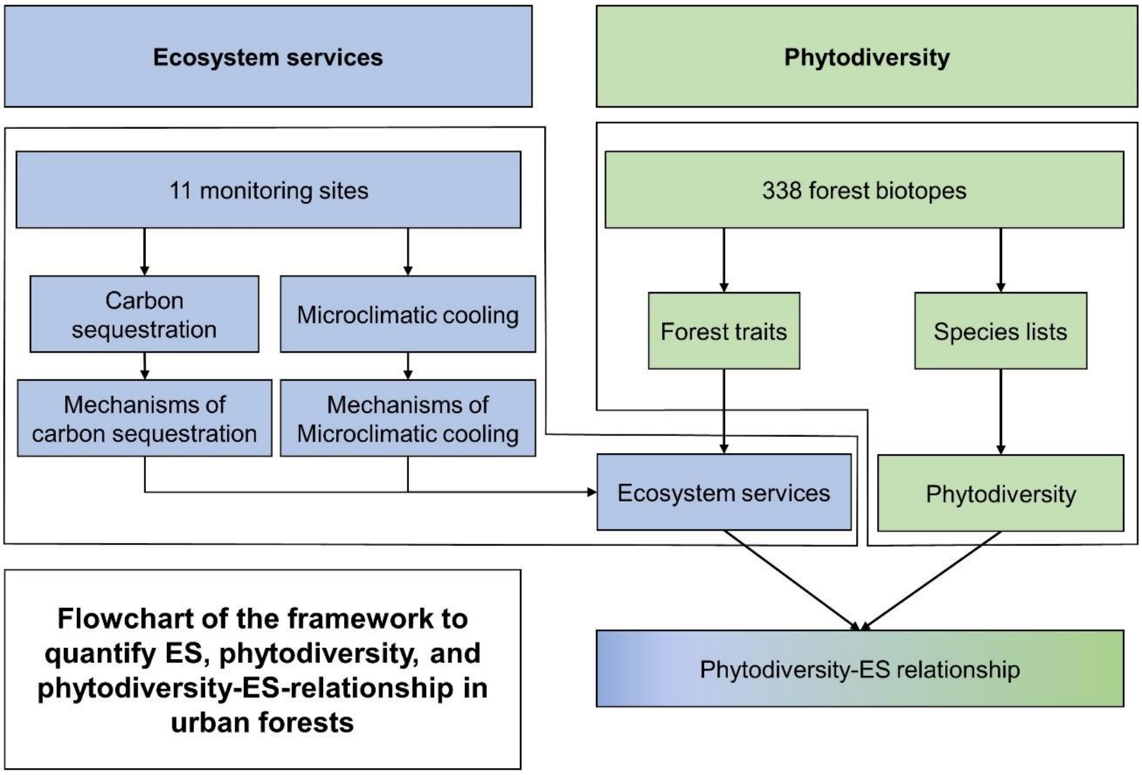


*Figure 2: Flowchart of the framework.*

In the second work package, the findings from the first package were used to model the ES of a sample of 338 forest biotopes depending on the forest traits. Additionally, the phytodiversity was investigated by an extensively species and biotope mapping for each forest biotope to, i) investigate the differences in phytodiversity between the forest types, and ii) to analyze the factors influencing phytodiversity in urban forests. Finally, in the third package, both aspects of previous work packages were brought together to analyze the phytodiversity-ES relationship (SCHWARZ et al. 2017) for the forest biotopes, to analyze if and how the phytodiversity influences the ES provision.

In general, the dissertation is organized in four parts (Table 2). The first part is a regional introduction of the urban forests of the Ruhr Metropolitan Region. Here, urban forests are defined and classified. The spatial distribution of urban forests in the region and the ecological and floristic attributes are described. Part II deals with the carbon sequestration and Part III with the microclimatic cooling of urban forests in the region. The methods and results from the field investigations are presented and discussed. The investigations on phytodiversity and the phytodiversity-ES relationship is shown in Part IV. Finally, a conclusion is presented, where the following aspects are addressed:

- How does the forest types differ in the provision of ES and in phytodiversity?
- How does the drought influence the ES provision?
- How does the phytodiversity influence the ES provision?
- Is the framework of the present work suitable for ES assessments compared to those using models, literature reviews, and indicators?

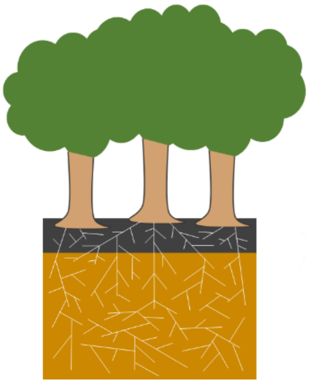
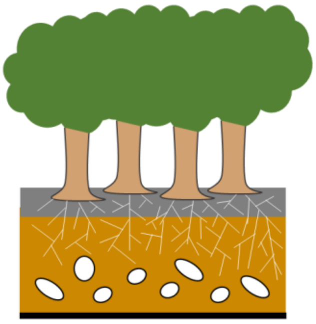
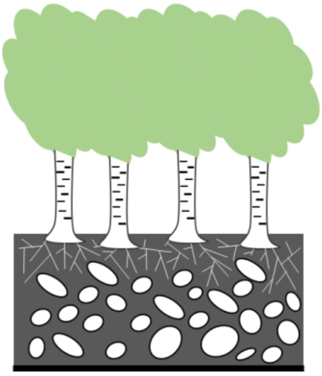
*Table 2: Organization of the dissertation. WP = work package*

| **Parts** | **Title** | **Work package** |
|---|---|---|
| Part I | Urban forests of the Ruhr Metropolitan Region | Regional Introduction |
| Part II | Carbon sequestration of urban forests | WP 1 |
| Part III | Microclimatic cooling of urban forests | |
| Part IV | Phytodiversity and Phytodiversity-ES relationship in urban forests | WP 2 + 3 |

## 2.2 Selected urban forest types

Three different urban forest types were defined: i) Semi-natural forests, ii) allochthonous forests resulting from urban greening (urban greening forests), and iii) autochthonous succession forests (Table 3). The definitions roughly following the concept of urban nature by KOWARIK (2005, 2011), but were modified for the Ruhr Metropolitan Region.

*Table 3: Definitions and attributes of the different forest types investigated in the dissertation.*

| **Attributes** | **Semi-natural forest** | **Urban greening forests** | **Succession forests** |
|---|---|---|---|
| Forest structure and management | Old forest relicts as remnants of the former natural and cultural landscape<br>Beech, maple, and oak forests (> 100 a)<br>Large forest traits (diameter at breast height, tree height, leaf area index)<br>Partly commercial used<br>Maintenance in case of road safety issues | Young allochthonous forests (< 100 a) planted in urban parks and on slag and slate heaps<br>Mixed deciduous forests with a high variety of different native and exotic tree species<br>Forests on slag and slate heaps were planted to stabilize the slopes and to prevent erosion<br>Partly commercial used<br>Maintenance in case of road safety issues | Autochthonous urban-industrial pioneer forests<br>Birch, poplar, willow, or robinia forests spontaneously colonized fallow land and brownfields of the former coal mining and steel industry<br>No commercial usage<br>Maintenance in case of road safety issues |
| Soil conditions | Deep native soils (mostly cambisols, luvisols, or stagnosols)<br>No limitation in rooting<br>Low proportion of coarse soil (> 2 mm diameter)<br>Large soil carbon content | Before planting, a topsoil is generally added for soil melioration<br>Shallow soils (mostly regosols)<br>Rooting is limited due to sealed soil layers<br>Anthropogenic substrates are present<br>Low soil carbon content | Forests growing directly on anthropogenic substrates of the former (industrial) usage (e.g., coal, coke, slag, slate, demolition waste)<br>Very shallow, coarse, and loose technosols<br>Rooting is limited due to sealed soil layers<br>Amount of soil carbon content depend on proportion of coke and coal |
| Sketch |  |  |  |
| Impressions |  |  |  |

Semi-natural forests are old-grown forests (> 100 a) representing the former natural and cultural landscape of the region. They consist mostly of native tree species like *Fagus sylvatica*, *Quercus robur*, or *Acer pseudoplatanus* and they grow on deep native soils (cambisols, luvisols, or stagnosols). Urban greening forests are relatively young (< 100 a), mixed deciduous forests planted in urban parks and on slag and slate heaps from the former coal mining and steel industry, where they stabilize the slopes and prevent erosion of the heap material. They are characterized by a high variety of native and exotic tree species. Before afforestation, a loamy topsoil is generally added for soil melioration, but the rooting is limited due to sealed soil layers. The succession forests are neither planted nor managed and are found on fallow land and brownfields of the former coal mining and steel industries. They consist of pioneer tree species like *Betula pendula, Robinia pseudoacacia,* and different poplar and willow species (*Populus spec.* and *Salix spec*.). The succession forests grow on very shallow, coarse, and loose technosols consisting of different anthropogenic substrates such as stone coal, coke, slate, slag, ash, or demolition waste. More information on the different forest types can be found in Table 3. The deduction of the forest types can be read in Part I – chapter 3.

## 2.3 Selected ES

According to BOLUND & HUNHAMMAR (1999) and TYRVÄIN et al. (2005), following ES are relevant for urban forests:

- Carbon storage and $CO_2$ sequestration
- Microclimatic cooling
- Air pollution filtering
- Noise reduction
- Rainwater retention
- Recreation and cultural values
- Biodiversity and habitat function

In the present work, only carbon storage, $CO_2$ sequestration, microclimatic cooling, and biodiversity of and habitat function for vascular plants (phytodiversity) were considered. It is hypothesized that ES provision might be influenced by the size of the tree traits: Carbon storage and $CO_2$ sequestration of forests might depend on the biomass and the annual growth of trees (NOWAK & Crane 2002, NOWAK et al. 2008) and microclimatic cooling might be controlled by the size and density of the trees canopy (RAHMAN et al. 2020a, GILLNER et al. 2015, REN et al. 2018, NOWAK et al. 2006, HARDIN & JENSEN 2007). The size and density of the canopy might also control air pollution filtering (JANHÄLL 2015), rainwater retention (KERMAVNAR & VILHAR 2017, KEIM et al. 2006), and noise reduction (ANDERSON et al. 1984). Therefore, it can be expected, that the provision of other regulating services will differ in the same manner as carbon storage, $CO_2$ sequestration, and microclimate regulation and those services can act as proxies for the provision of regulating ES in urban forest systems. The cultural and recreational value of urban-industrial forests of the region were already assessed by KEIL (2002) and FINDEL et al. (2003) and will not be addressed in this dissertation.

## 2.4 Field investigations on ES

Carbon storage, $CO_2$ sequestration, and microclimatic cooling were measured and modelled in detail at eleven monitoring sites in forested areas in the cities Bochum, Gelsenkirchen, and Herten (Figure 3), located in the center of the agglomeration.

*Figure 3: Location of the monitoring sites in the Ruhr Metropolitan Region. Basemap and land use is provided by GeoBasis NRW.*

*Table 4: Location, main tree species, size, and approximate age of the monitoring sites.*

| **Forest type** | **Location** | **ID** | **Main tree species** | **Area [m²]** | **Approximate Age** |
|---|---|---|---|---|---|
| Semi-natural forests | Rheinelbe Park | RePa-Bu | *Fagus sylvatica* | 309.4 | > 100 years |
| | | RePa-Ah | *Acer pseudoplatanus* | 293.4 | > 100 years |
| | Hertener Schlosswald | Hert-Bu | *Fagus sylvatica* | 623.4 | > 100 years |
| | | Hert-Ei | *Quercus robur*<br>*Carpinus betulus* | 531.6 | > 100 years |
| Urban greening forests | Halde Rheinelbe | ReHa-Ah | *Acer pseudoplatanus* | 82.0 | 30 - 40 years |
| | | ReHa-Ei | *Quercus robur*<br>*Carpinus betulus*<br>*Acer pseudoplatanus*<br>*Tilia cordata* | 151.2 | 30 - 40 years |
| | Blücher Halde | BlueHa | *Acer pseudoplatanus*<br>*Fagus sylvatica*<br>*Quercus robur*<br>*Carpinus betulus* | 157.3 | 20 - 30 years |
| | Bochum-Hordel | BoHo | *Carpinus betulus* | 93.8 | 20 - 30 years |
| Succession forests | Westpark | WePa | *Betula pendula* | 142.6 | 30 - 40 years |
| | Zeche Hannover | ZeHa | *Betula pendula*<br>*Tilia cordata* | 158.1 | 30 - 40 years |
| | Halde Rheinelbe | ReHa-Bi | *Betula pendula*<br>*Acer pseudoplatanus* | 74.1 | 60 - 70 years |

As semi-natural forests, two beech forests (*Maianthemum-Fagetum, Galio-Fagetum*), one sycamore forest (*Acer pseudoplatanus*), and one oak-hornbeam-forest (*Stellario-Carpinetum*) were chosen at the Rheinelbe Park in Gelsenkirchen-Ückendorf and at the Schlosspark in Herten-Süd-West. Urban greening forests are represented by a sycamore forest (*Acer pseudoplatanus*) and a mixed deciduous forest at the Halde Rheinelbe in Gelsenkirchen-Ückendorf, a hornbeam forest (*Carpinus betulus*) in Bochum-Hordel, and another mixed deciduous forest at the Blücher Halde in Bochum-Günnigfeld. The investigated succession forests were all birch forests (Betula pendula), located at the Halde Rheinelbe (Gelsenkirchen-Ückendorf), at the Zeche Hannover (Bochum-Hordel), and at the Westpark (Bochum-Innenstadt). The size of the monitoring sites varied between 82 m² and 623 m², depending on tree density (Table 4). The technical composition and the methods used to measure and model the ES are explained in Part II and III.

## 2.5 Investigations on phytodiversity

Five study areas were investigated for their phytodiversity (Figure 4): The UNESCO World Heritage Site Zollverein in Essen-Katernberg, the Rheinelbe Park and the Halde Rheinelbe in Gelsenkirchen-Ückendorf, the Alma-Gelände in Gelsenkirchen-Röhlinghausen, and the Rechener Park in Bochum-Wiemelhausen.

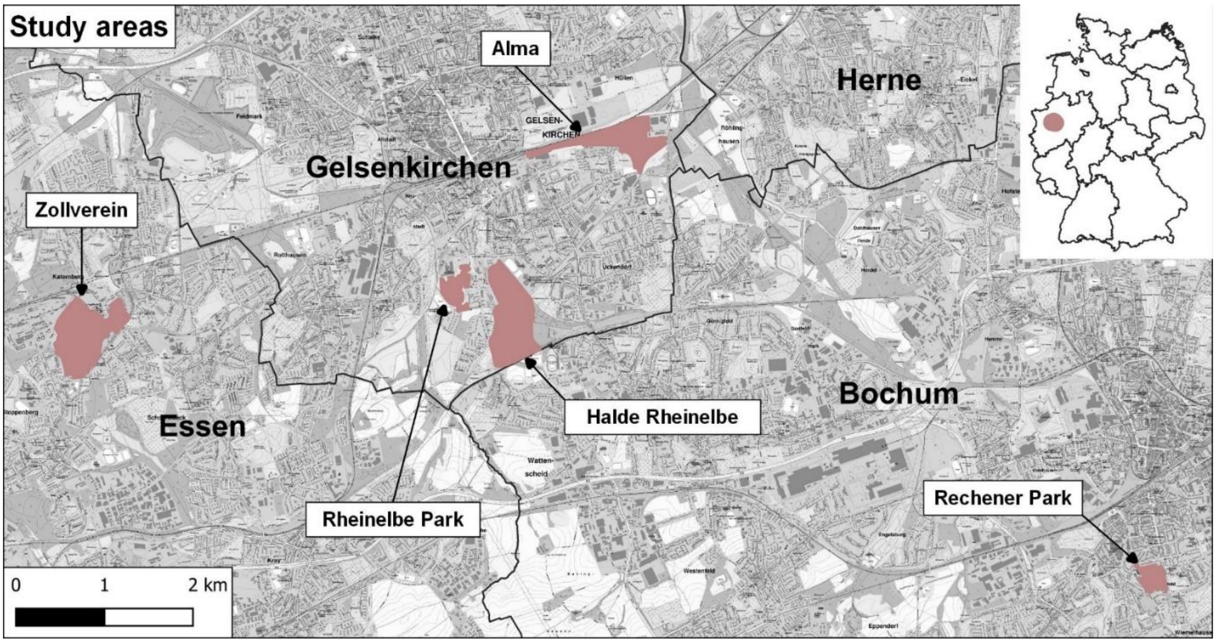


*Figure 4: Location of the study areas for the investigations on phytodiversity. Basemap is provided by GeoBasis NRW.*

Zollverein, Halde Rheinelbe, and Alma are brownfields from former coal mining industries. In the late 20th century, the industrial usage ended, and the former industrial areas were opened for the public as urban parks. Forests spontaneously colonized these sites by natural succession or were planted. The forests are not commercially used, and maintenance only occurs in case of safety issues. All three brownfields are part of the industrial forest project, where the processes of succession (soil development, forest growth, and changes in species composition of plants and insects) are monitored (BUCH et al. 2019, KEIL & SCHOLZ 2016, WEISS et al. 2005). In comparison, the Rechener Park and the Park Rheinelbe are old urban parks (> 100 years old), where forests from the old traditional landscape were conserved. Forest maintenance also occurs only in case of safety issues, but compared to the other sites, the wood is partly commercially used. The methods how phytodiversity were investigated are described in Part IV.

## 2.6 Physical landscape of the study area and climatic conditions during the field surveys

As described in chapter 2.3 and 2.4, the investigations took place in Bochum, Gelsenkirchen, and Herten. The region is characterized by a humid, temperate climate (Cfb, KÖPPEN classification) with an average air temperature of 11.0 °C and an average annual precipitation sum of 865 mm (long-term average 1981-2010 at the Ludger-Mintrop Urban Climate Station in Bochum, GRUDZIELANEK et al. 2011, compare Figure 5).

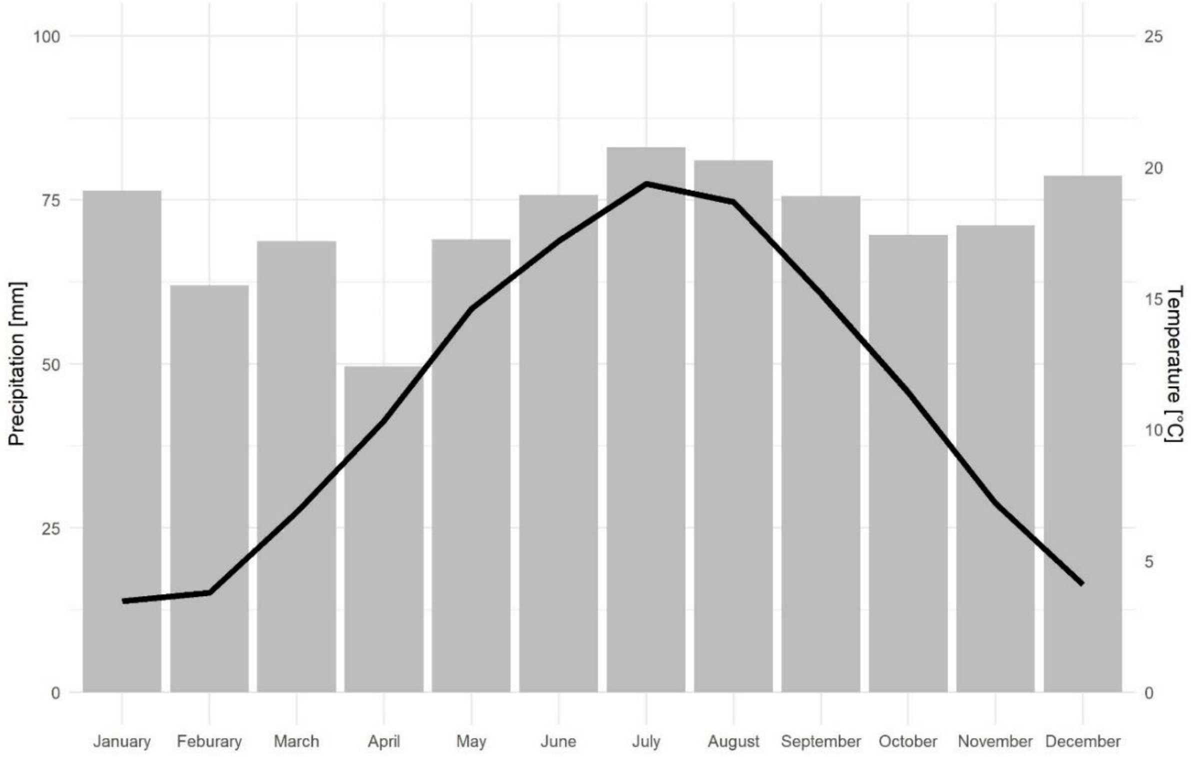


*Figure 5: Climate chart of the urban climate of Bochum created from the data of the Ludger-Mintrop Urban Climate Station for the period 1981-2010.*

The physical landscape of the study area was described by HETZEL (2013). Most natural soils in the region are characterized by loess as substrate for soil development. Therefore, loamy cambisols, luvisols, and stagnosols are most common. However, many soils are influenced by the anthropogenic usage or are completely anthropogenic (technosols). As potential natural vegetation (pnV, TÜXEN 1956), the *Maianthemum-Fagetum* would be expected for most parts of the study area. This plant community is described as beech dominated temperate deciduous forest on loamy soils without stagnating surface water. It is characterized by mesotraphent species like *Milium effusum, Carex sylvatica, Maianthemum bifolium, Oxalis acetosella, Viola reichenbachiana*, or *Poa nemoralis*. However, due to of the urban agglomeration only a small proportion of the current vegetation correspond to the pnV.

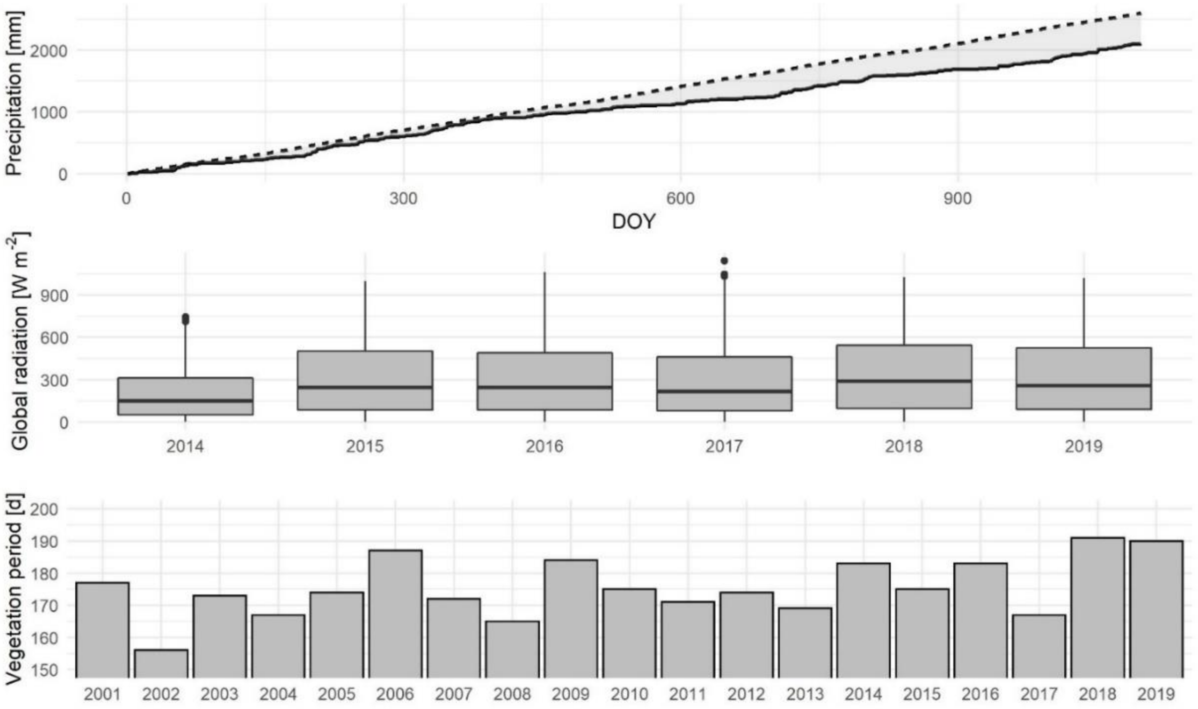


*Figure 6: Cumulative precipitation [mm] from 2017 till 2019 in relation to the climate normal mean (1981-2010) (dotted line) at the Ludger-Mintrop Urban Climate Station in Bochum, the global radiation [W m$^{-2}$] during vegetation period (April to September) measured at the Ruhr-University in Bochum, and the length of the vegetation period for European Beech observed at the monitoring site "Die Haardt" in Recklinghausen (LANUV NRW).*

During the field studies (2017 till 2019), the region experienced a historical three-year lasting drought period (ZSCHEISCHLER & FISCHER 2020). While temperature and precipitation in 2017 was nearly equal to the climate normal mean, the year 2018 and 2019 were much warmer and drier than typical for the region (Figure 6). The average air temperature was 11.6 °C in 2018 and 11.2 °C in 2019 and the precipitation was 544 mm in 2018 and 739 mm in 2019 leading to an annual precipitation deficit of 321 mm and 126 mm, respectively. The mean global radiation during the vegetation period (April-September) was slightly higher than in the years before with 288 W m$^{-2}$ in 2018 and 256 W m$^{-2}$ in 2019. The climatic conditions therefore increased the length of the vegetation period for deciduous trees in the region about 16 days in both years. The two compounded dry and hot summers 2018 and 2019 increased the vulnerability of ecosystems not only in the Ruhr Metropolitan Region but also in whole Europe (BURAS et al. 2020, BASTOS et al. 2021).

# Part I – Urban forests of the Ruhr Metropolitan Region

*This part of the dissertation was already published in:*

*Scholz, T. (2020): Urbane Wälder im Ruhrgebiet – Klassifikation, Merkmale und Regulationsleistungen. Jahrbuch des Bochumer Botanischen Vereins 11: 339-356.*

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# 1 Definition of urban forests

An urban forest is not a large, contiguous, and homogenous forest complex, rather it is an intensively used cultural area with diverse small forest patches embedded in the urban agglomeration with different functions und usages (BDF 2018). A global definition for urban forests does not exist. Rather different terms circulate in scientific literature to describe forest patches in cities and urban agglomerations like “forest in the metropolitan area” (BROGGI 1999), “forest close to cities and settlements” (KVR 1993), “forest close to metropolitan areas” (MARSCHNER 1990) or “city forest”, whereby the latter is a term to describe the forest ownership (DOHLEN 2006). ELLENBERG & LEUSCHNER (2010) did not explicitly consider urban forests for their description of the vegetation of Central Europe, because they defined natural or semi-natural tree population as woodlands while tree populations as products for timber were defined as forests. The term “urban forest” does not necessarily fit in this definition, because, on the one hand, urban forests are often planted, managed, and intensively used by the urban inhabitants which makes the forest not natural, but on the other hand, the forests are often not used for timber production. The Bundeswaldgesetz defines forests more globally, where every area with forest plants is defined as forest, independently from the naturalness, usage, or forest structure.

Also, the minimum area, tree height, and canopy closure of urban forests are not uniformly determined. Basically, the lower limit is to define, where, due to the small extent, a forest climate is not able to build, and specific habitat functions are not fulfilled anymore (BREUSTE 2019). LANUV NRW (2018a) determined all tree populations larger than one hectare and more than 50 m width as forests. The one-hectare limit agrees with BURKHARDT et al. (2008) and THOMASIS & SCHMIDT (1996), whereby one hectare is needed for ecosystem stability and a typical forest microclimate. However, KREFT (1993) mentioned, that it is the rule rather than the exception, that forests in agglomerations are smaller than one hectare due to fragmentation. DOHLEN (1996) considered this and counts all tree populations with a minimum area of 2.500 m² and a minimum width of 10 m to urban forests. Furthermore, urban forests must be strongly influenced by the urban agglomeration, the canopy must be closed more or less, 50 % of the area must be occupied by a herb, moss, or shrub layer, and an initial humus layer must be present. The current and former usage leads to measurable changes in forest structure and functions. LESER (2008), DIETRICH (2013), and LIU et al. (2003) defined the minimum area to 5.000 m², BURKHARDT et al. (2008) to 3.000 m² (BREUSTE 2019).

RANDRUP et al. (2005) introduced the term “urban woods and woodland” and agreed to the one-hectare criterium, supplemented by a minimum tree height of 5 m and a canopy closure of more than 30 %. ROWNTREE (1984) also determined the term “urban forest”, but in comparison to the term used in this dissertation, all urban tree populations are included, even trees at roadsides, in parks, gardens, and cemeteries. The reason for the variety of definition for urban forests might be that it strongly depends on the understanding different stakeholders and users have on urban forests (LUND 2002a, 2002b). Urban inhabitants who seek recreation in urban forests would name even small tree populations as forests, while in the view of forest economy and ecology, forests need to have a minimum size to fulfill specific functions. It also depends on the provenance: people who live in the center of urban agglomerations would name even small tree population as forests, in comparison to people from forest-rich mountain ranges. Furthermore, the understanding of forests is continuously changing: In the past, forests were primarily used for timber production. Nowadays, the regulating services and the social and recreational value is more important. Therefore, the current forest management can be described as multifunctional, where the interests of the different stakeholders (foresters, forest owners, residents, urban dwellers who are seeking recreation, and scientists) must be matched (HELMS 2002, MULNV NRW 2018).

For the presented work, the definition of “urban forest” introduced by ROWNTREE (1984) is not suitable, because it is too extensive. Therefore, an own interpretation of “urban forest” is used for the dissertation, which sums up all the definitions and aspects shown above:

> *Urban forests are tree populations of variable sizes in terms of area, height, and canopy closure, inside, at the fringe or in the effective range of urban agglomerations, whose site conditions are directly influenced by the vicinity of humans. Due to the forest microclimate and the specific habitat conditions, urban forests provide numerous ecological, regulative, and social-cultural functions.*

## 2 Forest characteristics of the Ruhr Metropolitan Region

After evaluation of the Digital Landscape Model of Northrine-Westphalia (DLM, GEOBASIS NRW 2017), the forested area can be quantified to 1,036 km² which is 23 % of the regions area. Around a quarter of the whole Ruhr Metropolitan Region is therefore occupied by forests. This number is in good agreement with the official numbers of the Ruhr Metropolitan Region forest district which is 22 % (LANDESBETRIEB WALD & HOLZ 2014). The proportion of forests in the region is, therefore, below average compared to the proportion of forests in Northrhine-Westphalia (27 %) (LANDESBETRIEB WALD & HOLZ 2014). 820 km² are large urban forests and those can be found largely at the edge of the central agglomeration in Wesel (Üfter Mark, Dämmerwald, Lichtenhagen), Recklinghausen (Die Haard, Borkenberge, Weißes Venn-Geisheide), Ennepe-Ruhr-Kreis, Hagen, in the north of Oberhausen and Bottrop (Köllnischer Wald, Hiesfelder Wald, Kirchheller Heide), as well as between the cities Mülheim and Duisburg (Mülheim-Duisburger Wald). Looking at the distribution of forests in the region (Figure 7), the question arises to what extent these large contiguous forest areas can still be described as urban forests. It is undisputed that the forests are used and influenced by people seeking for recreation. However, it is questionable, whether the site conditions are largely and comprehensively shaped by the vicinity of human settlements. In the core cities of the agglomeration (Duisburg, Mülheim, Oberhausen, Bottrop, Essen, Gelsenkirchen, Herne, Bochum, Dortmund), the forested areas are smaller and more fragmented.

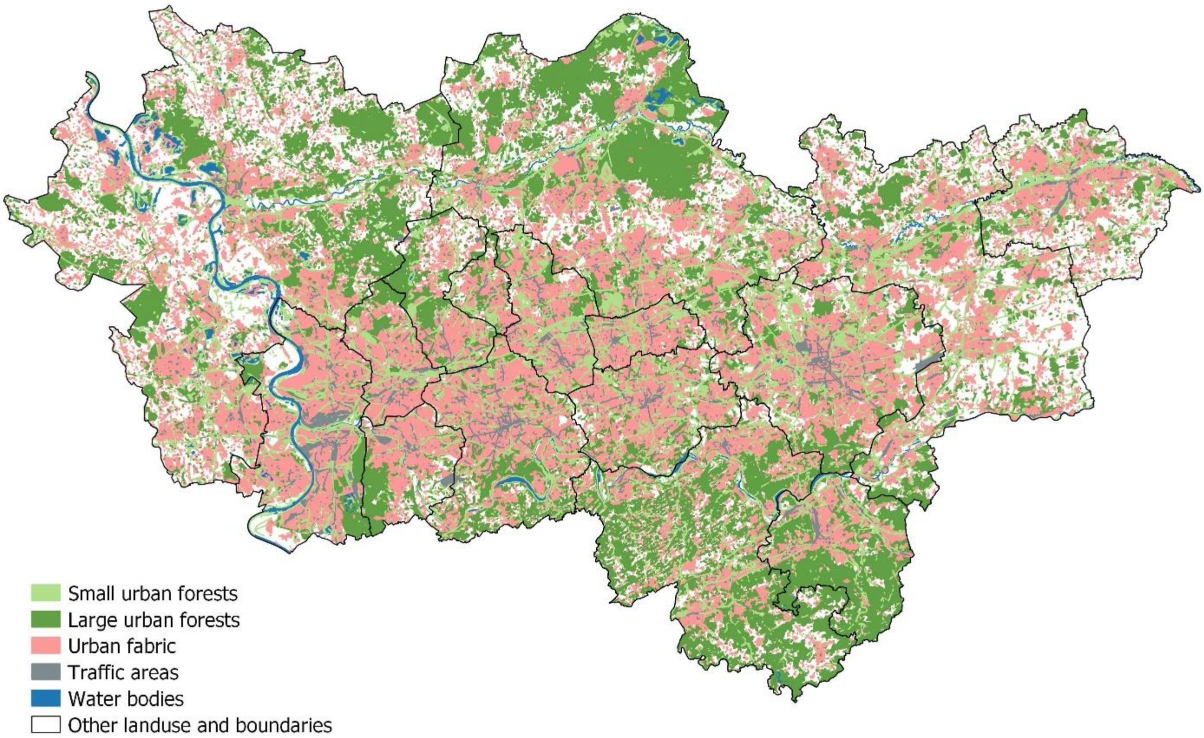


*Figure 7: Distribution of urban forests in the Ruhr Metropolitan Region. Data source: GeoBasis NRW 2017.*

The tree species composition is more balanced in the Ruhr Metropolitan Region compared to other forest districts in Northrhine-Westphalia. Oak species (*Quercus spec.*) are with around 25 % most common in the region, followed by *Fagus sylvatica* and other deciduous and coniferous species. In the Ruhr Metropolitan Region, deciduous tree species predominate with over 50 %, which is above the average of Northrhine-Westphalia. Together with the Rhein-Sieg-Kreis forest district, the Ruhr Metropolitan Region is the only region in Northrhine-Westphalia in which more deciduous than coniferous tree species grow (Landesbetrieb Wald & Holz 2014), which is mainly due to the fact that large Norway spruce forests (*Picea abies*), such as those in the southern mountains, are largely absent.

The balance of the tree species composition is related to the fact that very different forest regions come together in the Ruhr Metropolitan Region, which result on the one hand from the meeting of the atlantic and continental biogeographical regions, on the other hand from intra-regional differences in climatic and edaphic factors. Furthermore, due to the location in the largest metropolitan area in Europe, the human influence in the forest stands is significantly higher than in other forest districts, which has contributed to the formation of completely new forest types (urban greening and urban-industrial forests). In the north, parts of the (Sand)-Münsterland belong to the region, which is characterized by large contiguous pine, birch, oak, and beech forests on sandy soils such as Die Haard or the Hohe Mark. In the center between Duisburg and Dortmund and further east in transition to the Hellwegbörde near Unna, loess-related loamy soils occur with older deciduous forest stands. Additionally in the agglomerations, pioneer forests spontaneously colonized all kinds of fallow land, and in parks and on heaps forests were planted. Finally in the south, parts of the Süderbergland belong to the region, which is mainly characterized by beech and Norway spruce forests (Landesbetrieb Wald und Holz NRW 2012).

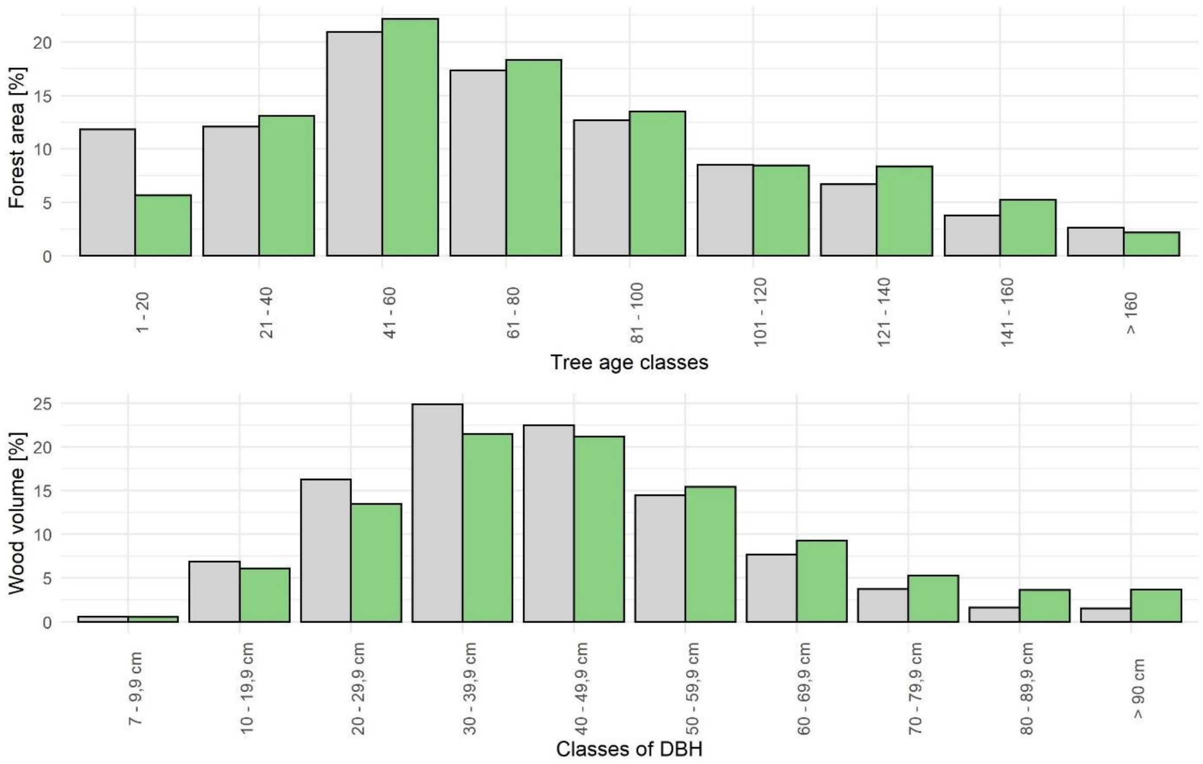


*Figure 8: Forest age and distribution of wood volume on diameter at breast height (DBH) classes for the forests of the Ruhr Metropolitan Region (green) compared to Northrhine-Westphalia (grey). Source: Landesbetrieb Wald und Holz NRW 2014.*

Most forests of the region are relatively young. Around one quarter of all forests are between 41 and 60 years old. 73 % are younger than 100 years and only 2 % are older than 160 years. The current average wood volume is about 319.7 m³ $ha^{-1}$. From that, 43 % account to the DBH (diameter at breast height) class 30 - 50 cm. Compared to all forests in Northrhine-Westphalia, more wood volume can be found in larger DBH classes.

# 3 Classification of urban forests

Urban forests can be classified by a spatial, functional, and historical perspective (Table 5).

*Table 5: Classification of urban forests by a spatial, functional, and historical perspective. Compiled from* KOWARIK *2005,* BURKHARDT *et al. 2008, and* BREUSTE *et al. 2016.*

| **Forest type (spatial)** | **Description** | **Characteristics** |
|---|---|---|
| Urban woodland | Forests surrounded by urban areas or at the urban fringe | Isolated in urban areas or between urban areas and the cultural landscape; strong urban influence and importance of social functions; forestry subordinate |
| Peri-urban woodland | Forests in the vicinity of urban areas | Element of the cultural landscape, but still influenced by the urban agglomeration |
| Non-urban woodland | Woodlands far from urban areas | Element of a semi-natural landscape largely without urban influence; importance for forestry, social functions subordinate |
| **Forest type (functional)** | **Description** | **Characteristics** |
| Neighborhood forest | Small forests in residential areas | Positive effects for local microclimate; accessible forest structure; high importance for people with restricted mobility (children, elderly people); insufficient maintenance and garbage dumping |
| District forest | Medium sized forests, often between city districts | Usage by residents, pedestrians, and cyclists; maintenance depends on usage |
| Recreation forest | Large forest areas (> 60 ha), mostly at the urban fringe | High diversity and closeness to nature; different opportunities for nature experience; infrastructure for recreation such as a path network, benches, and information signs |
| Production forest | Forests for timber production outside urban areas | Primarily timber production; as required, nature conservation and recreation are possible |
| **Forest type (historical)** | **Description** | **Characteristics** |
| Semi-natural forests | Remnants of old (semi-)natural forests | Old forests (> 100 years); species composition close the potential natural vegetation (pnV) |
| Silvicultural forests | Forests primarily for timber production | Forest structure primarily influenced by forestry; monocultures for timber production |
| Urban greening forests | Forests planted in urban parks and afforestation on slag or slate heaps | Afforestation under technical or aesthetic aspects; high diversity of tree species |
| Succession forests | Forests spontaneously colonized fallow lands on urban-industrial sites | Pioneer to intermediate forest communities spontaneously colonized brownfields and other fallow land; heterogenous forest structure; high diversity of plant species |

From the spatial perspective, KOWARIK (2005) differentiate between “urban woodlands” as isolated forests surrounded by urban areas or on the urban fringe, “peri-urban woodlands” as forests in the vicinity of urban areas which are deeply imbedded in the peri-urban cultural landscape, and “non-urban woodlands” as forests which lie far outside the urban impact area (KOWARIK 2005: 4). The closer the forests are to the human settlements, the greater is the impact of the urban influence on the forest. At the same time, the social and regulatory importance of the forest for humans is increasing, while the forestry usage is decreasing.

BURKHARDT et al. (2008) differentiate urban forests by their function for urban inhabitants. “Neighborhood forests” are small forests in residential areas, which have an importance for the local microclimate and are intensively used by the residents. Medium sized forests between city districts are called “district forests”. Those forests are crossed by residents, pedestrians, and cyclists. “Recreation forests” are large contiguous forests with infrastructure such as a path network, benches, or information signs. Forests primarily for timber production outside of cities are named “production forests”.

A classification by an historical perspective is given by KOWARIK (2005):

- Semi-natural forests: old forest remnants of the former natural and cultural landscape
- Silvicultural forests: forests shaped by forestry measures
- Urban greening forests: Afforestation in urban parks or on slag and slate heaps
- Succession forests: Forest succession on urban-industrial sites

Due to the long history of settlement and the urban and industrial development of the Ruhr Metropolitan Area, this scheme is most suitable to classify the forests found in the region. The names of the forest types are slightly changed to those introduced by KOWARIK (2005). In the following, the forests of the Ruhr Metropolitan Region are classified by this scheme (overview in Table 6).

*Table 6: Assignment of typical forest stands in the Ruhr Metropolitan Region to historical forest types according to KOWARIK (2005).*

| **Forest type (historical)** | **Forest stands in the Ruhr Metropolitan Region** |
|---|---|
| Semi-natural forests | Forests of the potential natural vegetation: *Galio-Fagetum, Hordelymo-Fagetum, Periclymeno-Fagetum, Maianthemo-Fagetum, Stellario-Carpinetum, Betulo Quercetum, Pruno-Fraxinetum, Betuletum pubescentis, Carici-Alnetum, Carici-Fraxinetum* |
| Silvicultural forests | Monocultures of Norway spruce, pine, larch, or other coniferous forests, forests consisting of *Quercus rubra*, short rotation plantations |
| Urban greening forests | Mixed deciduous forests resulting from afforestation in urban parks and on heaps with a high diversity of native and exotic tree species |
| Succession forests | Autochthonous birch, poplar, willow, or neophytic forests, especially on fallow land and brownfields of the former coal mining and steel industry |

**Semi-natural** forests are characterized by their high age and that the species composition is relatively close to the potential natural vegetation (pnV, TÜXEN 1956). Main forest functions are often nature conservation and recreation, while timber production is not the primarily usage. Depending on location and site, the plant communities can be described as *Luzulo-Fagetum* (south of the region on acidic rocks), *Galio-Fagetum* and *Hordelymo-Fagetum* (south of the region on calcareous rocks), *Periclymeno-Fagetum, Maianthemo-Fagetum*, and *Stellario-Carpinetum* (Figure 9) (center of the region on loess-related loamy soils), and *Betulo-Quercetum* (north of the region on sandy soils). On moist sites, the *Pruno-Fraxinetum, Betuletum pubescentis, Carici-Alnetum*, and *Carici-Fraxinetum* are expected for the pnV (HETZEL 2013). Large contiguous semi-natural forests can mostly be found at the fringes of the region. However, some small-sized and fragmented semi-natural forests were conserved even in the center of the agglomeration in form of urban parks. The Rheinelbe Park in Gelsenkirchen-Ückendorf is a former villa park, where an old European Beech stand can be found, which is approximately 150 till 200 years old (Figure 10). The plant community can be described as a *Galio-Fagetum* on loess-related loamy soils. In spring, many vernal geophytes occur such as *Anemone nemorosa, Allium ursinum, Polygonatum multiflorum*, and even *Corydalis solida*. Even though the species composition can be described as *Galio-Fagetum*, it is likely that this forest stand was a former oak-hornbeam forest of the type *Stellario-Carpinetum*, which was drained and afforested with *Fagus sylvatica* (HETZEL 2013). Some drainage ditches at the forest fringe indicate the thesis. Semi-natural forest communities are rare in the center of the agglomeration and endangered due to the high number of visitors and the isolation.

*Figure 9: Oak-hornbeam forest (Stellario-Carpinetum) in the Resser Mark in Gelsenkirchen.*

*Figure 10: Beech forest (Galio-Fagetum) at the Rheinelbe Park in Gelsenkirchen-Ückendorf with blooming of Anemone nemorosa*

The forest structure and species composition in **silvicultural forests** are strongly influenced by historical and modern silvicultural measures and maintenance. In Northrhine-Westphalia this definition is most likely to apply to non-native spruce, pine, or larch monocultures, forests consisting of red oak (*Quercus rubra*), and to short rotation plantations. Urban forests that are primarily used for wood production are rare in the Ruhr Metropolitan Region. The urban forests are often too small and not worthwhile for exclusive timber production. Furthermore, a purely forestry use would conflict with the recreational and climate protection function. Residents often have a strong emotional bond with neighboring forests and clearly perceive structural changes in forests. Therefore, the forests in the Ruhr Metropolitan Region are managed multifunctional (MULNV NRW 2018), so that the type of silvicultural forests does not usually apply to the forests of the region. On the other hand, efforts are being made to develop short rotation plantations, so-called biomass parks (ZEPP et al. 2012), on former industrial areas, which would correspond to the character of silvicultural forests (Figure 11), even though in forestry short rotation plantations are not counted as forests.

Large stand of trees planted in urban parks can be described as **urban greening forests**. The purpose of afforestation is purely functional based on technical or aesthetic aspects. Therefore, afforestation on heaps can also be counted in this category because the primary purpose for afforestation is to stabilize the heap material to prevent erosion (Figure 12). Before afforestation, soil melioration is often carried out. Urban greening forest can be described as mixed deciduous forests which have a high diversity of different native and exotic tree and shrub species (RINGENBERG 1994).

*Figure 11: Young plants of alder and willow from a short rotation plantation in the biomass park on the site of the former Hugo colliery in Gelsenkirchen-Buer.*

*Figure 12: Urban greening forest to stablize the Blücher Halde in Bochum-Günnigfeld.*

**Succession forests** differ from the other types in that they have emerged from natural succession processes, are still characterized by an undisturbed vegetation development, and are not changed by any silvicultural measures, except of road safety maintenance. In the Ruhr Metropolitan Region, this type of forest occurs primarily in the form of so-called urban-industrial forests, which were able to colonize spontaneously on fallow land or brownfields of the former coal mining and steel industries after they were successively closed since the 1950s (DETTMAR 1992, WEISS et al. 2005, GAUSMANN 2012). The forests consist of pioneer tree species such as *Betula pendula* (Figure 13), willow (*Salix* spec.) (Figure 14), or poplar (*Populus* spec.), and exotic tree species such as *Robinia pseudoacacia* or *Ailanthus altissima* (KEIL & LOOS 2005, WITTIG 2008).

The soils of the succession forests are testimony to the history of industry and settlement and consist almost entirely of anthropogenic substrates such as slate, slag, ash, mortar, bricks, or concrete. Depending on substrate composition, the growth conditions for forests can differ greatly (REBELE & DETTMAR 1996, HILLER & MEUSER 1998, KASIELKE & BUCH 2012, SCHOLZ et al. 2018b). An ecogram of the spontaneous trees in Berlin were presented by SUKOPP (1990). Most of the succession forests already have been matured and are currently in the decay phase. To this day it is not clear how the forests will change after the pioneer phase (Gausmann 2012). It is noticeable that only a few seedlings of the pioneer tree species can be found in the herbaceous layer, but the herb layer is characterized by seedlings of intermediate tree species such as *Acer pseudoplatanus* or *Tilia cordata*. Also, thermophilic species like *Juglans regia* are common in the understory vegetation of urban-industrial forests (HETZEL 2012). This could be a sign that the pioneer forests will continuously be replaced by forests consisting of intermediate tree species in future. The succession on the post-industrial areas is being investigated by an interdisciplinary working group as part of the industrial forest project (WEISS 2003, WEISS et al. 2005, KEIL & SCHOLZ 2016, BUCH et al. 2019).

*Figure 13: Birch pioneer forest on the site of the former Hannover colliery in Bochum-Hordel.*

*Figure 14: Willow forest (Salix viminalis) on the Halde Rheinelbe.*

## 4 Ecological and floristic attributes of urban forests

The urban forest types differ in their ecological characteristics (Figure 15). Semi-natural and silvicultural forests have largely homogenous site conditions in comparison to succession forests. Due to their undisturbed succession, the age and forest structure is very heterogenous. Additionally, the growth conditions differ very strong in small distances, depending on the composition of anthropogenic substrates in the soil, leading to small-scaled changes in forest communities (SCHOLZ et al. 2018b). This supports a variety of different small-scaled habitats, leading to high plant species richness in succession forests. In urban greening forests high species numbers can be found depending on the degree of maintenance and on the number of different planted tree species. In semi-natural forests, the species richness depends on the plant community. Especially plant communities on limestone such as *Hordelymo-Fagetum* or *Galio-Fagetum* or forest communities on wet or moist sites can have high plant species richness compared to those on acidic substrates such as *Luzulo-Fagetum* (MEYER & SCHMIDT 2008).

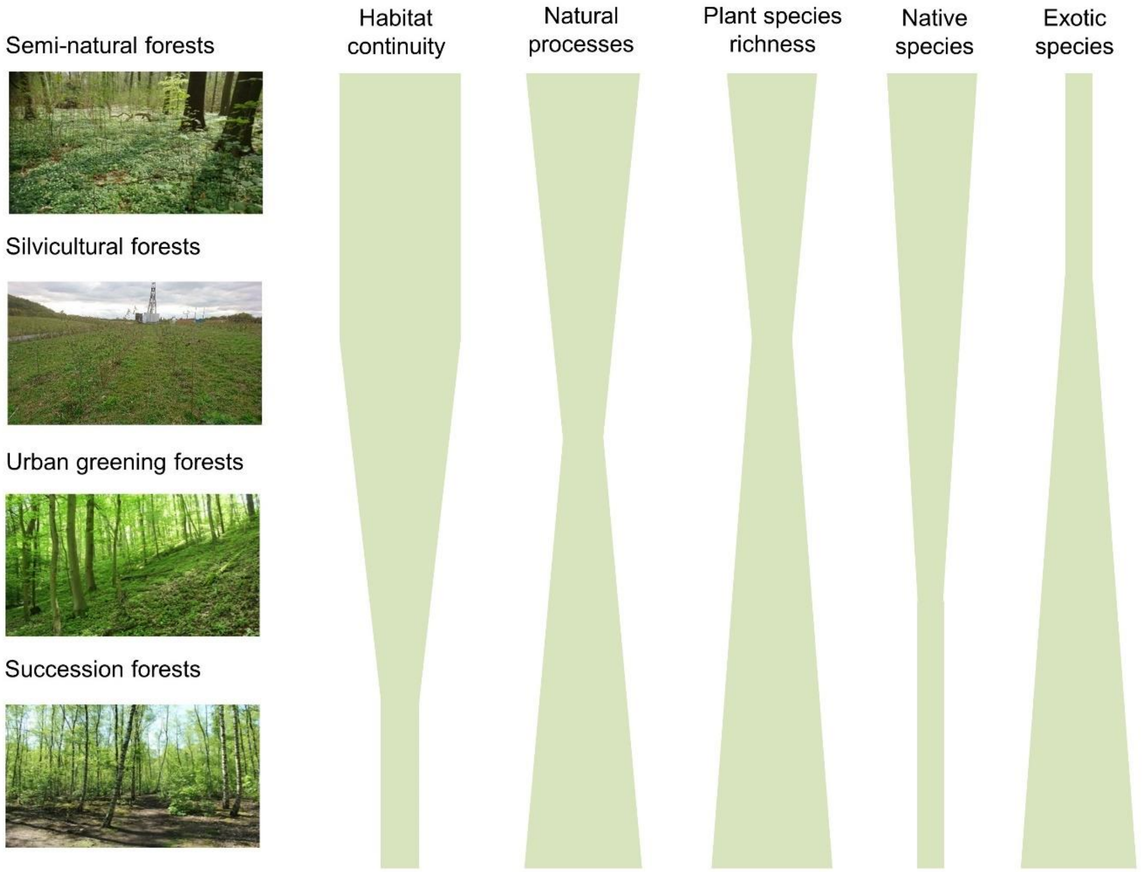


*Figure 15: Ecological characteristics of urban forest types (modified after Kowarik 2005).*

The species composition in succession and urban greening forests are more characterized by exotic species, compared to the species composition in semi-natural and silvicultural forests. Natural processes occur most in semi-natural and succession forests. In silvicultural forests cultivation and maintenance predominate. The degree of cultivation in urban greening forests differs depending on the location. In urban parks the maintenance of the forest stands is more intense, while on heaps the forest succession can be as undisturbed as in the succession forests.

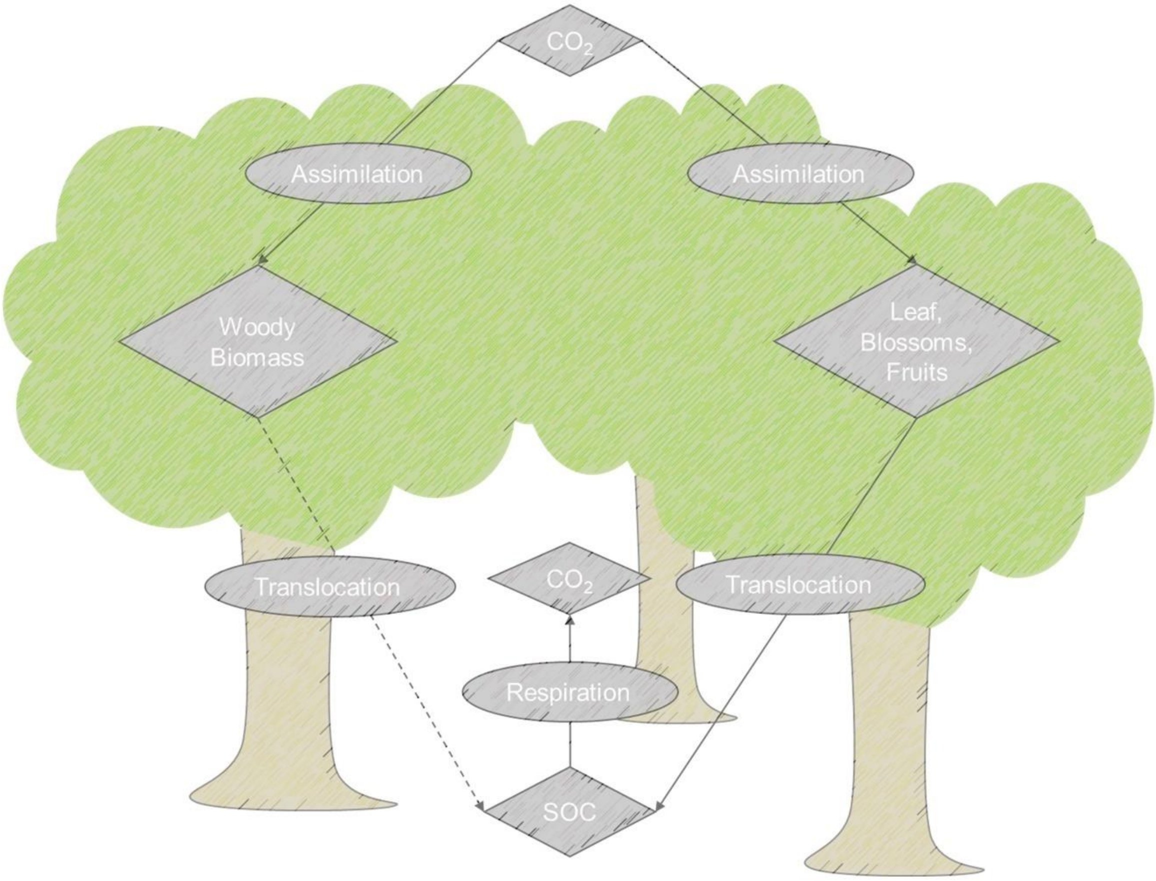
In urban forests, ecological conditions and species composition is significantly influenced by the human presence (Table 7). Mining activities in the past had led to subsidence, so that especially in the Emscher Region many forests are located below groundwater levels. On the other hand, forest stands were cultivated on heaps, which have no access to groundwater due to the sealing of the heap. Soil properties are influenced by admixtures of anthropogenic substrates or, in case of urban-industrial forests, depend on the composition of the substrates. Because forests in urban areas are intensively used for recreation, a high load of visitors influence the ecology and species composition. Beaten tracks through the forest lead to soil compaction and a reduced or missing humus layer. At forest paths, the canopy is more open, influencing light conditions and soil moisture. Eutrophication take place in nearly all urban forests due to air deposition of $NO_x$, disposal of garden waste, and by dog urine and feces. Due to the vicinity of settlement structures, forests are influenced by the urban heat island (UHI), leading to lesser frosts in winter. Isolation and disturbance in urban forests leading to a low density of hoofed game. The intensity of human influence strongly depends on the vicinity to urban areas and the accessibility of forests. In general, the anthropogenic influence is higher in urban forests than in peri-urban forests.

*Table 7: Influence of the urban vicinity on ecological and floristic characteristics of urban forests summarized from Kreft 1993, Dohlen 2006, and Hetzel 2012.*

| **Ecological characteristics** |
|---|
| • Changes in groundwater through mining, excavation, and backfilling<br>• Soil characteristics partly influenced by anthropogenic substrates<br>• Changed soil conditions due to compaction<br>• Reduced or missing humus layer<br>• Soil compaction due to the high number of visitors and a dense network of beaten tracks<br>• Changed light, water, and nutrient supply near the forest paths<br>• Lesser frosts in winter due to urban heat island effect<br>• Eutrophication through air deposition of $NO_x$, disposals of garden waste, and dog urine and feces<br>• Deposition of air pollutants such as $NO_x$, $SO_2$, $PM_{10}$, or $PM_{2.5}$<br>• Low tree damage due to low density of hoofed game |
| **Floristic characteristics** |
| Increased number of<br>• Exotic plant species<br>• Nitrophilic species<br>• Step-resistant plants<br>• Epizoochoric species<br>• Thermophilic species and evergreen plants, who profit from the urban heat island effect<br>• Light-adapted fringe communities |

The ecological characteristics influence the plant species composition, whereby nitrophilic species such as *Urtica dioica, Alliaria petiolate, Sambucus nigra* and various blackberry species (*Rubus spec.*), as well as step-resistant species such as *Poa annua* or *Plantago major* benefit. Epizoochoric species such as *Geum urbanum* or *Circaea lutetiana* benefit that the seeds get caught on clothes of visitors or in the fur of dogs and are spread over the forest. Garden disposals add exotic plant species such as *Rhododendron spec., Fallopia japonica, Mahonia aquifolium, Vinca minor, Aucuba japonica, Prunus laurocerasus, Viburnum rhytidophyllum*, and others to the forest species composition. Many of these horticulturists are evergreen species, which benefit from less frost events in winter because of the UHI effect.

# Part II – Carbon sequestration of urban forests



# 1 Theoretical background

The carbon (C) cycle in forests can be analyzed by changes in C stocks and by measuring the corresponding C fluxes (Figure 16). In forest systems, the largest carbon pools are found in the tree biomass and in the soil, while C in the understory vegetation only makes up a small proportion (PAN et al. 2018). Trees and other plants in the forest ecosystem take up $CO_2$ by the process of photosynthesis, assimilate it either in leaves and fruits or in the woody biomass (stem, branches, roots) while the tree grows. A small proportion is also exudated via the root system directly into the soil. While the storage in the woody biomass is persistent over years, the carbon in leaves and fruits is translocated to the soil after the vegetation period in temperate deciduous forests via litterfall. The plant debris and root exudates are decomposed and mineralized by heterotrophic microorganisms, who need the carbon for energy and biomass production. During decomposition and mineralization, carbon is transferred back to the atmosphere as $CO_2$. In combination with root (autotrophic) respiration, the process is described as soil respiration. Depending on the ratio between carbon assimilation by plants (mainly trees) and soil respiration, urban forest ecosystems can be either a carbon source or a sink (HADDEN & GRELLE 2016).

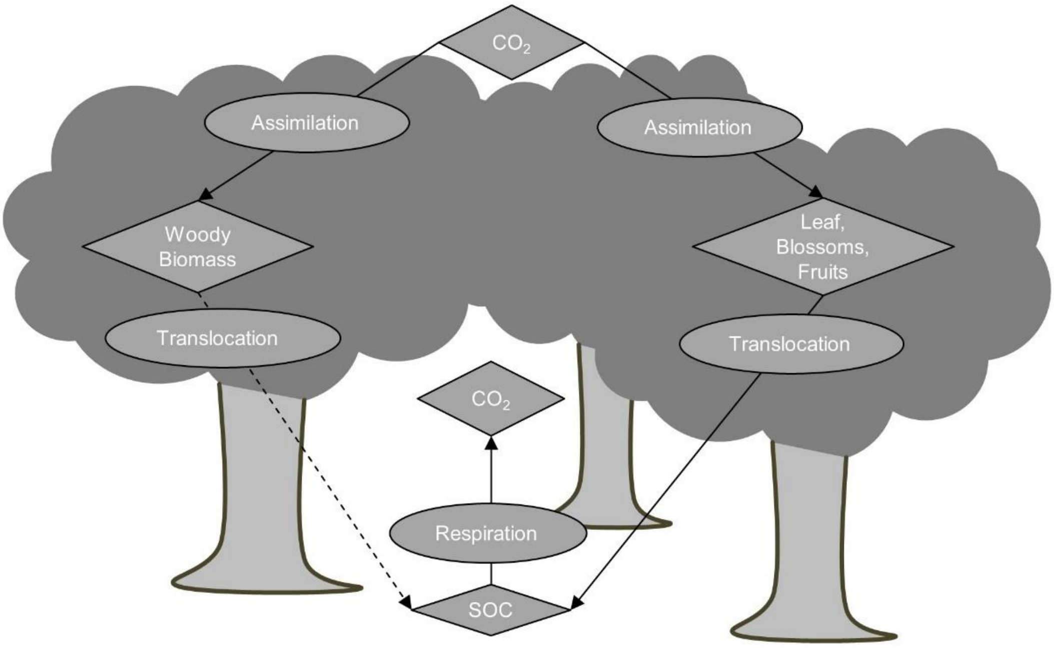


*Figure 16: Schematic carbon cycle in a temperate deciduous forest ecosystem. The dotted line indicates that the translocation process of woody biomass to the forest floor is a discontinuous process in comparison to the translocation of leaves, blossoms, and fruits, which occurs annually. SOC = Soil Organic Carbon.*

Flux measurements such as Eddy Covariance (EC) can provide information about the $CO_2$ exchange between the atmosphere, the vegetation, and the soil, whereby the $CO_2$ assimilation by the vegetation dominates over relatively small $CO_2$ efflux from autotrophic and heterotrophic respiration (GRANIER et al. 2001). Therefore, source portioning of the fluxes is not directly feasible from EC data. To overcome this problem, soil respiration is often additionally measured (e.g., HERBST et al. 2021). Furthermore, the EC technique is cost and maintenance expensive and measurements over a forest stand require the installation of large towers, which cannot be setup at all locations, especially if the sites are located in urban areas, which are open to the public. As an alternative, the C stocks can be measured on the single plant compartments directly and in combination with information about the soil respiration, the ecosystem C sequestration can be estimated by

$$C_{seq} = \Delta C_{WB} + C_{LB} - C_{Resp} \quad [1]$$

$$\Delta C_{WB} = \Delta C_{WBx+1} - C_{WBx} \quad [2]$$

where $C_{Seq}$ is the annual forest carbon sequestration [kg $a^{-1}$ $m^{-2}$], $\Delta C_{WB}$ is the annual change in carbon stock of woody biomass [kg $a^{-1}$ $m^{-2}$], $C_{LB}$ is the annual carbon stock in leaves [kg $a^{-1}$ $m^{-2}$], and $C_{Resp}$ is the annual sum of soil respiration [kg $a^{-1}$ $m^{-2}$]. Annual change in carbon content of the woody biomass can be computed by the difference between $C_{WBx}$ as the carbon stock in woody biomass of one year [kg $m^{-2}$] and $C_{WBx+1}$ as the carbon stock in the woody biomass in the following year [kg $m^{-2}$].

Quantification of $CO_2$ release and carbon storage by ecosystems is an important scientific field analyzing climate change impacts and feedbacks (GRAF et al. 2020, EUSKIRCHEN et al. 2016). The Kyoto protocol and Paris agreement defined the mitigation of $CO_2$ as an international task (ROGELJ et al. 2016). Therefore, the member states agreed to observe the changes in annual C pools and $CO_2$ fluxes as a baseline to define measures for land use policies to improve the storage potential of different ecosystems (DILLING et al. 2003). $CO_2$ sequestration of ecosystems are mainly monitored in rural areas, but different studies indicate that urban green infrastructure, especially urban forests, have great potential to act as carbon sinks as well (DOMKE et al. 2020, RICHTER et al. 2020, STROHBACH & HAASE 2013, NOWAK & CRANE 2002). As described in Part I – chapter 2, the area of urban forest stands in the Ruhr Metropolitan Region is estimated up to 1,036 km², which is a proportion of 23 % of total area in the agglomeration. Because these forests are located in one of the biggest agglomerations in Europe, most of this forested area is not or only extensively managed and is often used as free spaces and parks for the public, with tree logging typically limited to safety precautions. Therefore, these urban forest stands are allowed to grow old and are suitable for long-term carbon fixation in comparison to commercially used forests, which are typically logged before their carbon uptake is saturated.

## 2 Aim and design of the experiment

The aim of the experiment is to examine the differences in carbon stocks and $CO_2$ sequestration between the different forest types and to identify factors which influence the differences in $CO_2$ sequestration between the sites and forest types. To quantify how much carbon is stored, carbon stocks of the tree biomass, in the leaves, and in the soil were measured and modelled on the eleven monitoring sites already shown in the introduction (chapter 2.4). The $CO_2$ assimilation by tree growth, $CO_2$ translocation by litterfall, and the $CO_2$ release by soil respiration as components of the carbon cycle were examined for the years 2018 and 2019. As factors which potentially influence $CO_2$ sequestration, soil water and other physical, chemical, and biological soil attributes were measured. The study was carried out during a drought period, as described in chapter 2.6 of the introduction. Therefore, the resilience and vulnerability of different forest types to droughts were also analyzed.

# 3 Materials and Methods

## 3.1 Predicting carbon stocks and annual $CO_2$ assimilation in tree biomass

The method for calculating the C stocks and $CO_2$ assimilation in the tree biomass followed the approaches used for the National Forest Inventory and Greenhouse-Gas Monitoring in Baden-Wurttemberg and entire Germany according to PISTORIUS et al. (2006) and OEHMICHEN et al. (2011). The tree biomass was separated into the following C stocks: Aboveground C stocks in woody parts with diameter > 7 cm, aboveground C stocks in woody parts with diameter < 7 cm (brushwood), and belowground C stocks in the root biomass. A flowchart with the steps of analysis is given in Figure 17.

DBH (Diameter at Breast Height) at 1.3 m above soil surface and the tree and crown height were measured in the beginning of April 2018 before the vegetation period on every single tree within each monitoring site. Tree height (TH) was measured using the trigonometric method following WEST (2009). The distance between the viewpoint and the tree was measured with a distance laser (Leica Distance A6, Heerbrugg, Switzerland), the angle to the treetop, to the crown base, and to the trunk base were measured with a clinometer (SILVA Clino Master, Bromma, Sweden). The trees were labeled and the location where the DBH was measured was assigned with forestry chalk on each tree. The trees were remeasured in April 2019 and 2020 to determine annual DBH increment for the years 2018 and 2019.

DBH, TH, and the tree species were used as input variables for predicting the aboveground C stock in the woody biomass. To calculate the dry mass of the stem and branches with a diameter > 7 cm, the input variables were used for species-specific transfer functions (taper functions) to predict the fresh wood volume. The calculation was performed with the software BDATPro (KUBLIN & SCHARNAGL 1998, KUBLIN 2003). The fresh wood volume was multiplied with the species-specific volume shrinkage factors while drying and with the species-specific wood density factors, both published by KOLLMANN (1982), to predict the dry mass of the stem and branches > 7 cm. The advantage in using the wood density factors by KOLLMANN (1982) in comparison to others (IPCC 2003, KNIGGE & SCHULZE 1966) is that a minimum, a mean, and maximum potential wood density per species is given. Because different wood densities lead to large differences in total dry mass, it is possible to show uncertainties in the modelling approach by using species-specific minimum, mean, and maximum wood densities.

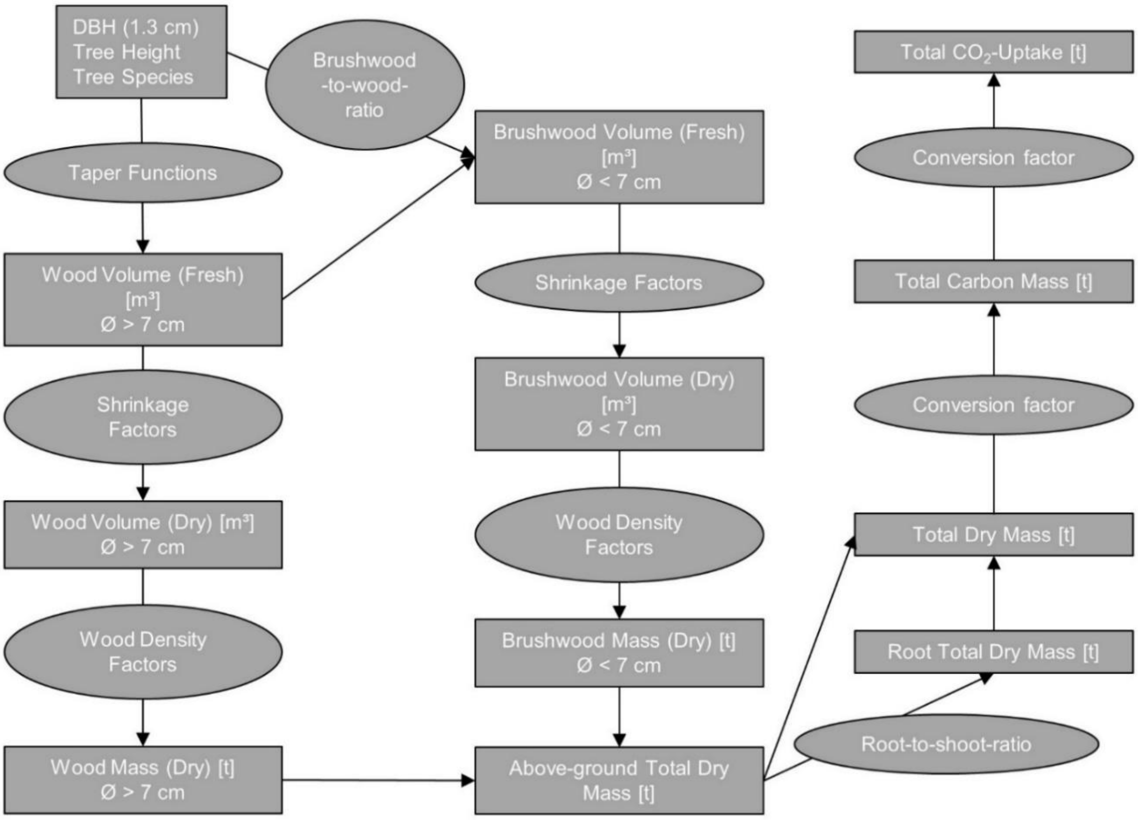


*Figure 17: Flowchart for predicting C stocks in the woody biomass. Rectangular boxes indicate variables and oval boxes calculations.*

The transfer functions only account for stems and branches > 7 cm in diameter. To predict the biomass of the brushwood (< 7 cm), brushwood-to-wood-ratios (bw-w-ratio) were determined from the mass panels by GRUNDNER & SCHWAPPACH (1952). The authors gathered the whole tree volume and the wood volume with a diameter > 7 cm, depending on DBH and TH, from a database of 71.051 trees from different species and locations. The difference between the whole tree volume and the wood volume with a diameter > 7 cm is equal to the volume of the brushwood with a diameter < 7 cm. This database was used to determine bw-w-ratios depending on the tree species, DBH, and TH. Bw-w-ratio and TH were correlated for each DBH (ranging from 7 cm to 100 cm) and a function were fitted via regression analysis. The following power function provided the best fitting:

$$bw - w - ratio = a * TH^{-b} \qquad [3]$$

where *bw-w-ratio* is the brushwood-to-wood-ratio, *TH* is the tree height [m] and *a* and *b* are coefficients. The coefficients for each species and DBH can be found in the appendix. Unfortunately, the database only contains data of four different deciduous tree species.

However, a model for other deciduous trees was developed by using the mean *bw-w-ratio* of all four different species in the database.

The *bw-w-ratio* were applied to the fresh wood volume of each tree to estimate the fresh volume of the brushwood. To calculate the brushwood dry mass, the brushwood volume was multiplied with the species-specific shrinkage factors while drying and the species-specific wood density factors, modified with wood density ratio by HAKKILA (1989). The author found out that brushwood has a higher wood density because of higher pressure loads. The difference between brushwood and wood density in ring-porous deciduous trees is 1.061 g $cm^{-3}$, in diffuse-porous deciduous trees 1.096 g $cm^{-3}$, and in coniferous trees is 1.34 g $cm^{-3}$ (PISTORIUS et al. 2006). Finally, the sum of dry mass of woody parts > 7 cm and dry mass of woody parts < 7 cm yields the total aboveground dry mass.

To extrapolate the root total dry mass, the root-to-shoot-ratio from the IPCC report (2003) were used. Before that, the aboveground dry mass was extrapolated over an area of one hectare by using the size of the monitoring sites shown in Table 4 (s. chapter 2.4 in the introduction). Following that, the root-to-shoot-ratio is 0.43 ± 0.24 when the aboveground biomass is < 75 t $ha^{-1}$, 0.26 ± 0.10 when the aboveground biomass is between 75 to 150 t $ha^{-1}$, and 0.24 ± 0.05 when the aboveground biomass is > 150 t $ha^{-1}$, respectively.

The sum of the aboveground and the belowground biomass is equal to the total dry mass in the woody biomass. The total dry mass is multiplied with a factor of 0.5 to convert the dry mass to the C content (WUTZLER et al. 2008). To calculate the $CO_2$ assimilation, the total C content is multiplied with a factor of 3.67 to convert the carbon content to $CO_2$ (ThüRIG & SCHMID 2008).

The model was performed for the measurements in 2018, 2019, and 2020. Annual $CO_2$ assimilation by the single trees was calculated by annual carbon stock changes in the living tree biomass.

## 3.2 Predicting $CO_2$ translocation by litterfall

To predict the $CO_2$ translocation by litterfall, the leaf area index (LAI) was measured and modelled over the experimental period. Leaf samples of the different tree species on the sites were collected and analyzed for their leaf mass area (LMA). LAI and LMA were than used to calculate the annual leaf dry mass and leaf carbon content. In the following, the single steps of the analysis are explained.

### 3.2.1 Measurement and modelling of LAI

The LAI was measured with the LAI-2200C Plant Canopy Analyzer (LI-COR, Lincoln Nebraska, USA) from March 2019 until May 2020 every two to three weeks along transects under the canopy using a 45° View Cap. A- and B-Readings were made in the same cardinal direction. Measurements were only done on days with uniform overcast or cloudless sky. When measurements were taken under cloudless sky, scattering correction was considered (KOBAYASHI et al. 2013). As the raw measurements under the canopy consider every tree element which blocks sunlight, the Plant Area Index (PAI) was measured (GARRIGUES et al. 2008). LAI was calculated based on PAI measurements performed below leafless canopy in winter.

As LAI was not measured continuously and for the complete experimental period (April 2018 - May 2020), LAI was additionally calculated from satellite images taken from the ESA Sentinel-2 Mission. 37 cloudless and suitable images from Sentinel-2 platform were taken from the study area between April 2018 and May 2020. The images were resampled and resized on the study area and the LAI was computed using the Biophysical Processor (WEISS & BARET 2016), integrated in the Sentinel Application Platform (ESA SNAP Version 7.0). A loess regression (CLEVELAND 1979) was performed on the single field measurements and the computed LAI from satellite images separately to interpolate the LAI development for both methods. The results from the loess regression for both methods were compared, and the LAI computed by satellite images were corrected by regression analysis using the LAI measured in the field. Additionally, leaf area density (LAD) was calculated by dividing the LAI from the mean stand crown height.

### 3.2.2 Investigating LMA

As leaf area gives only information about the surface area of leaves in the canopy, the total leaf mass or leaf carbon content had to be determined to calculate $CO_2$ translocation via litterfall. Therefore, 30 fully developed leaves of each tree species were collected on the sites, scanned, and the leaf area was calculated using ImageJ (FERREIRA & RASBAND 2012). After this, the leaves were dried at 60 °C over one week and weighed to determine the dry weight. The leaf area and the leaf dry mass was correlated by species and a linear regression was fitted. The division between the leaf area and the leaf dry mass represents the LMA [mg $cm^{-2}$].

The leaves were collected on the ground level, and therefore, the calculated LMA is only valid for the shaded tree crown. In forest stands, sun-exposed leaves are heavier and LMA increases exponentially in the vertical distribution of the stand due to light heterogeneity (HAGEMEIER 2002). To account the vertical differences between the shaded crown and the sun-exposed crown, the mean LMA for the stands were calculated according to NASAHARA et al. (2008):

$$LMA_a = \frac{(LMA_{top} - LMA_{bottom})}{(\ln LMA_{top} - \ln LMA_{bottom})} \qquad [5]$$

where $LMA_a$ is the mean LMA for the stand, $LMA_{top}$ is the LMA of the topmost leaves in the stand and $LMA_{bottom}$ is the LMA of the lowest leaves (ISHIHARA & HIURA 2011). HAGEMEIER

(2002) investigated the vertical distribution of LMA for *Betula pendula*, *Quercus robur, Fagus sylvatica, Tilia cordata,* and *Carpinus betulus* and LEGNER et al. (2013) for *Acer pseudoplatanus*. The authors introduced typical ratios between the LMA of sun-exposed and shaded leaves (Table 8), which are valid for Germany and those were used to calculate $LMA_{top}$ from $LMA_{bottom}$. Finally, the mean LMA of the stand was calculated using Eq. [5].

*Table 8: Typical ratio between the LMA of sun-exposed and shaded leaves for the six investigated tree species, following HAGEMEIER (2002) and LEGNER et al. (2013)*

| **Tree species** | **Ratio between the LMA of sun-exposed and shaded leaves [%]** | **Source** |
|---|---|---|
| *Betula pendula* | 55 | HAGEMEIER 2002 |
| *Quercus robur* | 91 | |
| *Carpinus betulus* | 200 | |
| *Tilia cordata* | 133 | |
| *Fagus sylvatica* | 178 | |
| *Acer pseudoplatanus* | 186 | LEGNER et al. 2013 |

In the last step, the carbon content of the leaves was calculated. The annual maximum LAI per site was taken and normalized to the leaf area per 100 m². Next, the leaf dry weight was calculated by multiplying the leaf area with the species-specific LMA. When the forest stand is built up by multiple tree species, the LMA was weighted with the DBH. Finally, the carbon content was calculated by multiplying the total dry weight with a factor of 0.47, which is the typical dry leaf carbon content in deciduous broad-leaved trees (MA et al. 2018) and the carbon content was than multiplied with 3.67 to convert the carbon content to $CO_2$.

## 3.3 Measurements of soil respiration, soil temperature, and moisture

Weekly measurements of soil respiration were performed with an EGM-3 Environmental Gas Monitor connected to a SRC-1 Gas Chamber (both PPSYSTEMS Amesbury, MA, USA) between July 2018 and November 2019. Unfortunately, the measurements did not cover the whole experimental period and data is missing for spring 2018 and winter 2019. For the measurements, four locations per site were selected (3 close to the location of soil sampling) and a PVC-collar (Ø 11 cm, height 10 cm) was inserted about 8 cm deep into the soil to hold the measurement system. Chamber closing time was 2 minutes and $CO_2$ concentration were recorded every 8 seconds. The $CO_2$ flux was calculated via quadratic fitting automatically by the device. In addition, soil temperature and moisture were measured to analyze their influence on soil respiration. Beside each collar a soil temperature and moisture probe (Decagon 5TE/5TM, Pullman, WA, USA) was installed in 5 cm depth and read out manually during flux measurements. One probe on each site was connected to a permanent data logger (Decagon EM50 Data Logger, Pullman, WA, USA), with a logging interval of 30 min. All $CO_2$

measurements below reading and all outliers (over 90 % quantile) were excluded from analysis.

## 3.4 Soil sampling and analysis

On three randomly chosen locations within each site, horizon-specific soil samples were taken in June 2018 up to the bedrock, to sealed layers on former coal mining sites, or to very dense soil layers in natural soils (e.g., Stagnosols). To measure soil bulk density and soil hydraulic properties of the mineral soil, undisturbed soil samples were taken using 250 $cm^3$ stainless steel rings. Additionally, disturbed samples were taken. All samples were kept cold at 4 °C prior to any analysis. Before analysis, the disturbed soil samples were sieved < 2 mm to estimate stone content and for homogenization. pH was measured in $CaCl_2$ (1:10) and soil organic carbon (SOC), N and C/N were determined using a vario MAX cube (Elementar Analysensysteme GmbH, Langenselbold, Germany). In soil samples with a high pH (> 6), inorganic C from carbonates was measured with a TOC-Analyzer (Mettler Toledo, Gießen, Germany) after destroying the organic C, and subtracted from the measured total C to recalculate SOC. C and N from microbial biomass ($C_{mic}$ and $N_{mic}$) were extracted by the Chloroform-Fumigation-Extraction-Method (JÖRGENSEN 1995a). Before the extraction, soil samples were adjusted to 60 % water holding capacity. Because the amount of microbial biomass rapidly decreases in the subsoil (JÖRGENSEN 1995a), $C_{mic}$ and $N_{mic}$ were measured in the organic layer and in the carbon rich topsoil horizons only. The soil carbon stock was calculated regarding the layer specific SOC content, bulk density, and respective horizon thickness. Finally, the calculated carbon stocks were normalized on 100 $m^2$ for comparison between the sites. Water retention curves of the undisturbed samples were measured using the Hyprop system and the saturated hydraulic conductivity ($K_s$) was measured using the $K_{sat}$ device (both Meter group Munich, Germany). To the data, the VAN GENUCHTEN (1980) model was fitted. Finally, the amount of plant available water was calculated between defined field capacity (FC) and permanent wilting point (pF 4.2).

## 3.5 Statistical operations

All statistical analysis were performed with Rstudio, version 1.2.5033, running with R version 3.6.3 (R Core Team 2020). Because the data was not normal distributed, non-parametric tests were used throughout the study and the data were not transformed. Test of significant differences were performed by Kruskal-Wallis-test (KRUSKAL & WALLIS 1952) and Dunn-Bonferroni Post-hoc-test (DUNN 1964). Principal Component Analysis (PCA) was used to identify soil properties which explain the differences in $CO_2$ assimilation, translocation, and release between both years and between the different sites. PCA was performed in Rstudio, using functions prcomp() and autoplot(). Additionally, correlation analysis for the principal components were performed using Spearman correlation (SPEARMAN 1904). Regression analyses were also performed in Rstudio.

# 4 Results

## 4.1 Forest structure, soil properties, and soil water supply at the monitoring sites

The species composition, tree age, and size of the monitoring sites were already shown in Table 4 in the introduction (chapter 2.4). Table 9 gives an overview of the forest structure on the sites.

*Table 9: Forest characteristics of the eleven monitoring sites. Standard deviation is provided. Mean LAI and LAD are given for the vegetation period (April – September) only.*

| **Urban Forest Type** | **ID** | **Tree Density** | **Mean DBH** | **Mean Height** | **Mean canopy height** | **Mean LAI** | **Mean LAD** |
|---|---|---|---|---|---|---|---|
| | | **[Trees 100 $m^{-2}$]** | **[cm]** | **[m]** | **[m]** | **[-]** | **[-]** |
| Semi-natural Forest | RePa-Bu | 5 | 47.7 ± 4.6 | 39.9 ± 2.5 | 27.2 ± 9.2 | 3.5 ± 0.8 | 0.13 ± 0.03 |
| | RePa-Ah | 5 | 34.6 ± 3.7 | 28.4 ± 2.3 | 19.2 ± 7.4 | 3.9 ± 1.0 | 0.20 ± 0.05 |
| | Hert-Bu | 3 | 46.2 ± 4.8 | 27.8 ± 2.7 | 19.9 ± 10.2 | 3.8 ± 1.0 | 0.20 ± 0.05 |
| | Hert-Ei | 3 | 35.7 ± 2.5 | 21.9 ± 1.8 | 16.5 ± 6.0 | 3.5 ± 0.9 | 0.21 ± 0.05 |
| Urban greening forests | ReHa-Ah | 32 | 14.2 ± 1.3 | 16.4 ± 1.4 | 9.5 ± 3.5 | 4.2 ± 1.1 | 0.50 ± 0.14 |
| | ReHa-Ei | 15 | 17.1 ± 1.5 | 16.0 ± 1.1 | 10.7 ± 4.0 | 2.3 ± 1.2 | 0.22 ± 0.12 |
| | BlueHa | 13 | 18.0 ± 1.3 | 18.4 ± 1.1 | 12.3 ± 3.7 | 2.6 ± 0.9 | 0.21 ± 0.08 |
| | BoHo | 22 | 13.2 ± 0.7 | 23.6 ± 1.2 | 13.3 ± 4.1 | 4.6 ± 1.1 | 0.34 ± 0.08 |
| Succession forests | WePa | 17 | 13.7 ± 1.0 | 16.8 ± 1.5 | 8.7 ± 5.2 | 1.6 ± 0.3 | 0.19 ± 0.04 |
| | ZeHa | 16 | 9.6 ± 0.8 | 10.8 ± 1.1 | 7.5 ± 3.8 | 2.5 ± 0.8 | 0.33 ± 0.11 |
| | ReHa-Bi | 28 | 18.5 ± 1.7 | 21.0 ± 2.6 | 13.1 ± 6.6 | 3.3 ± 0.8 | 0.25 ± 0.06 |

Because of their high stand age, the tree traits were largest in the semi-natural forests, while between the other forest types, which were significantly younger, no great differences existed. Semi-natural forests had also a lower tree density in comparison to the other forest types, because old forest stands have typically a larger distance between the single trees (PRETZSCH 2008).

Table 10 shows the soil properties of the monitoring sites. The semi-natural forests grew on deep native soils (Cambisols and Stagnosols) with a low stone content, low pH, and wide C/N and $C_{mic}/N_{mic}$ ratios. The vertical distribution of bulk density was typical for forest soils, where the bulk density in the carbon-rich A-horizon was small and large in the carbon-poor B-horizon (SCHEFFER & SCHACHTSCHABEL 2018). All semi-natural forest soils were unlimited in rooting, except at Hert-Ei, where a dense water logging-horizon was found. The urban greening forests mostly grew on former slag and slate heaps, and therefore, they were characterized by thin Regosols. Before tree planting, a topsoil was added above the anthropogenic substrates. Because of younger age and the influence of anthropogenic substrates, the soils had a higher stone content and pH compared to the semi-natural forests and the C/N and $C_{mic}/N_{mic}$ ratios were closer. On sites where demolition waste was found (ReHa-Ah, BlueHa, WePa), pH-values were neutral due to concrete and mortar components (MEUSER 2010).

*Table 10: Soil properties of the eleven sites. ± are the standard error (n=3). SOC = soil organic carbon,* $C_{mic}$ *= carbon content of the microbial biomass,* $N_{mic}$ *= nitrogen content of the microbial biomass.*

| **ID** | **Type** | **Soil Horizon** | **Depth** | **Stone content** | **Bulk density** | **pH** | $C_{mic}$ | $C_{mic}/N_{mic}$ | **SOC** | **C/N** |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | **[cm]** | **[%]** | **[kg dm$^{-3}$]** | **[-]** | **[µg g$^{-1}$]** | **[-]** | **[%]** | **[-]** |
| RePa-Bu | Cambisol | Of/Oh | 6.0 ± 0.6 | 7.3 ± 0 | 0.5 | 5.3 ± 0.1 | 753.5 ± 105.3 | 13.5 ± 3.1 | 11.3 ± 1.1 | 19.9 ± 0.8 |
| | | Ah | 13.7 ± 4.1 | 7.3 ± 0 | 0.7 | 3.8 ± 0.2 | 410.5 ± 29.2 | 9.5 ± 0 | 8.4 ± 0.1 | 21.5 ± 0.1 |
| | | Bv | 22.7 ± 2.2 | 0.9 ± 0.1 | 1.4 | 3.8 ± 0.6 | - | - | 2.0 ± 0.1 | 15.4 ± 0.4 |
| RePa-Ah | Cambisol | Of/Oh | 3.3 ± 0.9 | 2.5 ± 0.4 | 0.5 | 3.4 ± 0 | 897.1 ± 109.2 | 11.1 ± 0.5 | 14.6 ± 0.5 | 19.3 ± 0.8 |
| | | Ah | 11.3 ± 2.0 | 4.3 ± 1.1 | 0.9 | 3.4 ± 0.1 | 607.7 ± 79.0 | 10.9 ± 0.7 | 11.6 ± 2.3 | 21.4 ± 2.1 |
| | | Bv | 22.0 ± 0 | 2.9 ± 0.2 | 1.2 | 3.5 ± 0 | - | - | 2.0 ± 0.1 | 14.6 ± 0.3 |
| Hert-Bu | Stagnosol | Of/Oh | 6.3 ± 0.3 | 4.4 ± 1.6 | 0.5 | 3.2 ± 0 | 1,814.1 ± 261.5 | 8.6 ± 0.3 | 27.0 ± 1.1 | 18.1 ± 1.3 |
| | | Ah | 9.0 ± 0.6 | 1.7 ± 0.6 | 0.5 | 3.3 ± 0.1 | 759.8 ± 81.6 | 9.5 ± 0.3 | 12.6 ± 1.6 | 23.5 ± 0.4 |
| | | Sw | 13.3 ± 0.7 | 1.0 ± 0.6 | 1.1 | 3.5 ± 0.1 | - | - | 4.0 ± 1.0 | 16.5 ± 2.3 |
| Hert-Ei | Stagnosol | Of/Oh | 3.7 ± 1.2 | 6.4 ± 1.7 | 0.5 | 5.4 ± 0.4 | 681.0 ± 204.3 | 9.3 ± 1.3 | 6.8 ± 1.4 | 14.4 ± 1.4 |
| | | Ah | 18.0 ± 3.6 | 3.0 ± 1.3 | 1.0 | 5.0 ± 0.3 | 406.6 ± 81.6 | 8.4 ± 0.2 | 4.1 ± 0.5 | 12.7 ± 3.7 |
| | | Sw | 6.0 ± 0 | 0.6 ± 0 | 1.6 | 4.7 ± 0 | - | - | 1.0 ± 0 | 13.7 ± 0 |
| ReHa-Ah | Cambisol | Ah | 9.3 ± 1.3 | 3.8 ± 1.2 | 1.1 | 6.2 ± 0.2 | 696.2 ± 24.2 | 7.1 ± 0.3 | 3.4 ± 0.2 | 13.9 ± 0.5 |
| | | Bv | 19.7 ± 2.9 | 11.8 ± 4.8 | 1.2 | 7.1 ± 0.1 | - | - | 1.5 ± 0.3 | 13.9 ± 1.1 |
| ReHa-Ei | Regosol | Ah | 4.3 ± 0.3 | 5.5 ± 1.3 | 0.9 | 5.7 ± 0.4 | 742.3 ± 124.5 | 10.1 ± 1.0 | 7.3 ± 0.4 | 18.2 ± 1.9 |
| | | ylC | 9.7 ± 1.2 | 2.9 ± 0.3 | 1.1 | 4.9 ± 0.2 | - | - | 6.3 ± 0.1 | 21.9 ± 0.6 |
| BlueHa | Regosol | Ai | 10.3 ± 0.3 | 10.8 ± 4.3 | 1.1 | 7.0 ± 0.2 | 758.9 ± 177.7 | 5.9 ± 0.2 | 3.8 ± 1.0 | 14.2 ± 1.7 |
| | | ylC | 21.0 ± 2.6 | 15.7 ± 4.9 | 1.2 | 7.1 ± 0.2 | - | - | 1.4 ± 0.3 | 10.2 ± 0.9 |
| BoHo | Regosol | Ai | 4.0 ± 0 | 2.3 ± 1.2 | 1.2 | 4.6 ± 0.1 | 388.6 ± 55.2 | 9.8 ± 1.4 | 3.6 ± 0.4 | 15.2 ± 0.2 |
| | | ylC | 21.7 ± 1.9 | 1.2 ± 0.1 | 1.4 | 4.8 ± 0 | - | - | 2.4 ± 0.1 | 13.8 ± 0.2 |
| WePa | Regosol | Of/Oh | 8.3 ± 0.9 | 25.3 ± 2.9 | 0.5 | 6.8 ± 0.1 | 1,301.3 ± 124.8 | 5.9 ± 0.5 | 16.3 ± 1.8 | 21.7 ± 3.5 |
| | | ylC | 16.7 ± 3.2 | 47.1 ± 5.5 | 0.5 | 7.5 ± 0 | - | - | 9.4 ± 0.4 | 35.1 ± 3.9 |
| ZeHa | Regosol | Ai | 7.0 ± 0 | 67.6 ± 5.0 | 0.6 | 5.1 ± 0.1 | 489.7 ± 55.5 | 8.2 ± 0 | 12.5 ± 0.1 | 24.5 ± 0.4 |
| | | yC | 12.0 ± 1.5 | 73.0 ± 3.2 | 0.6 | 5.3 ± 0 | - | - | 15.1 ± 0.9 | 27.1 ± 1.5 |
| ReHa-Bi | Regosol | Ah | 7.7 ± 2.2 | 10.6 ± 3.6 | 0.8 | 4.1 ± 0.1 | 363.7 ± 15.6 | 7.9 ± 0.2 | 22.4 ± 12.2 | 30.2 ± 6.0 |
| | | ylC | 6.3 ± 0.9 | 10.5 ± 5.2 | 0.8 | 4.3 ± 0.1 | - | - | 29.5 ± 19.7 | 38.4 ± 6.8 |

The succession forests grew on former coal mining sites, too. In comparison to the urban greening forests, no topsoil was added. Consequently, they were characterized by the highest stone content (up to 73 %), lowest bulk density (0.5 - 0.8 kg dm$^{-3}$), and pH values from slightly basic to acidic, depending on the proportion of demolition waste and the weathering status of pyrite as component of slate (MEUSER 2010, KERTH & WIGGERING 1991). At the succession and urban greening forests the soils were limited in depth by a sealed soil layer.

The impact of the drought was reflected by the volumetric water contents measured for the vegetation periods 2018 and 2019 in comparison to the plant available water in the A-horizon at pressure head (h) = -100 cm, which were measured in the laboratory (Figure 18). At all sites the median is smaller than 0.2 cm$^3$ cm$^{-3}$, which shows that on all sites the soils were unsaturated during both entire vegetation periods. Smallest water contents were measured at

ZeHa with 0.1 cm$^3$ cm$^{-3}$, where the proportion of fine soil material is lowest and largest contents were measured at ReHa-Ei with 0.2 cm$^3$ cm$^{-3}$.

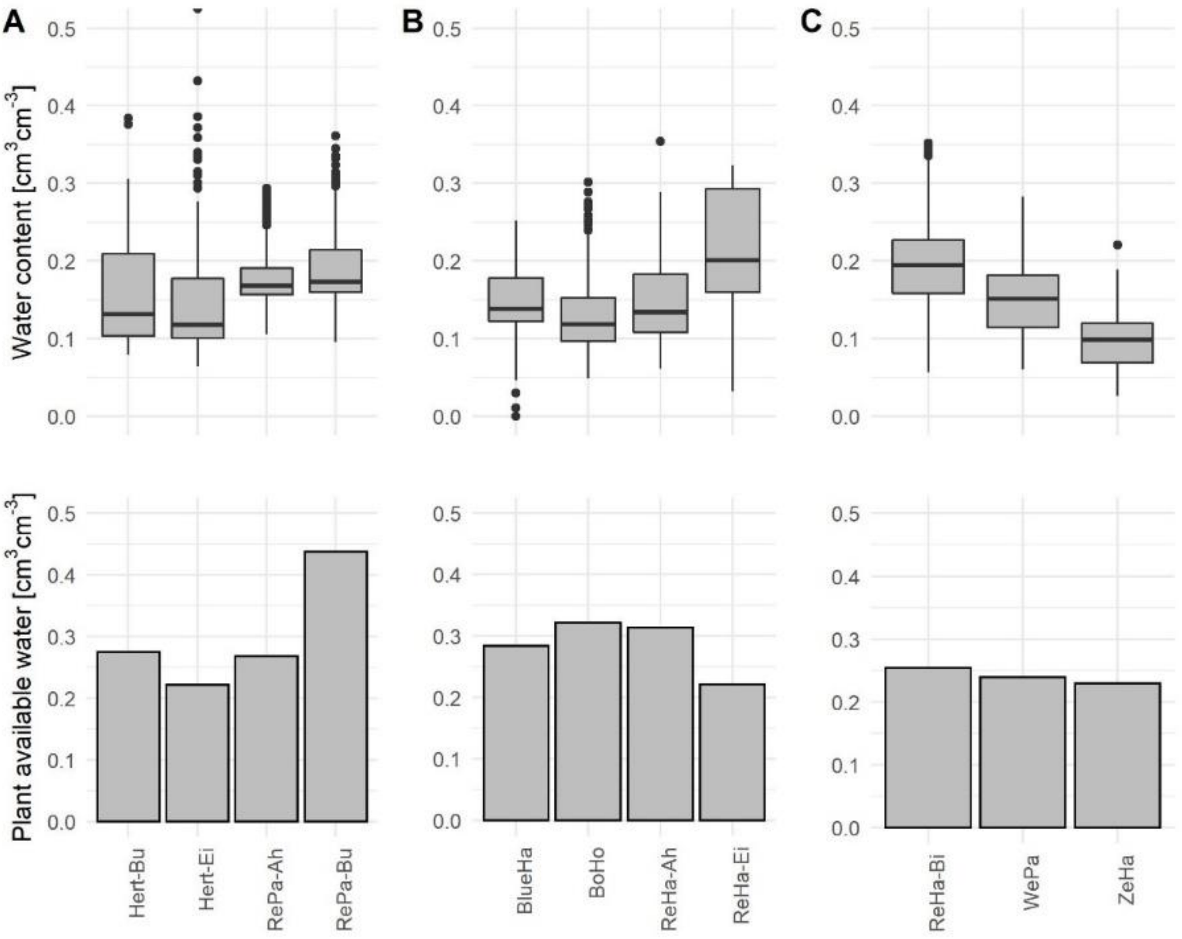


*Figure 18: Volumetric water content in 5 cm depth [cm$^3$ cm$^{-3}$] during the vegetation periods (April - September) of 2018 and 2019 and plaint available water [cm$^3$ cm$^{-3}$] for the A-horizon. A = semi-natural forests, B = urban greening forests, C = succession forests.*

## 4.2 Soil carbon stocks

C stocks differ between the sites and the forest types (Figure 19). Except of Hert-Ei, the largest average C stocks of about 1.8 to 2.2 t 100 m$^{-2}$ were found at the semi-natural forests. At Hert-Ei, soil C stock was relatively low (0.8 t 100 m$^{-2}$), almost on the same level as at the urban greening forests (0.7 to 0.9 t 100 m$^{-2}$). Slightly larger soil C stocks were found in the Regosols of the succession forests (1.3 to 1.7 100 m$^{-2}$). At one location at ReHa-Bi, SOC was about 46.8 % in the upper and 68.9 % in the lower soil (compare Table 10), which led to an extremely high C stock of about 6.0 t 100 m$^{-2}$. It is important to mention that SOC from coke and stone coal is not part of the carbon cycle in forests and does not improve the water and nutrient availability like SOC from organic debris.

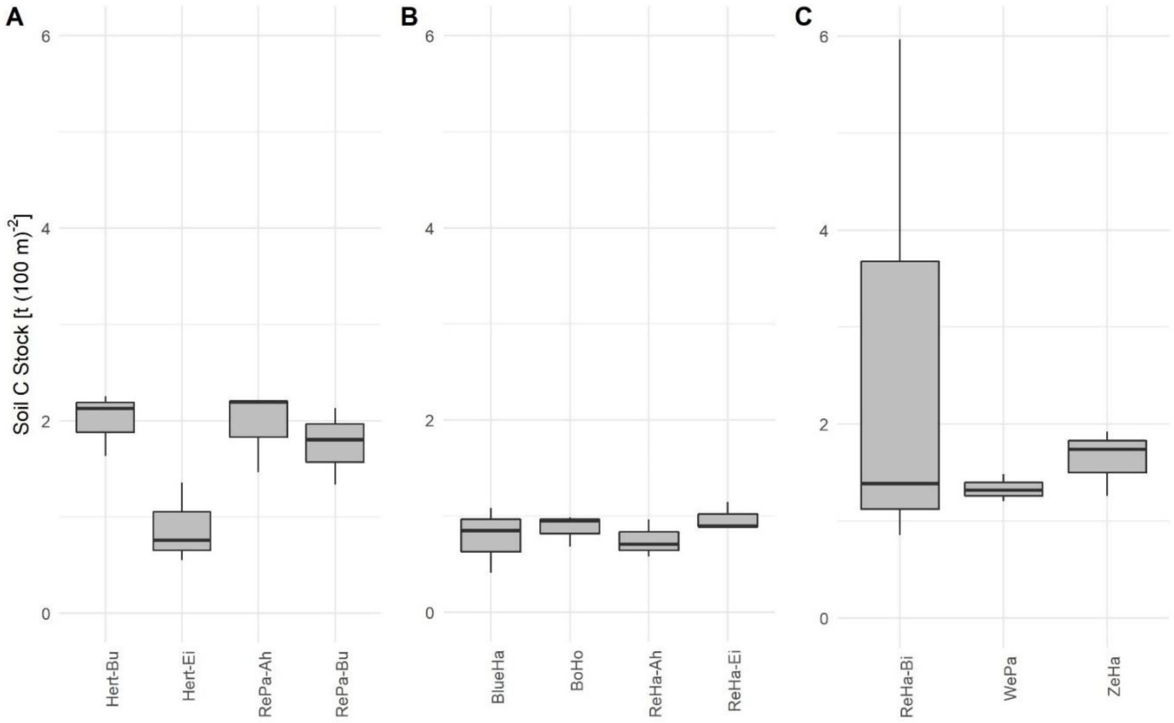


*Figure 19: Soil C stock at the monitoring sites separated by forest types (A = semi-natural forests, B = urban greening forests, C = succession forests).*

## 4.3 Carbon stocks in the tree biomass

Figure 20 shows the C stocks in the tree biomass on the monitoring sites separated by tree compartments and by using different wood densities published by KOLLMANN (1982). The figure shows how crucial the use of the right wood density in carbon balance modelling is, especially in forests with a large biomass pool. While at the site with the largest C stock (RePa-Bu), the carbon content ranged between 10.4 t 100 m$^{-2}$ using the maximum wood density and 5.7 t 100 m$^{-2}$ using the minimum wood density, the C stock on ZeHa, which is the smallest, ranged only between 0.5 and 0.3 t 100 m$^{-2}$ using the maximum and minimum wood density. As the real density remained unknown, the mean wood densities are used throughout the study.

In general, the total amount of carbon in the woody biomass is a function of tree size, and therefore, largest C stocks were found in the semi-natural forests. Especially, RePa-Bu with a mean C stock of 8.0 t 100 m$^{-2}$ highlighted the difference in comparison to the other semi-natural forests, which had C stocks of about 1.8 to 3.1 t 100 m$^{-2}$. This is caused by a mean tree height of 40 m, which is more than 10 m higher than the trees of the other sites. Additionally, the mean DBH is largest (47.7 cm). Second highest C stocks were calculated for the urban greening forests. At BoHo, BlueHa, and ReHa-Ei the C stocks were quite similar (1.3 to 1.4 t 100 m$^{-2}$), while at ReHa-Ah the stocks were larger (2.2 t 100 m$^{-2}$) due to higher tree density (32 per 100 m$^{-2}$). Smallest C stocks were found in the succession forests WePa and ZeHa (1.0

and 0.5 t 100 m$^{-2}$). In turn to these sites, the succession forest ReHa-Bi had with 4.7 t 100 m$^{-2}$ a very large C stock, which is caused by a relatively large mean DBH (18.5 cm), large mean height (21.0 m), and large tree density (28 per 100 m$^{-2}$). The allocation of C stocks in the biomass compartments are 70 % in wood, 20 % in roots, and in 10 % brushwood. Only at ZeHa, the C allocation differs with 60 % C stored in wood, 20 % in roots and brushwood, respectively, due to the small forest structure.

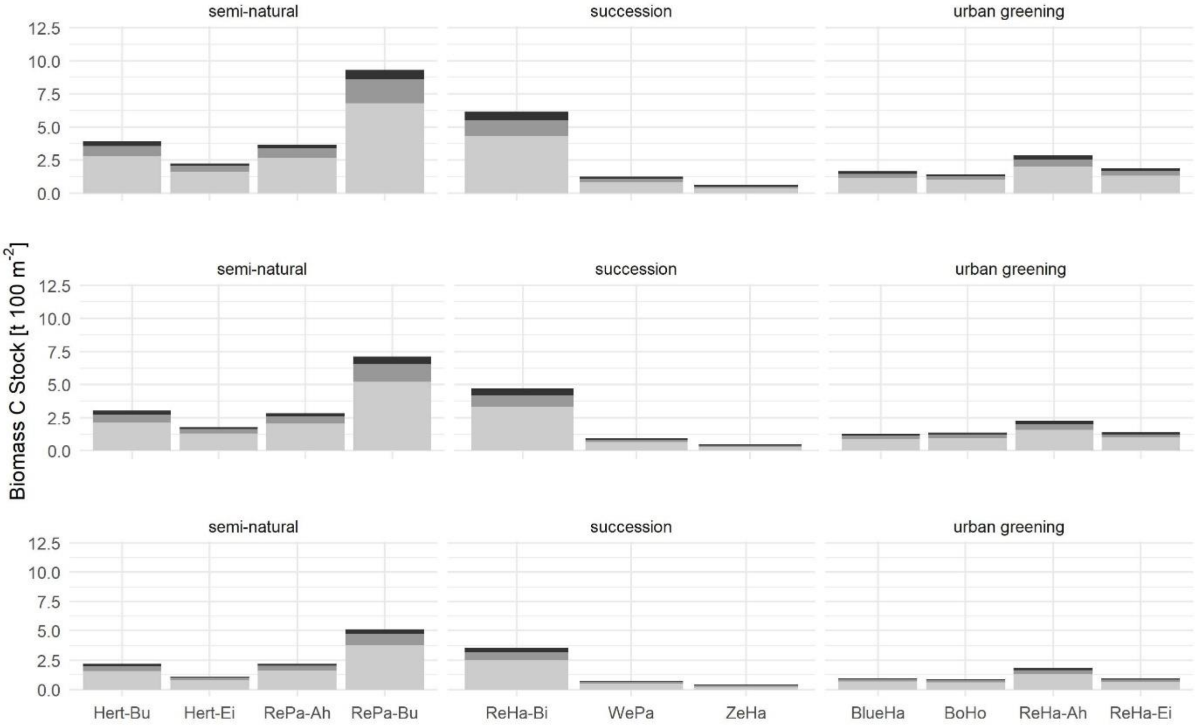


*Figure 20: C stocks in woody biomass separated by tree compartments on the monitoring sites using different wood densities by* K*OLLMANN (1982). First row presents the C stocks using maximum wood density, second row presents the C stocks using mean wood density and third row presents C stocks using minimum wood densities. Light grey indicates C stored in wood, grey indicates C stored in roots and black indicates C stored in brushwood.*

Figure 21 gives an insight in the model following the procedure described in part II – chapter 2.1. As can be seen clearly, biomass C stocks per tree can be calculated using following formula

$$C = (a * x + b)^2 \qquad [6]$$

where $C$ is the C stock [t] in all woody compartments, $x$ is the DBH [cm] measured in 1.3 m height and the tree height [m], respectively, and $a$ and $b$ are coefficients shown in Table 11. Overall, DBH fits better to the data (Adj. R² = 0.95) than height (Adj. R² = 0.68).

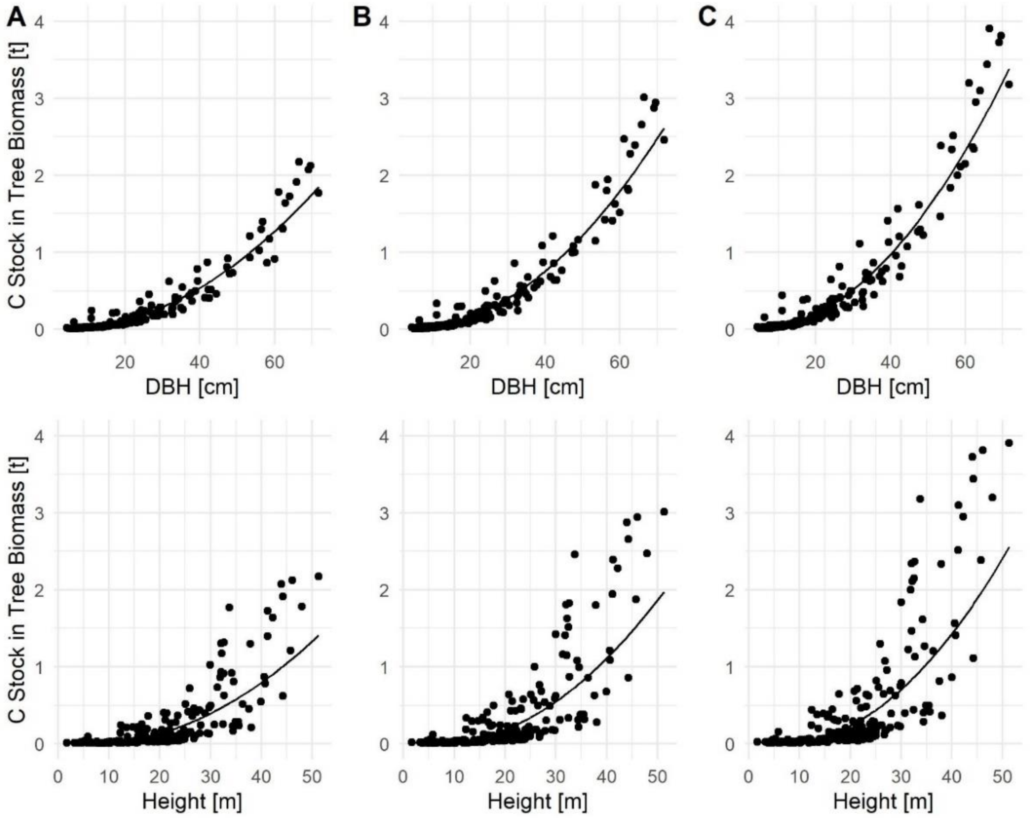


*Figure 21: Relationship between the DBH [cm], the tree height [m], and the C stock of the tree biomass (sum of wood, roots, brush) [t] for minimum wood density (A), mean wood density (B) and maximum wood density (C).*

*Table 11: Coefficients for calculating C stocks in tree biomass used in equation 4.*

| Wood Density | x | a | b | Adj. $R^2$ | p-value |
|---|---|---|---|---|---|
| Minimum wood density | DBH [cm] | 0.0197054 | -0.0545296 | 0.95 | < 2.2e-16*** |
| | Tree Height [m] | 0.026313 | -0.161516 | 0.68 | < 2.2e-16*** |
| Mean wood density | DBH [cm] | 0.0235411 | -0.0731494 | 0.95 | < 2.2e-16*** |
| | Tree Height [m] | 0.031253 | -0.197250 | 0.68 | < 2.2e-16*** |
| Maximum wood density | DBH [cm] | 0.0268516 | -0.0879685 | 0.95 | < 2.2e-16*** |
| | Tree Height [m] | 0.035628 | -0.229106 | 0.68 | < 2.2e-16*** |

## 4.4 Annual $CO_2$ assimilation in the woody biomass

### 4.4.1 Annual tree-specific $CO_2$ assimilation

As the annual changes of DBH, tree height, and $CO_2$ assimilation were not normally distributed, the median of the data is given in the following. As can be seen in Figure 22, annual DBH increment in 2018 was largest in the semi-natural forests (2.25 ± 0.3 mm), followed by the urban greening forests (2.0 ± 0.2 mm), and the succession forests (1.0 ± 0.2 mm). At the urban greening forests the annual increment in tree height was largest (11.6 ± 1.3 cm), followed by the semi-natural forests (9.9 ± 1.1 cm), and the succession forests (7.6 ± 3.8 cm).

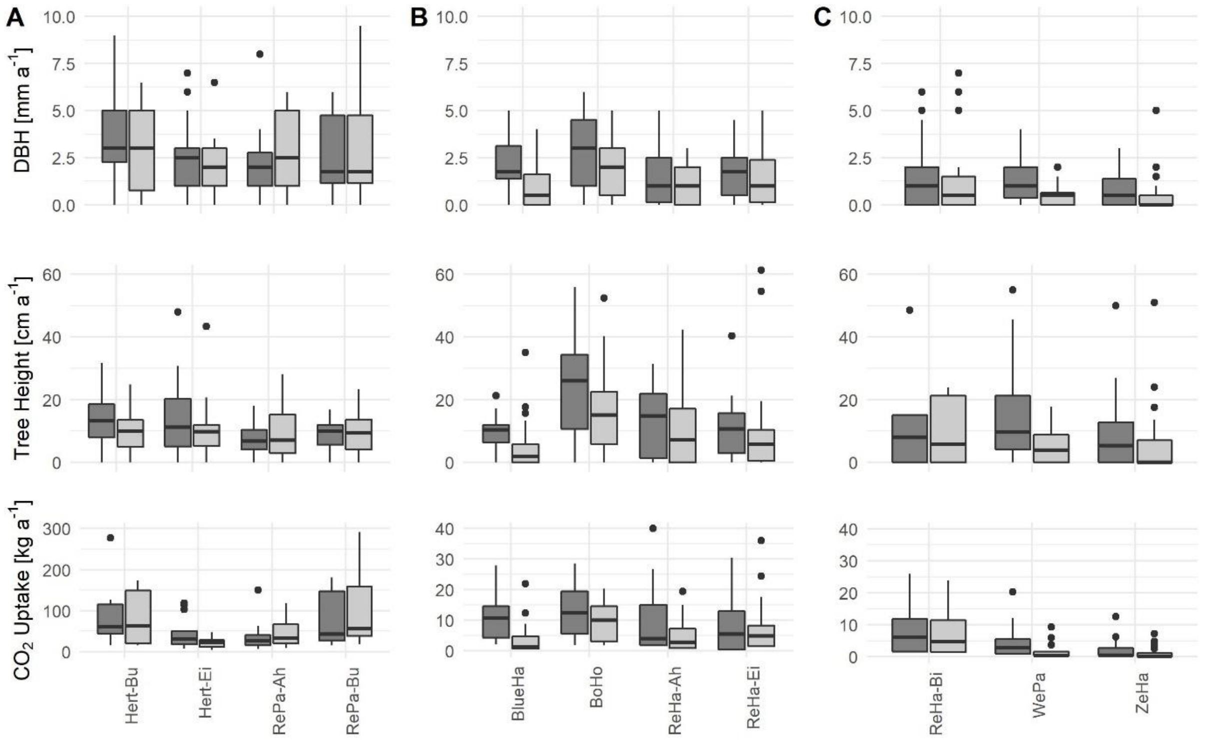


*Figure 22: Annual increment of DBH and tree height and annual CO₂ assimilation by the single trees in the woody biomass in 2018 (dark grey) and 2019 (light grey) for the different sites. A: semi-natural forests, B: urban greening forests, C: succession forests. Mind the different scale of the y-axis on the charts presenting the CO₂ assimilation.*

The annual change in tree structure led to different amounts of annual assimilated $CO_2$. With a median of 33.9 ± 7.2 kg per tree in 2018 the largest $CO_2$ assimilation was found in the semi-natural forests. In comparison, the amount of $CO_2$ assimilation in the urban greening (6.7 ± 0.9 kg per tree) and in the succession forests (2.2 ± 0.6 kg per tree) were quite small. These differences are caused by different tree sizes at the sites. In general, the relationship between DBH and C stock is exponential (Figure 21), and therefore, a small DBH increment for small trees lead to small absolute changes in $CO_2$ assimilation, while even a small DBH increment for large trees will lead to absolute large changes in $CO_2$ assimilation. For many temperate and tropical tree species it is reported that the mass growth rate increases continuously with tree size (STEPHENSON et al. 2014).

By grouping the results from the single sites according to forest types, the succession forests assimilated significantly less $CO_2$ per tree than the semi-natural forests in both years (Figure 23). Even the urban greening forests assimilated less $CO_2$ per tree and year than the semi-natural forests, but no significant differences in $CO_2$ assimilation per tree was found between the succession and the urban greening forests.

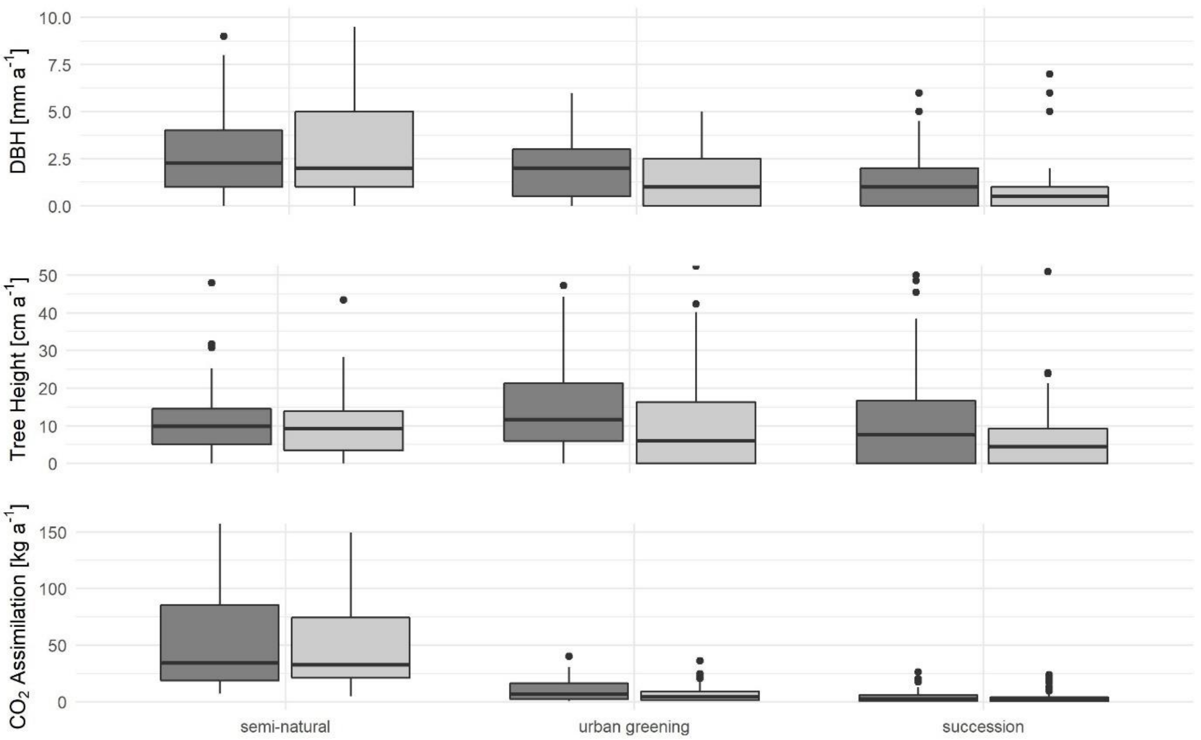


*Figure 23: Annual increment of DBH and tree height and annual CO$_2$ assimilation by the single trees in the woody biomass in 2018 (dark grey) and 2019 (light grey) referring to forest types.*

In 2019, the DBH and tree height growth, and the CO$_2$ assimilation per tree was slightly lower compared to 2018 by about 1.3 kg per tree at the semi-natural, 2.5 kg per tree at the urban greening and 0.8 kg per tree at the succession forests, but the differences between the years were not significant for the specific sites, nor between the forest types. In contrast, at the two beech forests (RePa-Bu and Hert-Bu) and at the sycamore forest RePa-Ah, CO$_2$ assimilation per tree was higher in 2019 compared to 2018 (Figure 22).

It was hypothesized that differences between 2019 and 2018 in CO$_2$ assimilation ($\Delta CO_2$) per tree were caused by water limitation in consequence of the ongoing drought. To test the hypothesis, a PCA analysis was performed using those soil properties affecting the water availability (Figure 24). $\Delta CO_2$ was correlated with soil carbon stocks, the depth of the effective rooting zone, the plant-available water, and mean volumetric water content during the vegetation period measured at 5 cm depth, while $\Delta CO_2$ was not correlated to the proportion of coarse soil and bulk density. Subsequent, correlation analysis using Spearman correlation coefficient shows that the few forests which showed an increase of CO$_2$ assimilation between both years, are characterized by a larger effective rooting zone, a larger soil C stock, and a larger amount of plant available water in the A-horizon. Vice versa, the decrease in CO$_2$ assimilation was even higher, the smaller the effective rooting zone, the soil C stock, and the amount of plant available water was. For the variables “plant-available water in the B-horizon” and “volumetric water content at 5 cm depth during the vegetation period”, no significant correlations were found. The latter might be caused by the tree rooting patterns, which enable acquiring water from deeper soil horizons, while the volumetric water content was only measured at 5 cm depth.

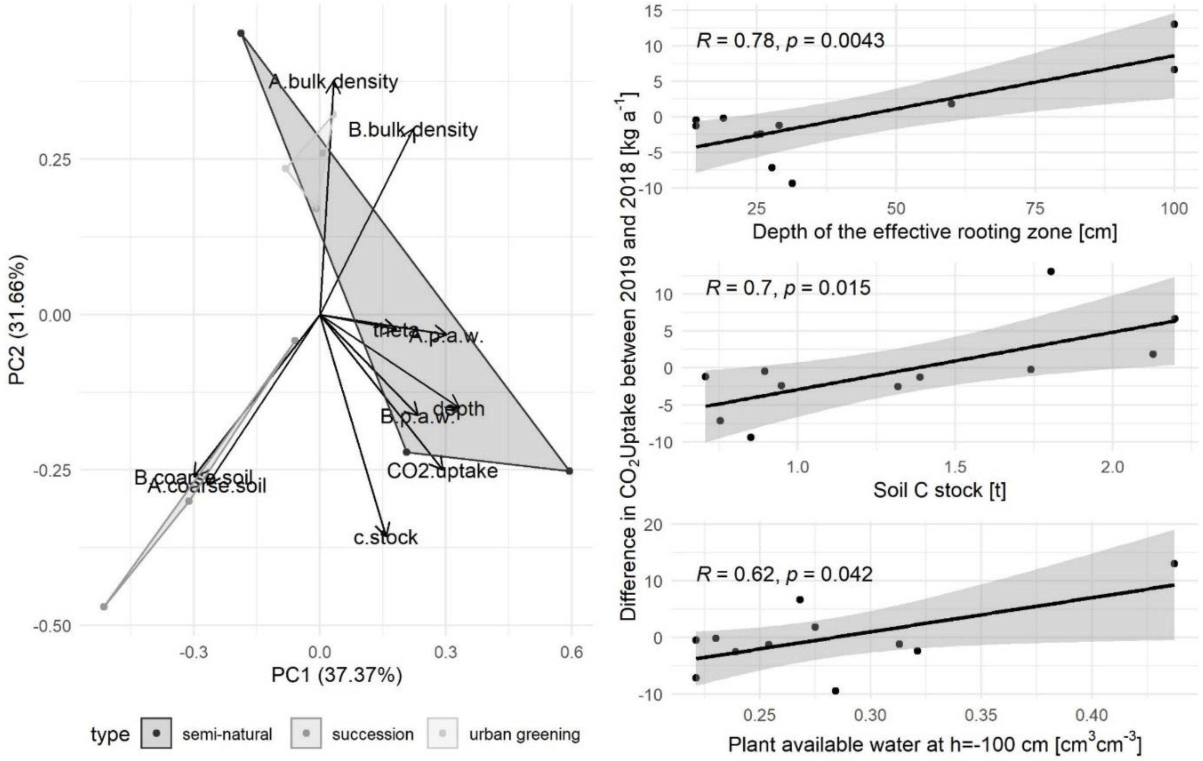


*Figure 24: PCA analysis for Δ $CO_2$ assimilation per tree between 2019 and 2018 and different soil properties influencing the water availability. depth = depth of the effective rooting zone [cm]; A.p.a.w. = Plant-available water measured for h=100 cm in the A-horizon; B.p.a.w. = plant-available water measured for h=100 cm in the B-horizon; theta = median of soil volumetric water content during vegetation period (April – September) measured in 5 cm depth [$cm^3$ $cm^{-3}$]; A.coarse.soil = proportion of coarse soil in the A-horizon [%]; B.coarse.soil = proportion of coarse soil in B-horizon [%]. On the right: correlation analysis between the difference in for Δ $CO_2$ assimilation between 2019 and 2018 and different soil parameters which indicate soil water availability.*

### 4.4.2 Annual stand-specific $CO_2$ assimilation

The $CO_2$ assimilation of each tree is summed up and normalized to an area of 100 $m^2$ to calculate the $CO_2$ assimilation for the entire forest stands per year and forest type. As can be seen in Figure 25, $CO_2$ assimilation increased from 181.7 ± 49.1 in 2018 to 206.7 ± 77.4 kg 100 $m^{-2}$ in 2019 at the semi-natural forests, irrespectively of the continuing drought, while the urban greening forests sequestered 82.7 kg 100 $m^{-2}$ less in 2019 compared to 2018, and the succession forests 52.0 kg 100 $m^{-2}$ less, by comparing the medians of both years. In 2018, the urban greening forests assimilated as much $CO_2$ as the semi-natural forests (200-300 kg $CO_2$ $a^{-1}$ 100 $m^{-2}$). However, in 2019 the patterns turned, and the semi-natural forests assimilated noticeably more $CO_2$ than the urban greening forests. The succession forests assimilated only 50-150 kg $CO_2$ $a^{-1}$ 100 $m^{-2}$.

All urban greening forests showed a decrease in $CO_2$ assimilation in 2019 compared to 2018. The strongest decrease of about 64 % was found at BlueHa, where 45 % of all trees (mainly *Acer pseudoplatanus*) died during the vegetation period of 2019. Also, the *Acer* *pseudoplatanus* trees on ReHa-Ei died, but already in 2018, and no remarkable changes in $CO_2$ assimilation between the years were detectable (Figure 25). The strongest decrease in $CO_2$ assimilation of about 69 % and 45 % between the years was observable at WePa and ZeHa. At ReHa-Bi the decrease was about 10 %.

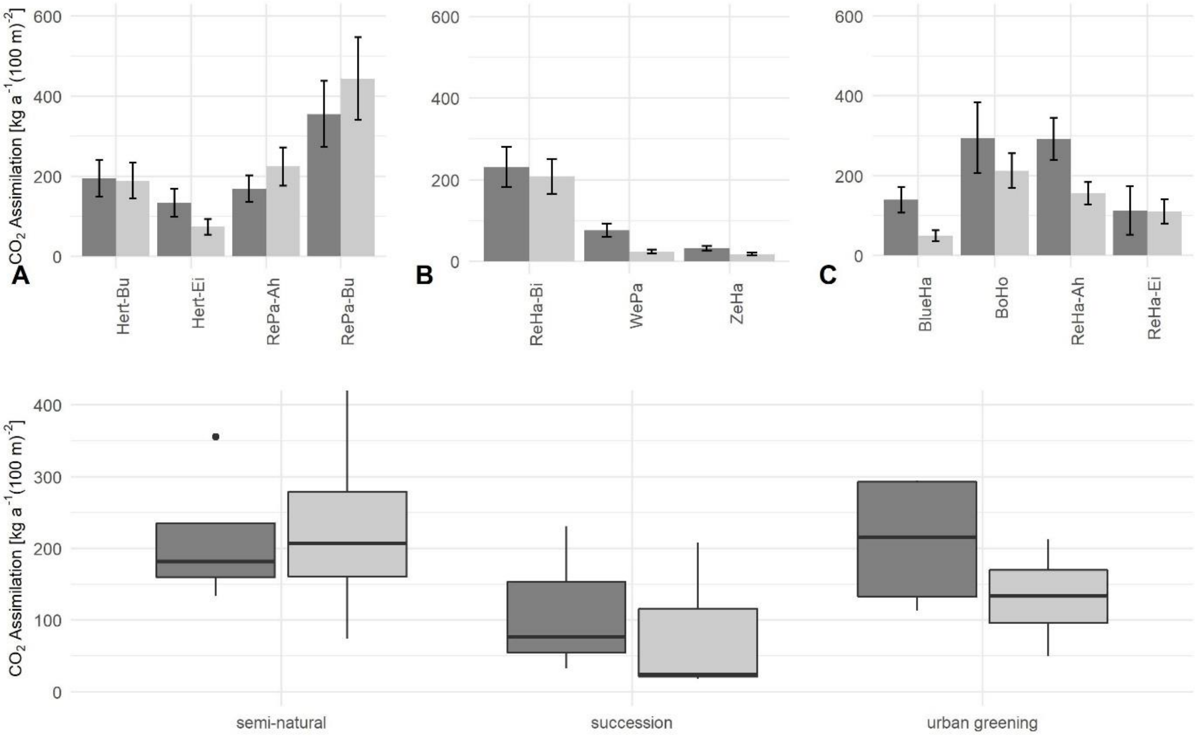


*Figure 25: $CO_2$ assimilation on the single sites (semi-natural forests (A), succession forests (B) and urban greening forests (C)) and $CO_2$ assimilation by the different urban forest types. 2018 = dark grey, 2019 = light grey.*

## 4.5 Annual $CO_2$ translocation by litterfall

### 4.5.1 Correction of the LAI computed by satellite images using field data

In crops, LAI computed from Sentinel-2 images with the Biophysical Processor (WEISS & BARET 2016) is quite precisely (XIE et al. 2019, DONG et al. 2020), whereas BROWN et al. (2019) showed for forests, that the algorithm underestimates the LAI in comparison to field measurements. The data from the sites agrees with BROWN et al. (2019), as the LAI computed from Sentinel-2 images underestimated the LAI in comparison to field measurements and the underestimation is even larger, the larger the LAI from measurements was (Figure 26): On WePa, measured LAI was smallest and the Sentinel-2 data fitted very close to the field data. In comparison, measured LAI was largest on BoHo, and the deviation from satellite derived data was largest.

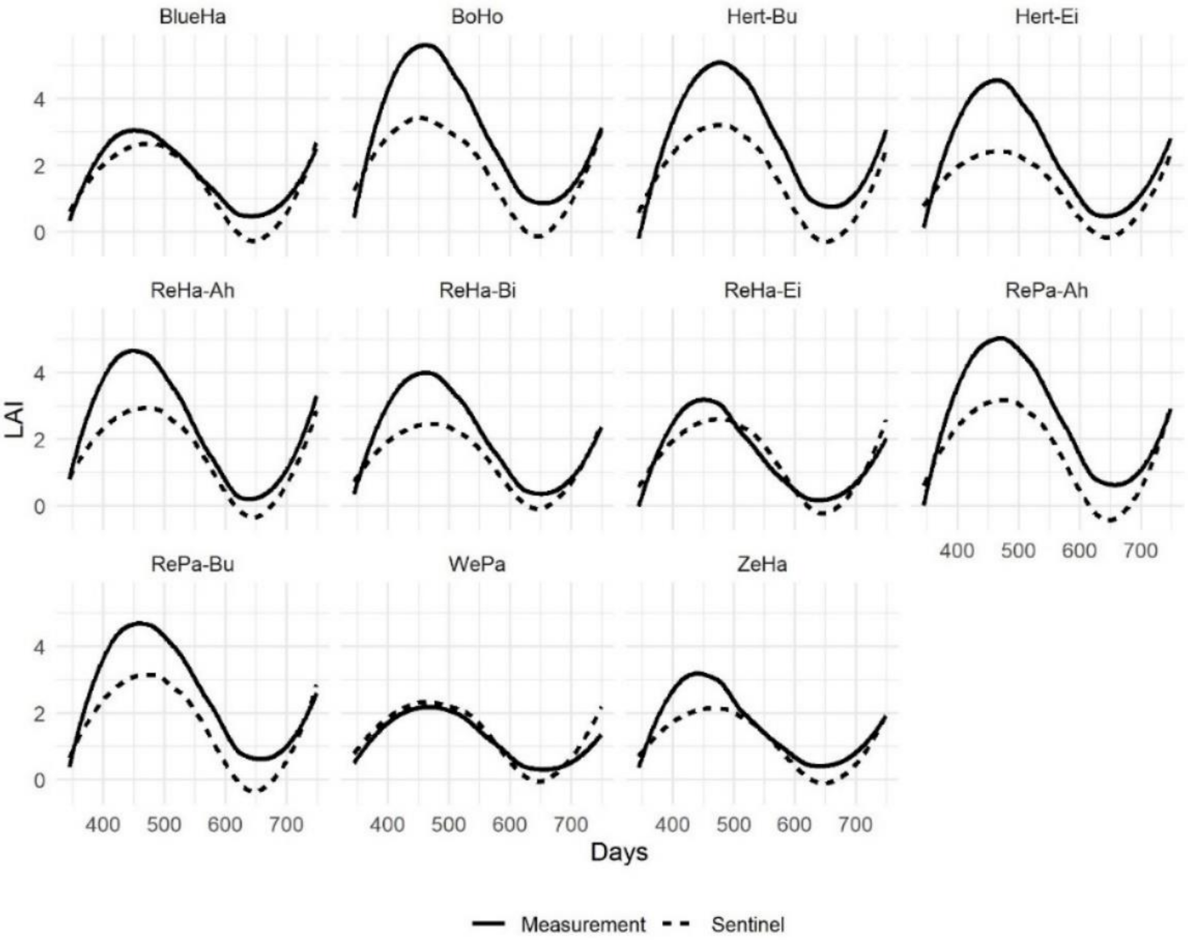


*Figure 26: Comparison between the LAI measured in field and the LAI computed from the Sentinel-2 images on each site between March 2019 and May 2020. The curves represent the result from the loess regression performed on the single measurements and computations from satellite images.*

*Table 12: Coefficients for LAI correction from Sentinel-2 satellite images for each site used in equation 4.*

| site | a | b | adj. R² | p |
|---|---|---|---|---|
| BlueHa | 0.362907 | 0.753007 | 0.96 | < 2.2e-16 |
| BoHo | 0.40942 | 0.886407 | 0.91 | < 2.2e-16 |
| Hert-Bu | 0.405736 | 0.913479 | 0.91 | < 2.2e-16 |
| Hert-Ei | 0.552105 | 0.704887 | 0.93 | < 2.2e-16 |
| ReHa-Ah | 0.506148 | 0.653392 | 0.98 | < 2.2e-16 |
| ReHa-Bi | 0.54446 | 0.599086 | 0.95 | < 2.2e-16 |
| ReHa-Ei | 0.468063 | 0.48413 | 0.93 | < 2.2e-16 |
| RePa-Ah | 0.408201 | 0.884442 | 0.91 | < 2.2e-16 |
| RePa-Bu | 0.40196 | 0.86693 | 0.94 | < 2.2e-16 |
| WePa | 0.39347 | 0.539317 | 0.96 | < 2.2e-16 |
| ZeHa | 0.493599 | 0.628984 | 0.93 | < 2.2e-16 |

Because the underestimation is systematical, it was possible to correct the LAI from the satellite data using regression analysis. For the period when the LAI measurements and the computed LAI overlapped (March 2019 - May 2020), the interpolated curves were compared for each site and functions were fitted by regression analysis to adjust LAI computed from Sentinel-2 images to the field data. The following formula was used:

$$LAI_{corr} = (a * x + b)^2 \quad [7]$$

where $LAI_{corr}$ is the corrected LAI, $x$ is the LAI derived from satellite images and $a$ and $b$ are coefficients shown in Table 12 for each site.

Figure 27 shows that after the correction the LAI values computed by Sentinel-2 images fits close enough to the LAI values measured in the field.

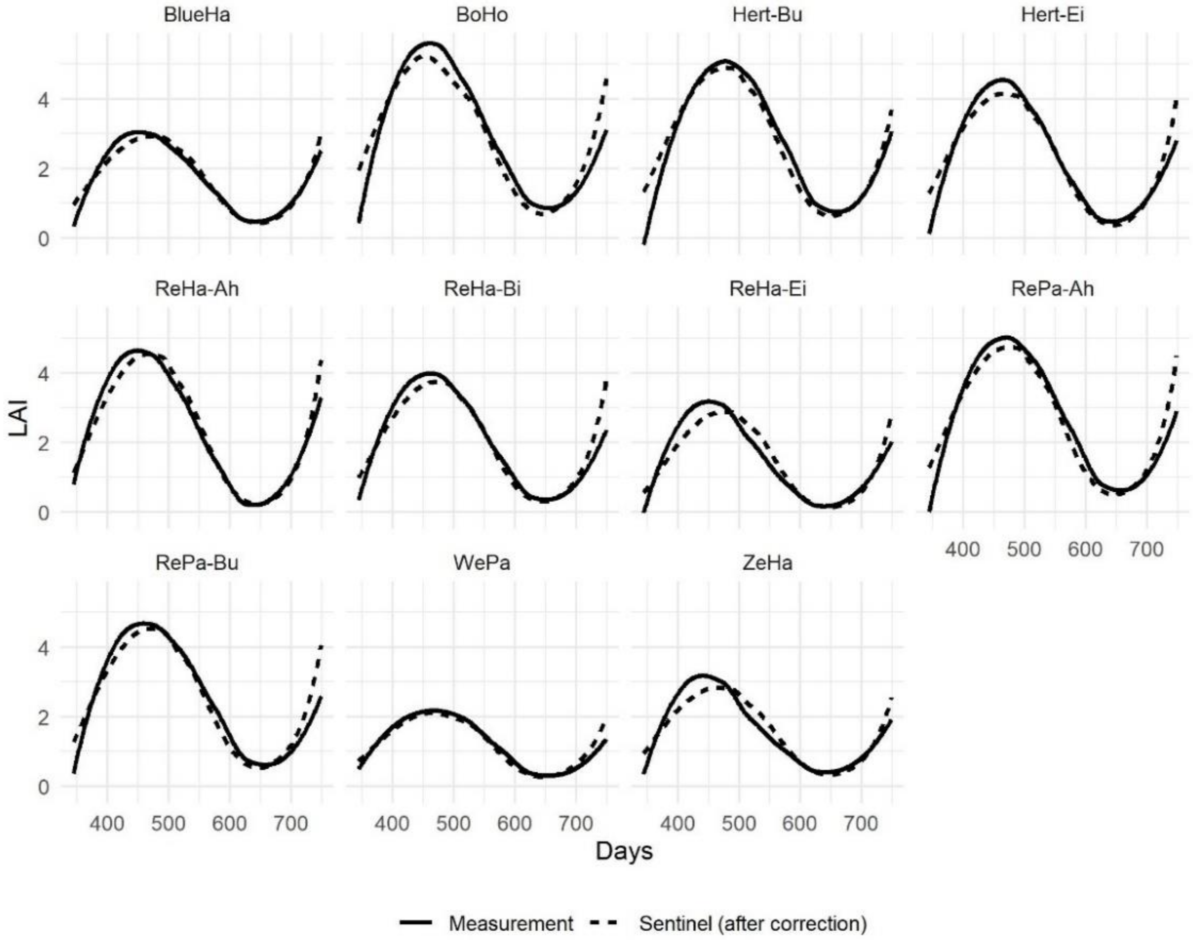


*Figure 27: Corrected LAI development from Sentinel-2 images in comparison to the LAI evolution measured in the field between March 2019 and May 2020.*

### 4.5.2 LAI development on the single sites

Figure 28 depicts the LAI development between April 2018 and December 2019. Largest LAI were found in 2018 for the urban greening forests (6.4 ± 0.5), followed by the semi-natural forests (5.3 ± 0.2), and the succession forests (4.0 ± 0.9). At the succession forests, LAI differed relatively strong between the sites, which is caused by the species composition. On WePa *Betula pendula* is the only tree species, resulting in a relatively small LAI (2.2), while on the other sites *Betula pendula* is mixed with *Tilia cordata* and *Acer pseudoplatanus*, which have wider leaves, resulting in larger LAI (4.8 and 4.9).

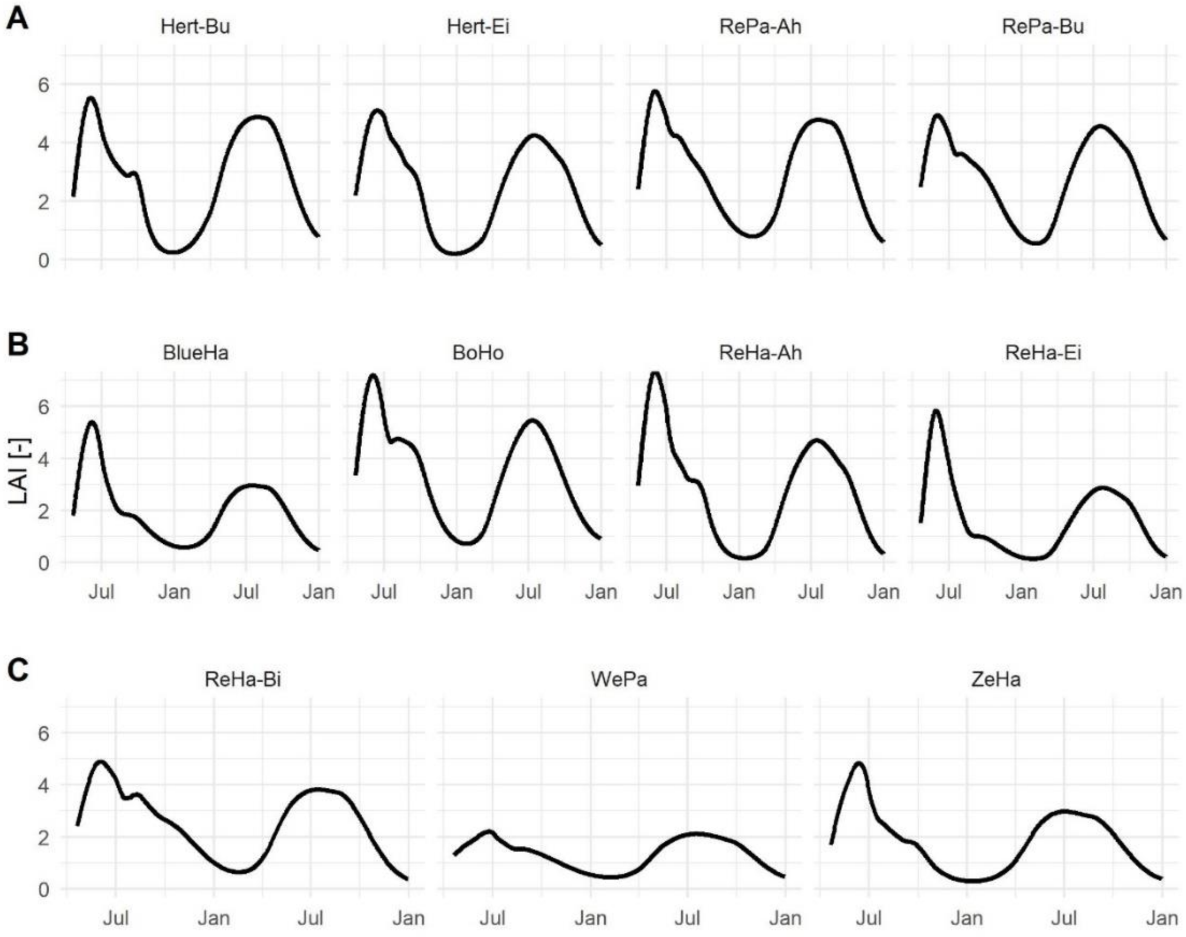


*Figure 28: LAI development at the sites between April 2018 and December 2019. Data taken from Sentinel-2 images. A = semi-natural forests, B = urban greening forests, C = succession forests.*

At all sites, LAI was larger in 2018 than in 2019. In general, the decrease in LAI was largest in the urban greening forests (38.0 %). At the single sites, the decrease was largest for ReHa-Ei (50.8 %) and BlueHa (45.3 %), where many trees died during the experimental period. Smaller changes were found in the semi-natural forests (-13.5 %) and at the succession forests (-25.3 %). The maximum LAI was reached very early in June on all sites in 2018, while in 2019 the leaf development occurred much slower, and the maximum LAI was reached later in July (Figure 28).

### 4.5.3 LMA

The relationship between leaf area [cm²] and leaf dry mass [mg] for the leaf samples collected at the sites can be seen in Figure 29. For all six species, linear correlations were found, indicating that the dry mass increased systematically with leaf size. Furthermore, the slope indicates, that the relationship differs strongly between the species. *Quercus robur* and *Betula pendula* have heavy leaves as an adaptation on open spaces and open forest stands with high solar radiation, where the trees find their ecological optimum (NIINEMETS & VALLADARES 2006), while in comparison the leaves of *Fagus sylvatica*, a tree species which is able to grow in deep woodlands under full shade (PETRITAN et al. 2007), are relatively light-weighted.

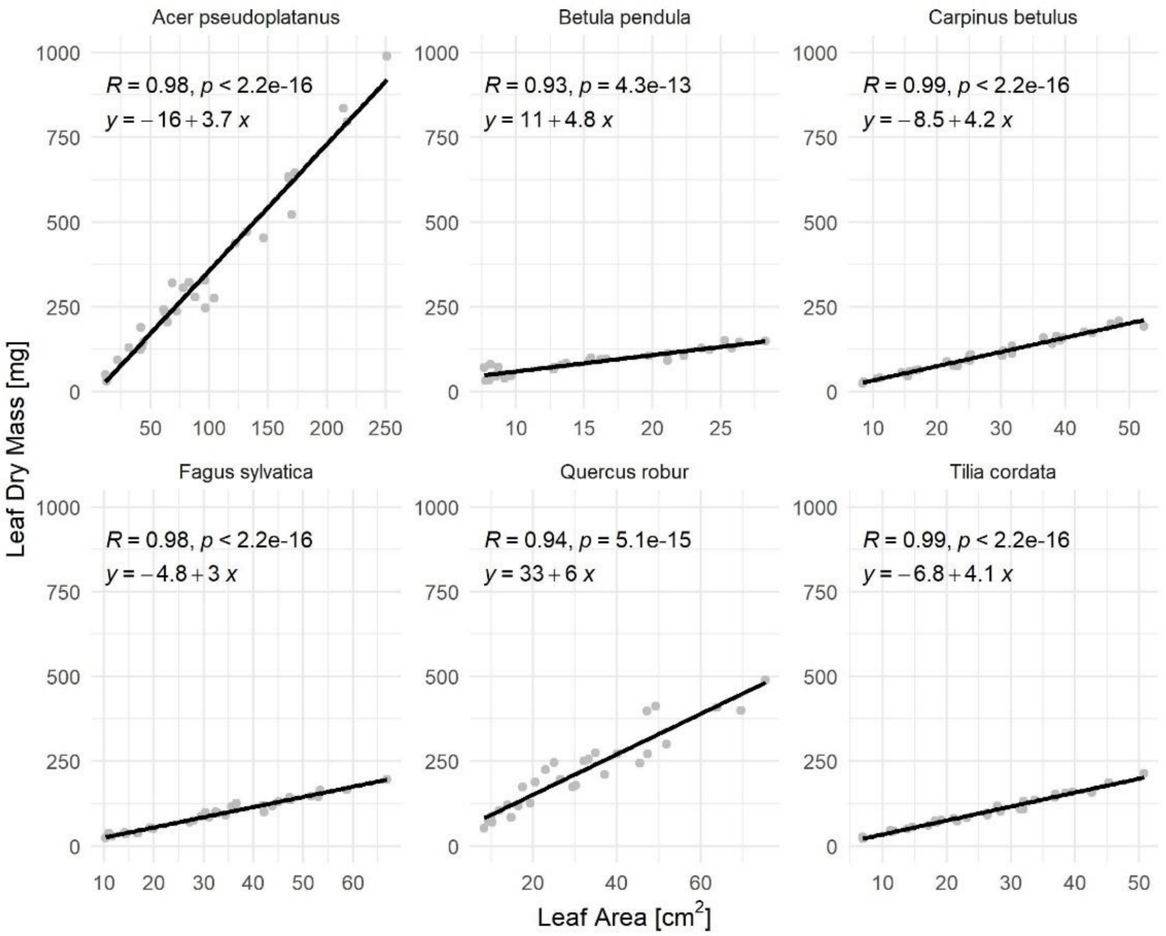


*Figure 29: Relationship between the leaf area [cm²] and the leaf dry mass [mg] of the 30 leaf samples per species collected at the monitoring sites*

*Table 13: Species-specific LMA [mg cm$^{-2}$] calculated from the leaf samples collected at the sites. Standard error is given. Factor for calculating LMA of sun-exposed leaves taken from HAGEMEIER (2002) and LEGNER et al. (2013). For comparison, the range of typical LMA values from Forrester et al. (2017) are provided.*

| **Tree species** | **LMA of shaded leaves** | **Factor for calculating LMA of sun-exposed leaves** | **Mean LMA with respect of the vertical distribution** | **Range of LMA values from the literature** |
|---|---|---|---|---|
| | **[mg cm$^{-2}$]** | **[-]** | **[mg cm$^{-2}$]** | **[mg cm$^{-2}$]** |
| *Acer pseudoplatanus* | 3.5 ± 0.1 | 2.86 | 6.2 | 4.7 - 8.8 |
| *Betula pendula* | 5.7 ± 0.3 | 1.55 | 7.2 | 4.9 - 22.7 |
| *Carpinus betulus* | 3.8 ± 0.1 | 3.00 | 6.9 | 4.1 - 5.8 |
| *Fagus sylvatica* | 2.8 ± 0.1 | 2.78 | 4.9 | 3.3 - 8.9 |
| *Quercus robur* | 7.3 ± 0.3 | 1.91 | 10.3 | 4.2 - 11.6 |
| *Tilia cordata* | 3.8 ± 0.1 | 2.33 | 6.0 | 3.2 - 5.1 |

These relationships are also represented by the LMA [mg cm$^{-2}$] shown in Table 13, which was used to calculate leaf mass for the single sites. Except of *Carpinus betulus* and *Tilia cordata*, all mean LMA values are within the typical species-specific range reported by FORRESTER et al. (2017). The reason why these two species exceed the typical range might be that for both species only three studies were considered by FORRESTER et al. (2017), which may not have covered all possible LMA values.

### 4.5.4 Leaf carbon stock and $CO_2$ translocation via litterfall

Figure 30 shows the calculated annual leaf dry mass, leaf C stock, and $CO_2$ translocation via litterfall at the sites. Depending on species composition and LAI, the $CO_2$ translocation differed between 26 and 90 kg $a^{-1}$ 100 $m^{-2}$. In 2018, $CO_2$ translocations smaller than 50 kg $a^{-1}$ 100 $m^{-2}$ were found in forests with small LMA (beech forests RePa-Bu and Hert-Bu) and on sites with small LAI (WePa), while $CO_2$ translocations larger than 50 kg $a^{-1}$ 100 $m^{-2}$ were found in the oak forests and birch-mixed forests (Hert-Ei, ReHa-Ei, ReHa-Bi, ZeHa) due to large species-specific LMA or large LAI (BoHo, ReHa-Ah). Like LAI, the $CO_2$ translocation decreased in 2019 in comparison to 2018 (Figure 30). In 2018, a clear difference between the forest types was visible, as semi-natural forests translocated 56.4 ± 7.6, urban greening 78.5 ± 6.7, and succession forests 48.0 ± 10.4 kg $a^{-1}$ 100 $m^{-2}$. In 2019, in contrast, $CO_2$ translocation more level between the forest types, as semi-natural forests translocated 48.4 ± 5.5, urban greening 48.1 ± 6.9, and succession forests 35.9 ± 5.6 kg $CO_2$ $a^{-1}$ 100 $m^{-2}$.

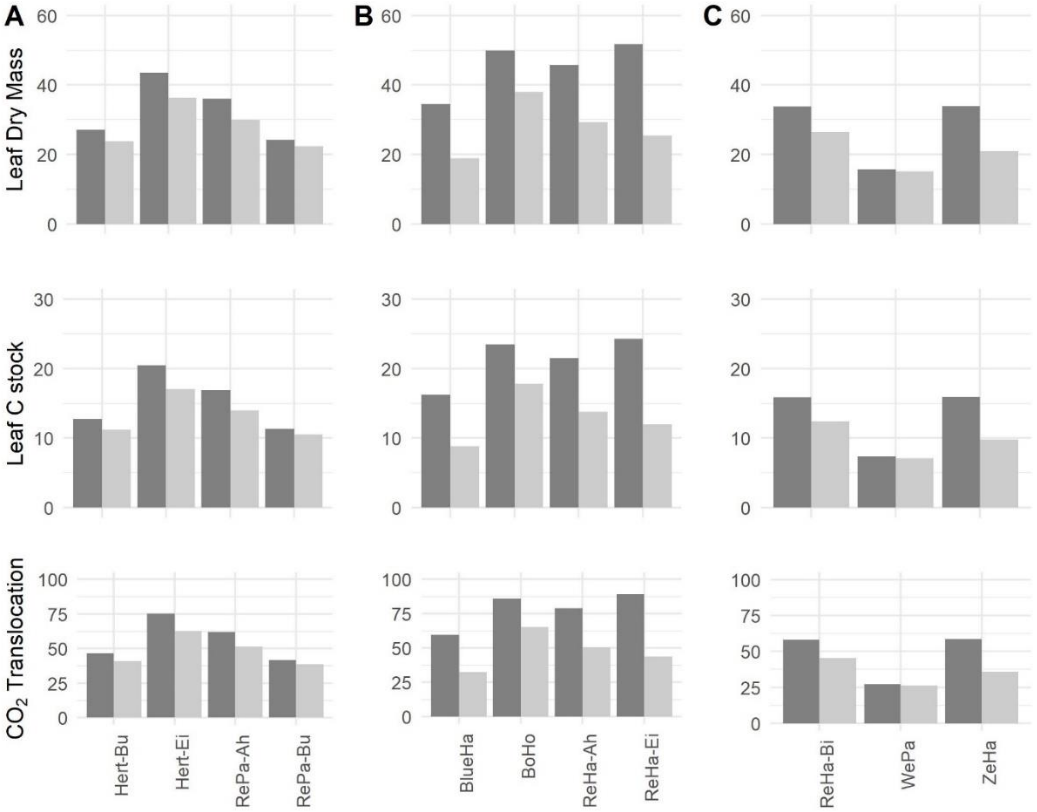


*Figure 30: Leaf dry mass [kg $a^{-1}$ 100 $m^{-2}$], leaf C stock [kg $a^{-1}$ 100 $m^{-2}$], and annual $CO_2$ translocation by litterfall [kg $a^{-1}$ 100 $m^{-2}$] at the sites. Dark grey bars show the values for 2018, light grey bars for 2019. A = semi-natural forests, B = urban greening forests, C = succession forests.*

The decrease in $CO_2$ translocation was smallest in the semi-natural forests, in comparison to the other forest types. To test if the different intensity of annual changes in $CO_2$ translocation was also triggered by the soil hydraulic characteristics, a PCA and correlation analysis were performed (data not shown). Like the $CO_2$ assimilation by tree growth, weak correlations were found between the decrease in $CO_2$ translocation (e.g., decrease in LAI) and the soil carbon stock and the depth of the rooting zone. Drought effects on leaf development were stronger in forests with small soil carbon stocks, a limited rooting zone, and low plant-available water. Visa versa, forests with a large soil carbon stock, non-limited rooting zone, and large amount of plant available water were more resilient against the water shortage and only a small decrease in leaf development had occurred.

## 4.6 $CO_2$ release by soil respiration

$CO_2$ effluxes measured at the sites and the annual $CO_2$ output is plotted in Figure 31. Between the single sites, no significant differences were found in $CO_2$ release. The median $CO_2$ efflux varies in a narrow range between 0.5 µmol $m^{-2}$ $s^{-1}$ measured on BoHo and RePa-Ah and 1.0 µmol $m^{-2}$ $s^{-1}$ measured on Hert-Ei and WePa. Hence, the annual $CO_2$ output from soil respiration ranges between 69.4 and 138.8 kg $CO_2$ $a^{-1}$ 100 $m^{-2}$.

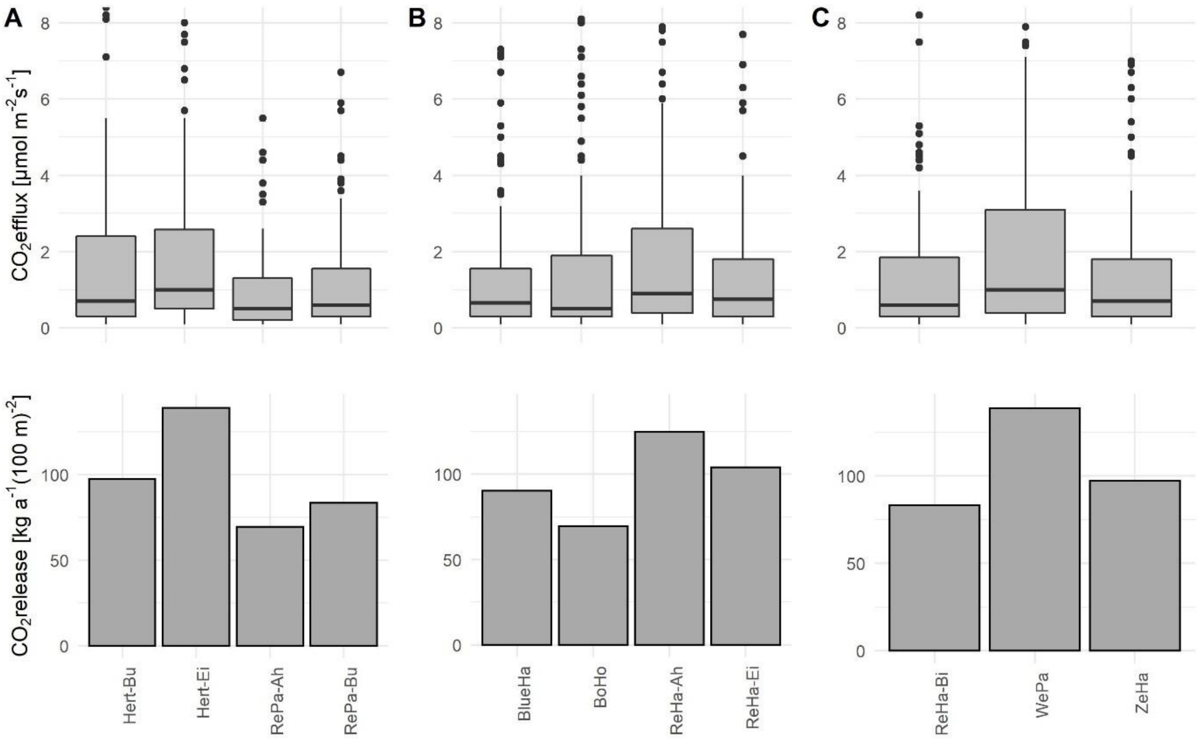


*Figure 31: $CO_2$ efflux [µmol $m^{-2}$ $s^{-1}$] and annual $CO_2$ release [kg $a^{-1}$ 100 $m^{-2}$] at the sites by soil respiration measured between July 2018 and October 2019. Annual $CO_2$ release was converted using the site-specific median of $CO_2$ respiration. A = semi-natural forests, B = urban greening forests, C = succession forests.*

Again, a PCA and correlation analysis was performed to evaluate the relationship between the site-specific median of the soil respiration rate and the soil properties (Figure 32). $CO_2$ efflux is highly positively correlated with $C_{mic}$ content, representing the amount of microbial biomass in the soil and with $C_{mic}$/SOC ratio, which is a proxy for the decomposability and accessibility of organic matter for the microorganisms (JÖRGENSEN 1995b). Consequently, the respiration rate is clearly controlled by the amount of microbial biomass and the availability of organic C in the urban forest soils. A $C_{mic}$/SOC ratio smaller than 1.2 % indicates inhibiting conditions for decomposition activity (JÖRGENSEN 1995b). Only at WePa the $C_{mic}$/SOC ratio is larger than 1.2 %. Hence, on all other sites the decomposition of organic matter is inhibited, which might be a consequence of the drought.

The sites Hert-Ei and ReHa-Ah were excluded from analysis because the correlation between $C_{mic}$/SOC and soil respiration is not valid for these sites. On these sites, soil respiration is relatively high despite of a low $C_{mic}$/SOC ratio.

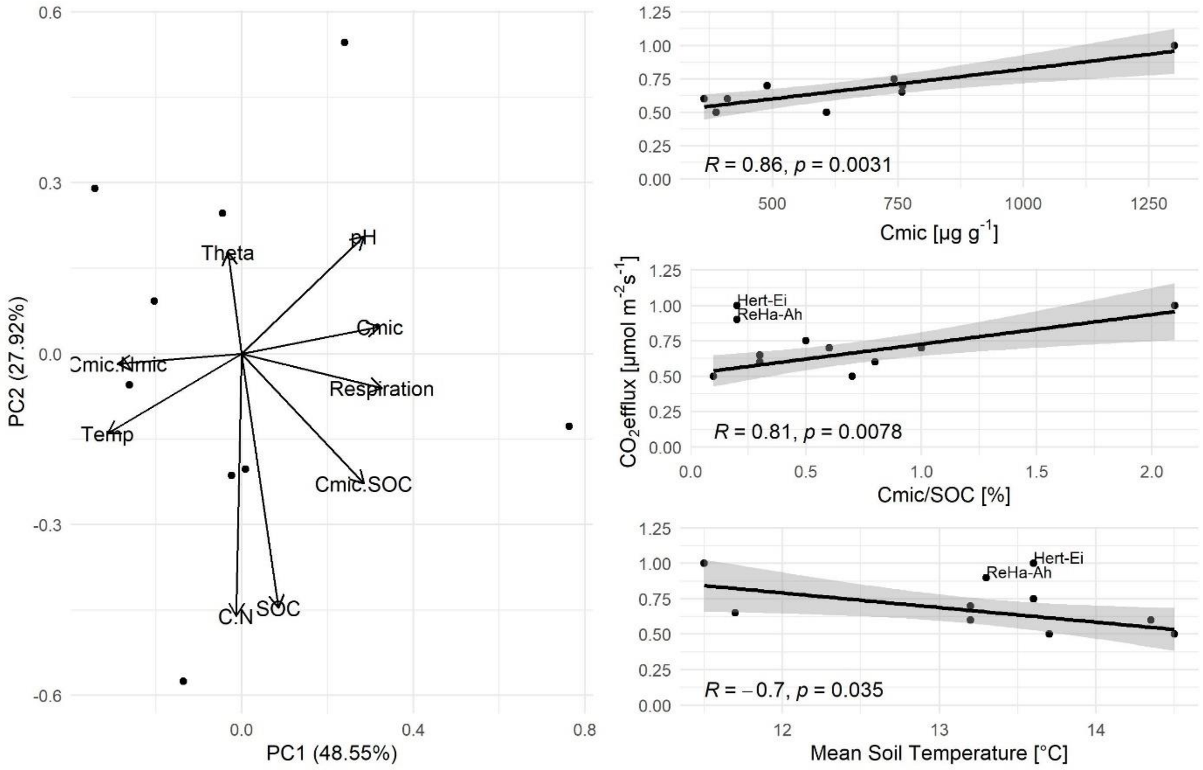


*Figure 32: PCA and correlation analysis to evaluate the relationship between site-specific median of the respiration rate [µmol $m^{-2}$ $s^{-1}$] and soil properties. Temp = mean soil temperature [°C] in 5 cm depth taken during respiration measurements, Theta = mean volumetric water content [$cm^3$ $cm^{-3}$] in 5 cm depth measured during respiration measurements. ReHa-Ah and Hert-Ei were excluded from analysis because of stress-induced patterns of soil respiration, which were not found for the other sites (compare chapter 5.5).*

Furthermore, $CO_2$ efflux is highly negatively correlated with mean soil temperature and positively correlated with pH (r = 0.63, p = 0.067) and $C_{mic}/N_{mic}$ ratio (R = -0.58, p = 0.1) which is an indicator for the vitality of the microorganisms (JÖRGENSEN 1995b). C/N ratio, SOC, and soil water content showed no statistical relationship with $CO_2$ efflux. Also, no correlation was found between the single respiration measurements and the simultaneously taken temperature and soil water content measurements at the locations.

Looking at the monthly $CO_2$ effluxes at the sites (see Figure 33), only ReHa-Ah and ReHa-Bi significantly differed for the period July and October 2019 in comparison to all other months. At all other sites, no significant differences were found. However, small changes in monthly $CO_2$ effluxes show, that the fluxes were largest in autumn and winter. By comparing the time series, two different patterns can be identified: Firstly, at Hert-Bu, Hert-Ei, BlueHa, ReHa-Ah, ReHa-Bi, and WePa the $CO_2$ effluxes were relatively large in summer and autumn 2018 but decreased rapidly in spring 2019, where they remained low for the remaining experimental period. Secondly, on RePa-Ah, RePa-Bu, BoHo, ReHa-Ei, and ZeHa $CO_2$ effluxes were already small in summer 2018, increased in autumn and winter and decreased in spring and summer 2019 to the same level as 2018. Here, the microbial community obviously shifted their decomposition and mineralization activity towards colder seasons.

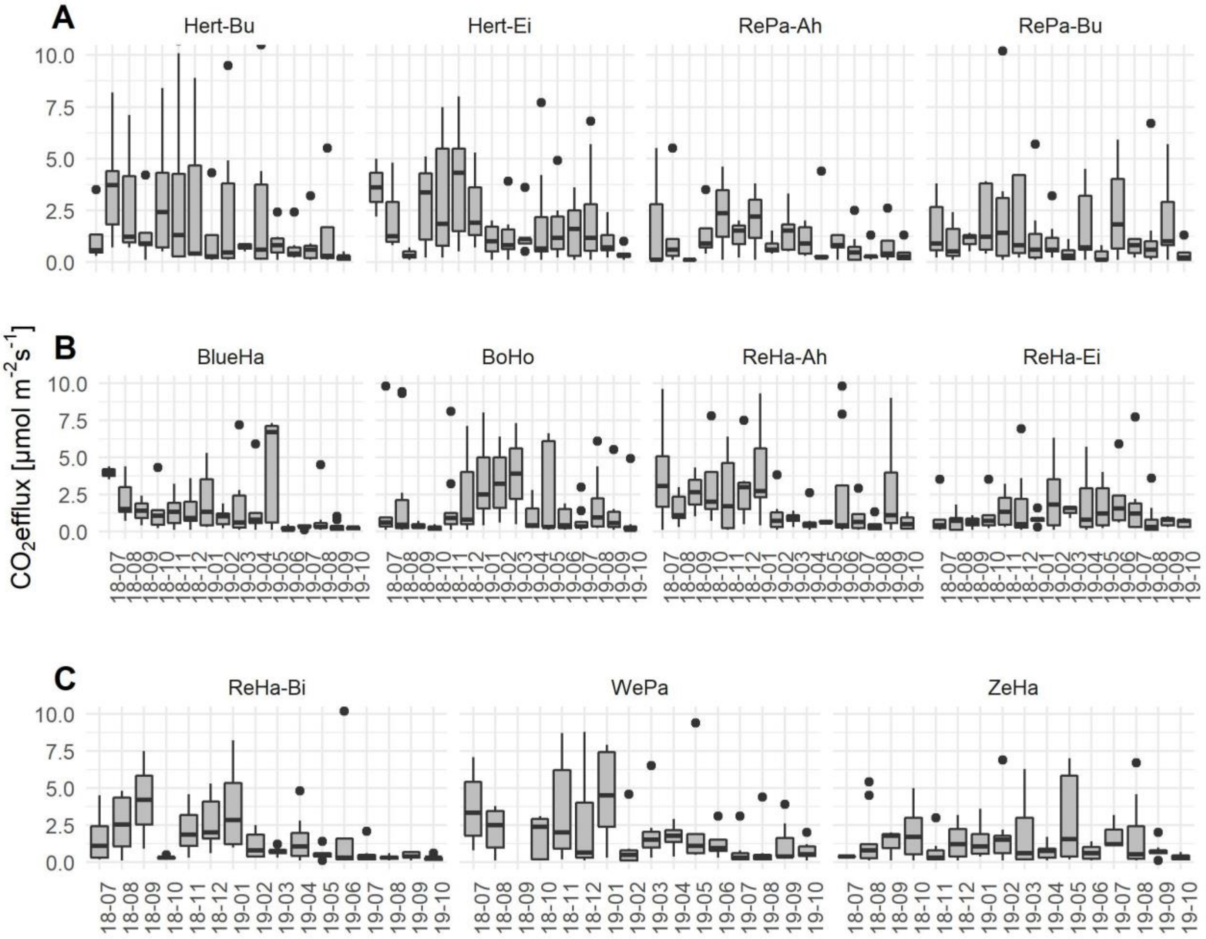


*Figure 33: Monthly $CO_2$ effluxes [µmol $m^{-2}$ $s^{-1}$] at the sites between July 2018 and October 2019. A = semi-natural forests, B = urban greening forests, C = succession forests.*

## 4.7 Forest carbon stocks

The allocation of the single carbon stocks at the sites are listed in Table 14. Largest total C stocks are found in the semi-natural forests (2.6 to 8.9 t 100 $m^{-2}$), while the total C stocks in the urban greening and succession forests are quite similar between 2 to 3 t 100 $m^{-2}$. Only the old birch forest ReHa-Bi has a total carbon stock of 6 t 100 $m^{-2}$, which is as high as the C stocks of the semi-natural forests. Averaged across all sites, the total carbon stock is composed by 60 % of carbon in the wood and 40 % of carbon in the soil, while the carbon stock in the leaves is less than 1 %. The proportional composition differs depending on forest age and forest type. In old forests with large forest traits (DBH, tree height), carbon pools in the wood were more than 60 %, while in younger forests with small structural parameters, carbon pools in the wood were smaller than 60 %. Different patterns are found for the succession forests. Here, more carbon is stored in the soil (57.6 %) than in the wood (42.1%). At WePa and ZeHa soil carbon stocks represent 57.6 and 78.1% of the whole forest carbon stock.

*Table 14: Allocation of carbon stocks at the sites shown for 2018.*

| urban forest types | site | Soil C stock | Wood C stock | Leaf C stock | Forest C stock |
|---|---|---|---|---|---|
| | | [kg (100 m)$^{-2}$] | [kg (100 m)$^{-2}$] | [kg (100 m)$^{-2}$] | [kg (100 m)$^{-2}$] |
| Semi-natural forests | Hert-Bu | 2,128.1 | 3,027.1 | 12.7 | 5,168.0 |
| | Hert-Ei | 754.8 | 1,789.2 | 20.5 | 2,564.5 |
| | RePa-Ah | 2,196.3 | 2,823.0 | 16.9 | 5,036.2 |
| | RePa-Bu | 1,789.2 | 7,991.3 | 11.3 | 8,944.2 |
| Urban greening forests | BlueHa | 851.2 | 1,270.1 | 16.2 | 2,137.5 |
| | BoHo | 948.6 | 1,376.5 | 23.5 | 2,348.5 |
| | ReHa-Ah | 707.8 | 2,249.6 | 21.5 | 2,978.9 |
| | ReHa-Ei | 895.7 | 1,400.2 | 24.3 | 2,320.2 |
| Succession forests | ReHa-Bi | 1,388.5 | 4,698.6 | 15.9 | 6,103.0 |
| | WePa | 1,318.1 | 963.5 | 7.4 | 2,289.0 |
| | ZeHa | 1,740.6 | 475.6 | 16.0 | 2,232.1 |

## 4.8 $CO_2$ sequestration

The $CO_2$ sequestration for both years at the single sites and for the different forest types are depict in Table 15 and Figure 34, considering tree growth, litterfall, and soil respiration. The semi-natural forests took up 152.8 ± 51.2 kg $CO_2$ $a^{-1}$ 100 $m^{-2}$ in 2018 and were able to increase the $CO_2$ sequestration in 2019 to 169.7 ± 83.8 kg $CO_2$ $a^{-1}$ 100 $m^{-2}$, even though the drought lasted on. In detail, RePa-Ah and RePa-Bu increased the $CO_2$ sequestration (+44.8 and +85.6 kg $CO_2$ $a^{-1}$ 100 $m^{-2}$), while the sequestration at the other semi-natural forests decreased (Hert-Bu -11.1, Hert-Ei -72.3 kg $CO_2$ $a^{-1}$ 100 $m^{-2}$) (Table 5). The urban greening forests clearly decreased the $CO_2$ sequestration in 2019 in comparison to 2018 from 177.5 ± 52.5 to 65.8 ± 45.7 kg $CO_2$ $a^{-1}$ 100 $m^{-2}$.

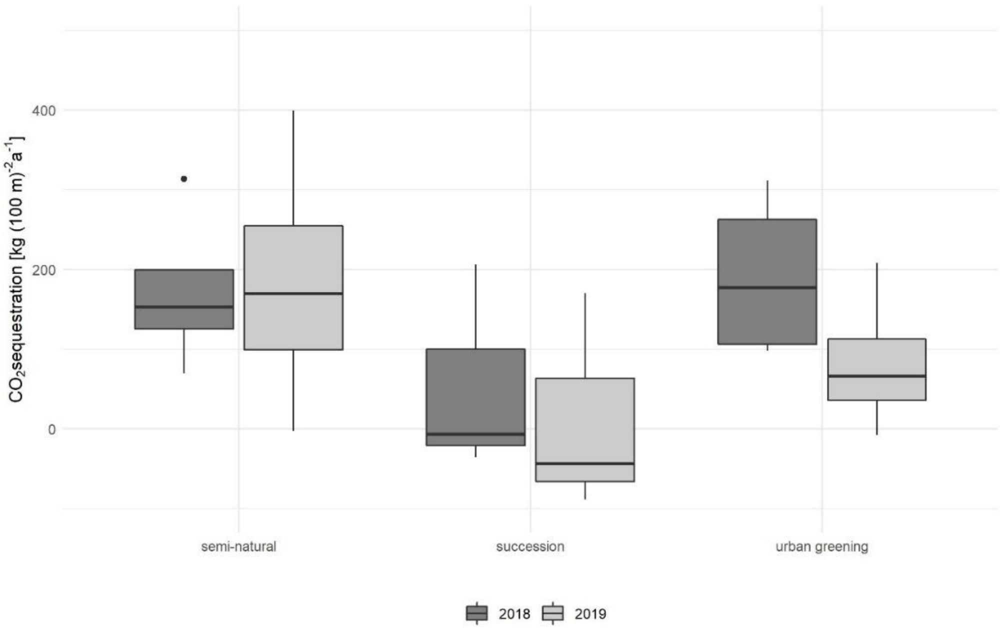


*Figure 34: Annual $CO_2$ sequestration of the forest types for the years 2018 and 2019.*

*Table 15: Annual $CO_2$ sequestration at the eleven monitoring sites.*

| **Urban forest types** | **site** | **year** | **$CO_2$ assimilation** | **$CO_2$ translocation** | **$CO_2$ release** | **$CO_2$ sequestration** | **Δ $CO_2$ sequestration** |
|---|---|---|---|---|---|---|---|
| | | | **[kg $a^{-1}$ 100 $m^{-2}$]** | | | | **[%]** |
| Semi-natural forests | Hert-Bu | 2018 | 194.7 | 46.7 | -97.1 | 144.2 | -7.7 |
| | | 2019 | 189.2 | 41.1 | -97.1 | 133.1 | |
| | Hert-Ei | 2018 | 133.4 | 75.2 | -138.8 | 69.8 | -103.6 |
| | | 2019 | 73.7 | 62.6 | -138.8 | -2.5 | |
| | RePa-Ah | 2018 | 168.8 | 62.1 | -69.4 | 161.5 | +21.7 |
| | | 2019 | 224.1 | 51.5 | -69.4 | 206.2 | |
| | RePa-Bu | 2018 | 355.4 | 41.6 | -83.3 | 313.7 | +21.4 |
| | | 2019 | 444.1 | 38.5 | -83.3 | 399.3 | |
| Urban greening forests | BlueHa | 2018 | 139.3 | 59.5 | -90.2 | 108.6 | -107.2 |
| | | 2019 | 49.9 | 32.6 | -90.2 | -7.8 | |
| | BoHo | 2018 | 294.8 | 86.1 | -69.4 | 311.6 | -33.1 |
| | | 2019 | 212.3 | 65.4 | -69.4 | 208.3 | |
| | ReHa-Ah | 2018 | 292.2 | 79.0 | -124.9 | 246.3 | -67.1 |
| | | 2019 | 155.4 | 50.5 | -124.9 | 81.0 | |
| | ReHa-Ei | 2018 | 113.0 | 89.2 | -104.1 | 98.1 | -48.4 |
| | | 2019 | 110.8 | 43.8 | -104.1 | 50.6 | |
| Succession forests | ReHa-Bi | 2018 | 230.9 | 58.3 | -83.3 | 205.9 | -17.3 |
| | | 2019 | 208.1 | 45.5 | -83.3 | 170.4 | |
| | WePa | 2018 | 76.0 | 27.1 | -138.8 | -35.6 | -59.8 |
| | | 2019 | 23.9 | 26.1 | -138.8 | -88.8 | |
| | ZeHa | 2018 | 32.0 | 58.6 | -97.1 | -6.5 | - 85.0 |
| | | 2019 | 17.8 | 36.1 | -97.1 | -43.3 | |

The strong decrease in $CO_2$ assimilation and $CO_2$ translocation, and the relatively large soil respiration on Hert-Ei and BlueHa led even to a negative $CO_2$ sequestration in 2019 on these sites. The $CO_2$ release was about 2.5 kg $a^{-1}$ 100 $m^{-2}$ on Hert-Ei and 7.8 kg $a^{-1}$ 100 $m^{-2}$ at BlueHa. The succession forests WePa and ZeHa were a source for $CO_2$ in both years, but the $CO_2$ release was larger in the second drought year than in the first year. The source role of those succession forests is caused by the very slow tree growth on WePa and ZeHa, which faced relatively high rates of soil respiration. The tree growth on ZeHa is even smaller than the $CO_2$ translocation by litterfall. But not all succession forests are sources for $CO_2$. ReHa-Bi is a strong sink for $CO_2$ with a $CO_2$ sequestration of 205.9 kg $a^{-1}$ 100 $m^{-2}$ in 2018 and 170.4 kg $a^{-1}$ 100 $m^{-2}$ in 2019.

# 5 Discussion

## 5.1 Urban forest carbon stocks

Largest soil C stocks were found at the semi-natural forests, followed by the succession and urban greening forests. Because of their age, carbon had been accumulated at the semi-natural forests RePa-Bu, RePa-Ah, and Hert-Bu, and the mineralization might be inhibited due to low pH values found in these forests (SCHEFFER & SCHACHTSCHABEL 2018). In comparison, Hert-Ei had a relatively low soil C stock for its age, which may indicate a fast mineralization of the soil organic material, which is caused by high pH values of about 5 in all horizons and a close C/N and $C_{mic}/N_{mic}$ ratio, which are indicators for a good quality of organic matter for (C/N) and a good vitality of microorganisms ($C_{mic}/N_{mic}$) (JOERGENSEN 1995b, SCHEFFER & SCHACHTSCHABEL 2018). Urban greening forests are younger and have not accumulated that much carbon, yet. At the succession forests, stone coal and coke compartments were found, which explains the extremely high SOC, especially at ReHa-Bi and the fact that at WePa and ZeHa more C is stored in the soil than in the wood.

The German National Forest Inventory quantifies the average carbon stock of German forests to 2.24 t 100 $m^{-2}$, including above- and belowground biomass, deadwood, and soil carbon (WELLBROCK et al. 2017). The carbon stocks of the urban greening and succession forests (both 2.3 t 100 $m^{-2}$) are within this range, while the semi-natural forests have larger carbon stocks (5.1 t 100 $m^{-2}$) caused by larger DBH and tree height than average and long developed and undisturbed soils with larger carbon content. On average, the allocation of carbon in German forests are 46 % (1.05 t 100 $m^{-2}$) in the living biomass and 53 % (1.18 t 100 $m^{-2}$) in the soil (WELLBROCK et al. 2017, GRÜNEBERG et al. 2019). The semi-natural and urban greening forests have a different allocation, where 60 % of carbon is stored in the living biomass and 40 % in the soil. In urban greening forests, the soil C stock is slightly smaller (0.87 t 100 $m^{-2}$) and the C in living biomass is slightly larger (1.39 t 100 $m^{-2}$) than average, while in the semi-natural forests both C stocks are larger (2.93 t 100 $m^{-2}$ in the living biomass and 1.97 t 100 $m^{-2}$ in soil). The succession forests on former industrial sites are not comparable, because this forest type is not considered by the National Forest Inventory. As shown in NORD-LARSEN et al. (2019), the carbon inventory of Denmark quantifies the carbon stock of managed beech-dominated forests to be 4 t 100 $m^{-2}$ on average, which shows that large carbon contents especially in semi-natural forests are possible. By aggregating all forest types, the urban forests in the Ruhr Metropolitan Region can be quantified to 2.56 t C 100 $m^{-2}$. Related to the estimated area of urban forests in the Ruhr Metropolitan Region (1,036 $km^2$) (SCHOLZ 2020), the total carbon stock is up to 26.5 Mio. t. Similar estimates were given for Berlin as a comparable metropolitan region, which quantifies the total tree carbon pool on 24 Mio. t or 2.7 t 100 $m^{-2}$ (RICHTER et al. 2020).

## 5.2 $CO_2$ assimilation in urban forests

The annual $CO_2$ assimilation by stand growth ranges between 17.8 at ZeHa and 444.1 kg 100 $m^{-2}$ at RePa-Bu with a median of 162.1 kg 100 $m^{-2}$. In detail, semi-natural forests assimilated in average 181.7 in 2018 and 206.7 kg 100 $m^{-2}$ in 2019, urban greening forests 215.8 in 2018 and 133.1 kg 100 $m^{-2}$ in 2019, and succession forests 76.0 in 2018 and 23.9 kg 100 $m^{-2}$ in 2019 in the living biomass. In comparison, the National Forest Inventory quantifies the annual $CO_2$ assimilation, depending on stand age, in a range of 34 and 162 kg 100 $m^{-2}$, with an average of 102 kg 100 $m^{-2}$ (WELLBROCK et al. 2017, RIEDEL et al. 2019). The investigated semi-natural and urban greening forest stands had noticeably higher $CO_2$ assimilation rates than the average reported. The differences in $CO_2$ assimilation can be explained by methodological and physiological reasons.

On the methodological side, firstly, data collection in the German National Forest Inventory is carried out periodically (1987, 2002, 2008, 2012, 2017) and the annual $CO_2$ assimilation is quantified by carbon stock changes in the specific period (RÖHLING et al. 2016). Variations in annual growth rates or a loss of trees due to logging or dying cannot be considered, but it might be more representative for long-term $CO_2$ sequestration than periodical approaches, like in the presented study. Secondly, the data collection for the National Inventory occurs also on a permanent location, but the sample of trees are defined by randomized angle-count sampling (WELLBROCK et al. 2017). Hence, it is not assured that in each year the same trees are measured, even though the sampling should consider all trees that were measured in the years before (RIEDEL et al. 2017). If single trees died or were logged in the period or are not traceable, not the same trees will be measured. Therefore, the sampling design of the German National Forest Inventory yield the carbon stock change of the forest stand, not of the single trees. In comparison, in the presented study all trees were measured and revisited on the specific sites, which ensured that annual growth rates for single trees can be calculated. Thirdly, for the latest

German national greenhouse gas monitoring in 2017, the approaches for calculating carbon stocks in the above- and belowground biomass changed in comparison to early studies (RIEDEL & KÄNDLER 2017). The new approach relies more on empirical data but provides a biomass function only for five different tree species (beech, oak, poplar, pine, and spruce). Generally, in urban forests the diversity of tree species is noticeably higher and tree species like *Betula pendula* or *Acer pseudoplatanus* are not considered separately in those functions. The early approach, described in PISTORIUS et al. (2006) and OEHMICHEN et al. (2011), considers more tree species, for example by using the different wood density factors and drying factors by KOLLMANN (1982), and therefore, appeared more suitable for the presented study. The new approach led to 1.3 % smaller carbon contents in the aboveground biomass, but in comparison to the early method no significant differences were found (RIEDEL & KÄNDLER 2017). It can be concluded that the early approach is still suitable for calculating carbon stocks and carbon stock change in living biomass. However, the less conservative estimation by using the early approach might led to larger $CO_2$ assimilation rates estimated for the urban forest ecosystems.

Besides methodological differences, forest stands in urban environments might have a faster growth than forests in rural areas, which are primarily considered by the National Forest Inventory. Forest growth depends on many environmental factors like solar radiation, $CO_2$ concentration in the atmosphere, temperature, length of the growing season, water availability, and the nutrient supply (MITSCHERLICH 1970). The average temperature in urban areas is at least 3 °C warmer compared to rural sites (OKE et al 2017), resulting in an extension of the growing season length (WHITE et al. 2002). In the years 2018 and 2019 solar radiation and temperature was higher, and the length of the growing season was 16 days longer than average, which might have amplified forest growth, while water availability became more and more a limiting factor during the experimental period. KOWALSKA et al. (2020) found for a floodplain forest in 2018 the highest total gross primary production of the investigated period (2015-2018). Warmer spring temperatures and sufficient soil water content in 2018 led to early leaf development which overcompensated the effects of the summer drought. This might be also valid for the presented study because the precipitation in 2017 was nearly equal to the long-term average and an early leaf development were observed at all sites.

Secondly, in comparison to rural environments the investigated urban forest soils are characterized by high nitrogen content. Overall forest types, the nitrogen content was about 69.5 kg 100 $m^{-2}$, where the N content was largest in the semi-natural forests (88.5 kg (100 m)-2), followed by the urban greening forests (57.5 kg 100 $m^{-2}$), and the succession forests (60.3 kg 100 $m^{-2}$). In comparison, the forests soils in Germany have an average nitrogen content of about 63 kg 100 $m^{-2}$, measured for the whole soil profile down to 90 cm (FLECK et al. 2019). The fact that urban greening and succession forests have lower N contents might be caused by their thin soil profiles, while the semi-natural forests clearly exceed the average value. Additionally, the average C/N ratio in the semi-natural forests is 19.8 in the A-horizon and 17.9 in top layer and in urban greening forests the average C/N ratio in A-horizon is 16.1. In comparison, the average C/N ratio in forest soils in Germany is 25.2 in top layer and 20.6 in the upper 5 cm of the mineral soil. Therefore, the C/N ratio of the urban forest soils are lower than 25, which is a threshold for enhanced nitrate leaching via seepage water (GUNDERSEN et al. 1998), underlining the high N content in urban ecosystems. Sources of nitrogen in urban areas are mainly fossil fuel combustion by traffic, but also the deposition of urea by dogs might be an important source of N in the urban forests in the Ruhr Metropolitan Region (BOBBINK et al. 2010). Different studies show that urban forests are highly exposed to anthropogenic N deposition in comparison to forests in the rural environments (BAI et al. 2015, LOVETT et al. 2000, BETTEZ & GROFFMAN 2013, WITTIG 1991). The N deposition in deciduous forests is quantified to 22-24 kg ha$^{-1}$ a$^{-1}$ in the experimental area (SCHAAP et al. 2018). Many studies showed positive effects of high nitrogen content on forest growth and $CO_2$ sequestration (HYVÖNEN et al. 2007, DE VRIES et al. 2006). REHFUESS et al. (1999) showed that a combination of $CO_2$ increase and elevated N depositions led to 15-20 % increase in forest net primary production, where N deposition is claimed to be more important than increased $CO_2$. Observed increases in annual height growth of Scots pine, Norway spruce, and European beech compared to 40 years ago were mainly caused by elevated N deposition (KAHLE et al. 2008). LAUBHANN et al. (2009) found that an increase of 1 kg N ha$^{-1}$ a$^{-1}$ leads to a basal area increment between 1.2 and 1.49 % depending on species.

In summary, the larger $CO_2$ assimilation of the urban forests in comparison to the findings of the German National Forest Inventory might be, beside of methodological differences, caused by higher temperatures, higher solar radiation, and a longer growing season induced by the extreme years 2018 and 2019, combined with the specification of urban environments (additionally higher temperatures, additionally extended growing season, and higher soil N content in comparison to rural forests considered in the National Forest Inventory). This hypothesis is supported by the fact, that extremely high annual growth rates were detected in the semi-natural forest RePa-Bu, where the N content is largest, and the water availability is not the limiting factor for forest growth.

## 5.3 Effects of drought on $CO_2$ sequestration and $CO_2$ assimilation

As shown, the ongoing drought decreased forest growth, leaf expansion, and soil respiration in 2019 in comparison to 2018, leading to a decrease in net $CO_2$ sequestration at the urban greening forests by 62.9 % and at the succession forests by 85.0 %, while the net $CO_2$ sequestration increased at the semi-natural forests by 9.9 %. This increase is caused by the increase in net $CO_2$ sequestration at RePa-Ah (21.7 %) and at RePa-Bu (21.4 %), while at

Hert-Bu (-7.7 %), and Hert-Ei (-103.6 %) the net $CO_2$ sequestration decreased. The sites BlueHa and Hert-Ei turned from a sink of $CO_2$ in 2018 to a source in 2019. These findings are in line with GRAF et al. (2020), who analyzed data from 56 eddy covariance sites from 2018 across Europe and found a decrease of net $CO_2$ sequestration by 17.8 % over all different land use types (forests, grasslands, crops, and peatlands). Unfortunately, the study of GRAF et al. (2020) did not include urban forest ecosystems. Therefore, the larger decrease in net $CO_2$ sequestration calculated for the urban forests (-48.4 % in average) indicate, that they are more vulnerable to drought events. First assessments of the impact of the extreme 2018 summer drought on central European forests showed that an unprecedented tree mortality occurred in 2018 with strong drought-legacy effects in 2019 (SCHULDT et al. 2020), which are in line with the findings at the monitoring sites. BMEL (2019) estimates the total loss of wood volume to 105 million $m^3$ for the drought years 2018 and 2019. CIAIS et al. (2005) showed that during the heat wave 2003 the gross primary productivity of ecosystems in Europe decreased by more than 30 %, resulting in a net $CO_2$ source of 0.5 Pg C $a^{-1}$ due to precipitation deficit and extreme summer heat. In consequence, the effect of four years of net ecosystem carbon sequestration were reversed. The drastically decrease of $CO_2$ sequestration in the second drought year is typical for forest ecosystems, where tree mortality does not occur instantly at the beginning of the drought but is delayed and occur in the following years after the event (KANNENBERG et al. 2018, ANDEREGG et al. 2015).

The changes of $CO_2$ sequestration differ between urban forest types. In 2018, the semi-natural forests sequestered 152.8 kg $CO_2$ 100 $m^{-2}$, the urban greening forests 177.5 kg $CO_2$ 100 $m^{-2}$, and the succession forests were a source for $CO_2$, where 6.5 kg $CO_2$ 100 $m^{-2}$ were released. No significant differences in annual $CO_2$ sequestration between semi-natural forests and urban greening forests was found in 2018. The slightly higher $CO_2$ sequestration in urban greening forests compared to semi-natural forests might be caused by a higher tree density in the urban greening forests. However, this pattern changed in 2019, where the urban greening forests sequestered significant less $CO_2$ compared to the semi-natural forests. The succession forests WePa and ZeHa were $CO_2$ sources in both years. This is caused by small $CO_2$ assimilation and translocation due to unfavorable growth conditions such as high stone content, low bulk density, and low soil depth, which faced relatively high $CO_2$ effluxes caused by high soil pH values.

The results from PCA and correlation analysis show that the $CO_2$ assimilation in urban forest ecosystems in the two drought years strongly depended on water availability in the rooting zone (Figure 35). Therefore, the semi-natural forest sites were less affected by the two years drought because of deeper and carbon rich soils, which were able to hold much more water, and therefore, increase plant available water for tree growth. Furthermore, they were not limited in root growth and were able to reach water resources in deeper zones. In contrast, the forests on former anthropogenic sites were limited in rooting due to sealed soil layers. They explored a smaller soil volume with a lower carbon content and a higher proportion of coarse material, which negatively affected plant available water. Because of that, tree growth on former anthropogenic sites depended more on regular precipitation, which is absent during drought events. In conclusion, the semi-natural forests were more resilient to drought conditions and can maintain $CO_2$ sequestration, in terms of ES, even over the second drought year, while urban greening and succession forests clearly decreased their $CO_2$ sequestration function.

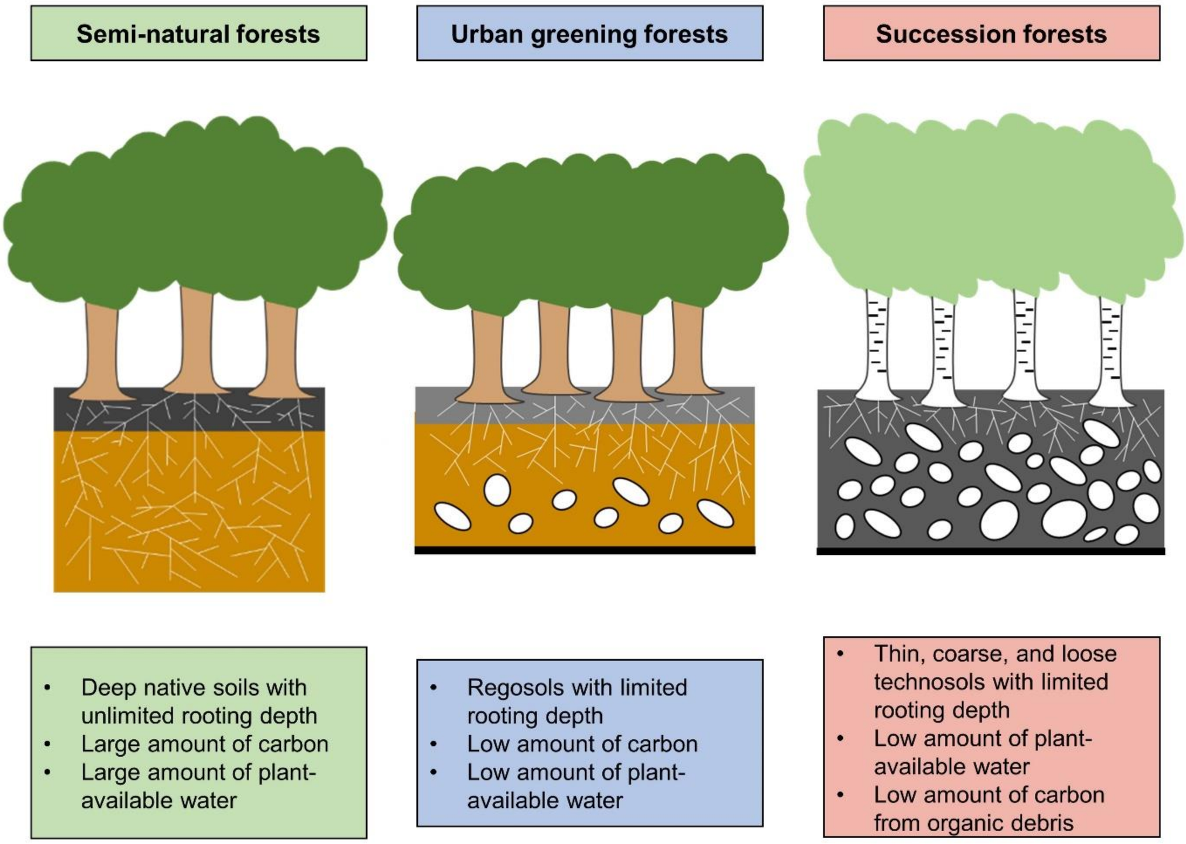


*Figure 35: Schematic depiction of the soils found in the different forest types and characteristics which explains the difference in $CO_2$ assimilation between 2018 and 2019.*

Additionally, the results show that not all semi-natural forest types were equal in their resilience against droughts. Semi-natural forests on mesophilic locations, like RePa-Bu, RePa-Ah, and Hert-Bu, which can be described as *Galio-Fagetum* and *Maianthemum-Fagetum*, were more resilient against droughts compared to semi-natural forests on more extreme locations, like Hert-Ei (*Stellario-Carpinetum*), which was an alternately wet forest type. The dense water logging-horizon found here restricted deep rooting and capillary upward flow from the groundwater. Like the forests on former anthropogenic sites with sealed horizons, water supply depended here on regular precipitation. Furthermore, Hert-Ei had a soil C stock more comparable to the urban greening forests, which might be caused by fast mineralization, supported by relatively high pH-values and close C/N and $C_{mic}/N_{mic}$ ratio in all horizons. In conclusion, Hert-Ei was the only semi-natural forest which assimilated less $CO_2$ per tree in 2019 than 2018 and this site even turned from a carbon sink in 2018 to a carbon source in 2019. BEHRENS et al. (2009) already predicted negative effects from climate change on *Stellario-Carpinetum* forests, while the effects on beech forest communities is predicted as indifferent. Those findings are in good agreement with the results from the presented experiment.

The repeated drought led to a loss of trees at the urban greening sites BlueHa and ReHa-Ei due to sooty bark diseases caused by the fungus *Cryptostroma corticale*, which infested especially young sycamore trees (*Acer pseudoplatanus*) under water stress in warm environments (DICKENSON & WHEELER 1981). Consequently, BlueHa turned from a carbon sink in 2018 to a source in 2019, but ReHa-Ei showed no changes between both years in $CO_2$ assimilation. The reason for that might be that the trees at ReHa-Ei already died in 2018. Due to the death of a large portion of trees in 2018 at ReHa-Ei, this stand might had showed the highest soil water contents with a mean of 0.2 $cm^3$ $cm^{-3}$ at 5 cm depth during the vegetation periods. The reason for such high soil water contents might be the reduced total water consumption of the forest stand by transpiration due to the loss of single trees, whereby the remaining trees still shaded the soil to such extent, that the reduced transpiration was not compensated by higher evaporation (YOUNG & MITCHELL 1994). This water compensation due to a slowed forest growth by a loss of single trees improved the growth conditions for the remaining trees and might be an initial mechanism for natural rejuvenation and recovering, which is already described for crops (HECHT et al. 2016). Further research is needed to examine this reaction of urban forest ecosystems on drought events.

In conclusion, the data presented showed that the studied temperate urban forests are affected by two years drought and can even turn from a carbon sink into a source. However, the reaction on droughts also differs between tree species. RÖTZER et al. (2017) showed experimentally, that DBH increment in a beech forest under drought condition was not significantly different from the one without drought, underlining the resilience of $CO_2$ sequestration of beech forests even under water stress (PRETZSCH 2005, PRETZSCH et al. 2012). As anisohydric species, beeches close their stomata slowly during droughts, ensuring an ongoing gas exchange and photosynthesis, while the species accept xylem damages by drying out (RÖTZER et al. 2017, PRETZSCH 2019). This might explain why especially the beech forests (RePa-Bu and Hert-Bu) kept up high rates of annual $CO_2$ assimilation in the second drought year.

## 5.4 $CO_2$ translocation

In general, the observed LMA values are in good agreement with those reported in different studies (e.g., FALSTER et al. 2015, FORRESTER et al. 2017, LEUSCHNER et al. 2006). The LMA values for shaded leaves investigated by HAGEMEIER (2002) for Lower Saxony and Saxony-Anhalt in Germany are very close to the LMA values of the investigated urban forests (Birch: equal, Beech: +0.2 mg $cm^{-2}$, hornbeam: -0.4 mg $cm^{-2}$, lime: +0.1 mg $cm^{-2}$, oak: +0.8 mg $cm^{-2}$). Therefore, the values used in the presented study to calculate the leaf mass seem to be reliable. LEUSCHNER et al. (2006) examine the leaf mass in different European beech stands, which was between 29 and 39 kg 100 $m^{-2}$. The leaf mass of the urban beech stands in this study was between 22.3 and 27.1 kg 100 $m^{-2}$, which is smaller than the reported, whereby the beech stands investigated by LEUSCHNER et al. (2006) had a higher LAI (5.6 - 9.5) compared to the urban forest stands (LAI 4.5 - 5.5), which can explain the gap between both findings. Overall, the method used for modelling leaf mass seems to provide reliable results, even for small experimental plots (100 $m^2$). Additionally, this method is less time-consuming and cost intensive compared to the use of litter traps, which have their own bias (FINOTTI et al. 2006) and are not easy to install and maintain in urban public forests.

The drought affected leaf development at the sites. $CO_2$ translocation decreased in 2019 in comparison to 2018 and, like $CO_2$ assimilation, differences in $CO_2$ translocation between the years can be explained by soil carbon stock and depth of the rooting zone. Both characteristics, $CO_2$ assimilation and $CO_2$ translocation, are connected, because less leaves in 2019 might led to a smaller photosynthesis rate. Strongest decrease in $CO_2$ translocation were found at BlueHa and ReHa-Ei, where many trees died during the experimental period, as described above.

The fast leaf development at all sites in the beginning of 2018 was triggered by high spring temperatures, high solar radiation, and water availability from winter 2017/2018. The massive production of leaves, the corresponding transpiration, and the lack of precipitation led to water deficiency. To compensate the water deficiency, the trees started to drop their leaves quite early in July. The forest stands adapted their leaf development to the ongoing drought and built up their canopy more slowly in 2019. The maximum number of leaves was reached later in July, while in 2018 all leaves were already developed in June. Furthermore, the total leaf amount was smaller in 2019 than in 2018 on all sites. The reduction is linked with soil properties influencing the amount of plant-available water, indicating, that the decrease in leaf amount is a consequence of the water deficiency. The reason for the reduced leaf amount in 2019 might be that the bud formation felt into the drought period 2018, which impacted the number of leaves formed in the bud and as well as the formation of new stem segments and shoot elongation. In consequence, total leaf number was reduced in the year following the drought as also observed by BRÉDA et al. (2006). Furthermore, studies showed that it can take up to two to three years until the LAI recovers to a pre-drought level (ABER et al. 2002, BATTAGLIA et al. 1998, LE DANTEC et al. 2000). Hence, further research is needed to observe how LAI will change in the years after two growing periods under drought conditions.

## 5.5 Soil respiration

Overall, the mean annual $CO_2$ effluxes were quite small and ranging from 0.5 and 1 µmol $CO_2$ $m^{-2}$ $s^{-1}$, leading to an annual $CO_2$ release between 69.4 and 138.8 kg 100 $m^{-2}$. MEYER et al. (2018) estimated the annual heterotrophic $CO_2$ release for deciduous forests in the Lower Rhine-Valley at Cologne and in the Eifel between 177.7 and 311.2 kg $CO_2$ 100 $m^{-2}$, depending on the $Q_{10}$-value for 30 % water holding capacity. At all sites, the volumetric water content was lower than 30 %, which might have inhibited soil respiration, leading to lower annual $CO_2$ releases in comparison to MEYER et al. (2018).

Furthermore, because of the extremely low soil water contents and high soil temperatures no statistical relationship between $CO_2$ effluxes, soil temperature, and soil water content, postulated by different studies (REY et al. 2002, RAICH & SCHLESINGER 1992), were found for any site. Because of the extreme climatic situation, measurements only took place under hot and dry or, in winter, under cold and wet conditions. But hot and wet conditions, which typically lead to high respiration rates, are missing in the dataset. MANZONI et al. (2012) showed that soil respiration and temperature are decoupled during droughts, because drought stress reduced the diffusion of soluble C substrate as driver for microbial activity (WANG et al. 2014). This might also be valid for the presented study, even though correlations were found regarding the means over the experimental period.

A threshold for inhibiting effects for microbial activity in the soil is a $C_{mic}$/SOC ratio < 1.2 % (JÖRGENSEN et al. 1995b). Only WePa exceeded this threshold. Therefore, the data indicates that the drought situation diminishes $CO_2$ effluxes on nearly all sites due to a lack of soil water, which hinders access to organic matter, reduces the mobility of microorganisms, and shifts the microbial composition from bacterial- to fungal-dominated communities, leading to more stable C and a smaller $CO_2$ efflux by soil respiration (WANG et al. 2014, JÖRGENSEN 1995b). In addition, a negative correlation was found between $CO_2$ efflux and mean soil temperature. Classically, high soil temperatures amplify $CO_2$ efflux (LUO & ZHOU 2006), but when the soil temperature stays high over longer periods, like during the experimental period, the death rate of the microbes increases due to thermal denaturation, which will decrease the $CO_2$ efflux (JÖRGENSEN et al. 1990). This interaction appears to apply to the monitoring sites because $C_{mic}$ content is negatively correlated with mean soil temperature ($R = -0.79$, $p = 0.0042$), underlining that high soil temperatures decrease the amount of microbial biomass, which is a second effect occurring during the drought influencing $CO_2$ efflux.

The correlations between $CO_2$ efflux, temperature and $C_{mic}$/SOC content were not valid for Hert-Ei and ReHa-Ah, where stress-induced respiration activity can be clearly identified. Here, soil respiration is relatively high despite of a low $C_{mic}$/SOC ratio. This indicates an inefficient usage of organic matter by the microbes: more organic matter must be respired to produce the same amount of microbial biomass (JÖRGENSEN 1995b). This happens under stress conditions, which might be, in case of the presented study, the drought situation. By regarding the time series on both sites, the $CO_2$ effluxes were quite high in 2018 but rapidly decreased in 2019, indicating that the microbial community might be damaged by thermal denaturation and could not recover until the end of the experimental period due to the ongoing drought. These stress reactions led to a temporary increase in $CO_2$ efflux, but later to a massive decrease, which, in long-term, diminishes $CO_2$ effluxes, as well. The same temporal patterns can also be seen for Hert-Bu, BlueHa, ReHa-Bi and WePa. The other sites show a different pattern in the time-series, where microbial communities obviously shift their decomposition and mineralization activity into the cold season. In winter, the soil was rewetted in the upper soil layers, and organic material got available, leading to an increase of $CO_2$ efflux as found by GÖRANSSON et al. (2013), despite of low soil temperatures that typically inhibit microbial activity. In the warm season the $CO_2$ effluxes decreases again. The microbial communities on these sites might be better adapted to droughts than the communities on the other sites. Here, further research is needed to investigate how the microbial community composition differed between different urban forest soils and how they react on stress situation. Microorganisms react on stress conditions with an inefficient decomposition and mineralization of organic matter, where more respiration activity is needed to build up the same amount of microbial biomass compared to non-stress conditions. JÖRGENSEN (1995b) named different stress parameters, such as temperature (JÖRGENSEN et al. 1990), high contents of heavy metals (CHANDER & BROOKES 1991), or biocides (HARDEN et al. 1993). Biocides and heavy metals can be excluded for the sites because biocides are not added to forest soils and at both sites the pH is between 5 and 7, where most heavy metals are not soluble. Therefore, the stress reaction can be explained by high temperatures and low soil water contents.

In summary, it is assumed that the drought reduced the $CO_2$ effluxes in both ways: firstly, dry soil conditions limited the access of the microbes to organic matter and, secondly, high soil temperatures over longer periods decreased the amount of microbial biomass. On the other hand, the drought might have led to stress-induced effects on ReHa-Ah and Hert-Ei, leading to relatively large $CO_2$ efflux despite of low $C_{mic}$/SOC ratio and relatively high soil temperatures. Two types of seasonal patterns in $CO_2$ effluxes indicate, that the microbial communities differ in their strategies coping with stress situation.

Mean $CO_2$ effluxes were also positive correlated with pH and negative with $C_{mic}/N_{mic}$ ratio. Low pH values inhibit decomposition and mineralization activity, and wide $C_{mic}/N_{mic}$ ratio indicate poor quality of SOC for microbial biomass, resulting both in smaller $CO_2$ effluxes (JÖRGENSEN 1995b, LUO & ZHOU 2006). However, these correlations are weak in comparison to the consequences of the drought described above, showing secondary effects explaining the different patterns of $CO_2$ effluxes between the sites.

# 6 Conclusion

The aim of the experiment was to examine the differences in carbon stocks and $CO_2$ sequestration between the different forest types and to analyze factors which influence the $CO_2$ sequestration. Because the experiment took place during a drought period, the resilience and vulnerability of the forest types to droughts were also analyzed.

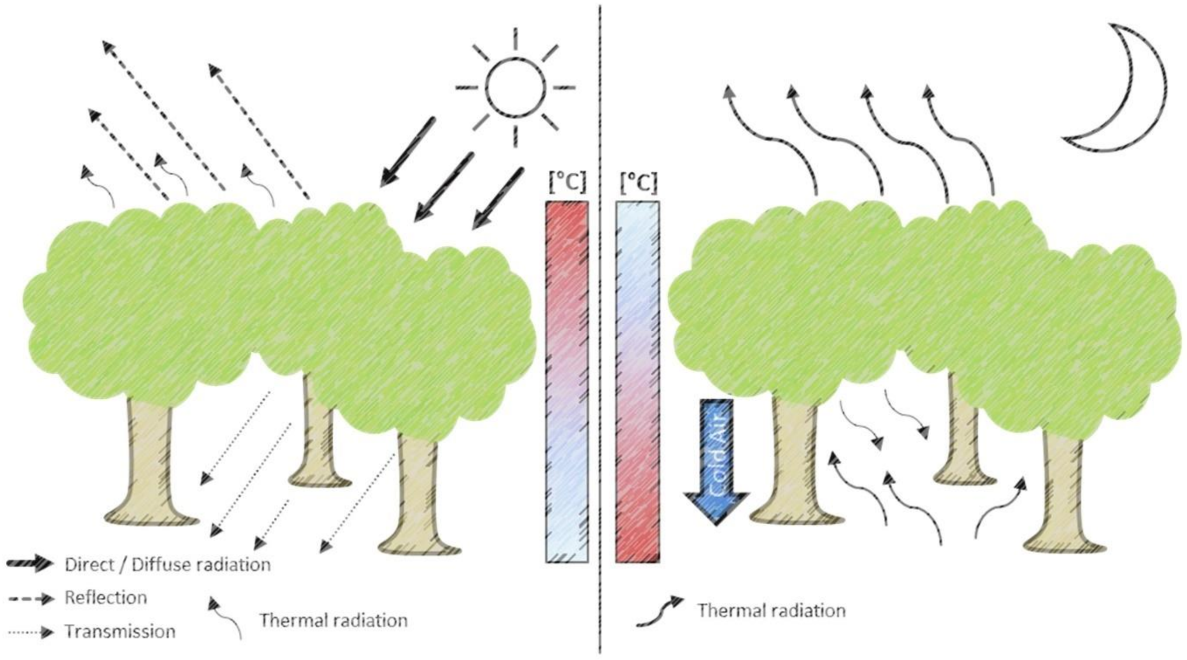
The amount of carbon stored in the urban forests depended on the sizes of the tree traits and the stand age. Here, the semi-natural forests had an advantage, where the largest carbon stocks were found, while the urban greening and succession forests had smaller carbon stocks. However, succession forests of higher ages can also have large carbon stocks which can be seen at ReHa-Bi. Vice versa, not all semi-natural forests have large carbon stocks: At Hert-Ei the forest carbon stock was as large as those found for the urban greening forests.

The $CO_2$ sequestration during the experimental period was clearly influenced by the drought. In the first year the urban greening forests showed a slightly larger $CO_2$ sequestration compared to the semi-natural forests but in the second year the $CO_2$ sequestration drastically decreased at the urban greening forests, while the semi-natural forests sequestered slightly more $CO_2$ compared to the first year. In conclusion, both forest types were able to sequester similar amounts of $CO_2$ per year. But the semi-natural forests were more resilient against droughts, while the urban greening forests are highly vulnerable against periods with water shortage. The reason for that was a smaller water supply due to a limited rooting depth, small amounts of soil organic carbon, and small amounts of plant-available water. The succession forests were $CO_2$ sources in both years resulting from a relatively low $CO_2$ assimilation and translocation and a relatively high $CO_2$ release by soil respiration due to high pH values. In the second year, $CO_2$ sequestration also decreased at the succession forests, showing that these forests are also vulnerable against periods with water shortage.

# 7 Implications for urban forestry and planning

In urban areas the growth conditions for urban forest stands differ widely, leading to different amounts of annual $CO_2$ sequestration, depending on the individual interaction between forest growth, litterfall, and soil respiration. Furthermore, these interactions will be even more impacted by climate change. Longer and more intensive heat waves and droughts globally decrease forest growth and soil respiration, but the different urban forest types react differently. While the old-grown semi-natural forests growing on soils characterized by high soil carbon content, plant-available water, and a deep rooting zone are able to buffer the elevated temperatures and low precipitation and keep up the $CO_2$ sequestration on the same level as in the first drought year, the urban greening and succession forests growing on soils with low carbon content, lower plant-available water, and limited rooting zone drastically decreased their $CO_2$ fixation potential, where some forests even turned from a $CO_2$ sink in the first year to a source in the second. Therefore, forest maintenance should focus on conserving semi-natural forests to sustain $CO_2$ sequestration as an ES even under drought conditions. Urban planners should ensure good soil condition for establishing new urban forest stands, whereby especially a deep rooting zone and carbon rich soils help to tackle drought periods, which are expected to increase under future climate.

# Part III – Microclimatic cooling of urban forests



# 1 Theoretical background

As a consequence of global warming, the occurrence of heat waves already increased and will continuously increase in duration, intensity, and frequency in the next decades (MEEHL & TEBALDI 2004, BARRIOPEDRO et al. 2011, HABEEB et al. 2015, SUAREZ-GUTIERREZ et al. 2020). Urban dwellers are more prone to heat waves because the Urban Heat Island (UHI) effect leads even to higher air temperatures in cities in comparison to the rural land (ORTIZ et al. 2018, ROGERS et al. 2019, HE et al. 2020, RICHARD et al. 2021). Increasing UHI intensity and duration causes serious health problems, increases mortality (BALDWIN et al. 2019, GUO et al. 2018, ROBINE et al. 2008, VANDENTORREN et al. 2004) and causes a degradation in air quality (CHURKINA et al. 2017, KALISA et al. 2018).

A prominent and intensively discussed measure to cope with thermal stress in cities is to implement green and blue infrastructure such as water bodies, trees, lawns, and parks in highly sealed urban areas, to initiate natural cooling effects (BOWLER et al. 2010, IOJĂ et al. 2021, ZÖLCH et al. 2016, SHASHUA-BAR et al. 2009). Due to the process of evapotranspiration, incoming solar radiation is more invested in latent heat than in sensible heat, leading to lower air temperatures in comparison to highly sealed areas. Furthermore, vegetation has a higher albedo in comparison to grey infrastructure (e.g., buildings, roads, pavements) and lower energy loads are saved over the day and emitted at night (TAHA 1997, TAHA et al. 1988). Especially urban trees are very effective in cooling urban areas because urban trees cool their environment additionally to transpiration by shading, which can be more effective than transpiration, especially under water shortage (RAHMAN et al. 2020a, HARDIN & JENSEN 2007, KORNASKA et al. 2013, OKE 1989).

Recent research on cooling effects of urban forests has been notably focused on single or small groups of trees in highly sealed areas (e.g., PACE et al. 2021, RAHMAN et al. 2020b, HELLETSGRUBER et al. 2020, RÖTZER et al. 2019, LINDÉN et al. 2016, GILLNER et al. 2015), while research on larger forest stands located in the urban matrix has been only sparsely explored. If urban forest stands are studied, research is often based on remote sensing techniques, measuring the surface temperatures of the tree canopies instead of the air temperature itself (GAGO et al. 2020, REN et al. 2018, DU et al. 2017, ALAVIPANAH et al. 2015). The land surface temperature (LSA) is indeed easier to obtain for larger study areas than field measurements. However, the LSA is not a good proxy for the cooling function because of i) surface temperature is not equal to air temperature, which is a direct indicator for human thermal comfort, and ii) the surface temperatures of the canopies do not represent the thermal conditions on the level of urban inhabitants. Even though, micrometeorological characteristics of forests in rural environments are well explored (e.g., HOLST et al. 2004, VAN EIMERN 1984, RENAUD & REBETEZ 2009, MORECROFT 1998, VANWALLEGHEM & MEENTEMEYER 2009, BAKER et al. 2016), little is known about the interaction between the urban climate and the urban forest climate.

## 2 Aim and design of the experiment

To address these knowledge gaps, air temperature and humidity measurements were performed between April 2018 and May 2020 at the eleven monitoring sites already shown in the introduction (chapter 2.4).

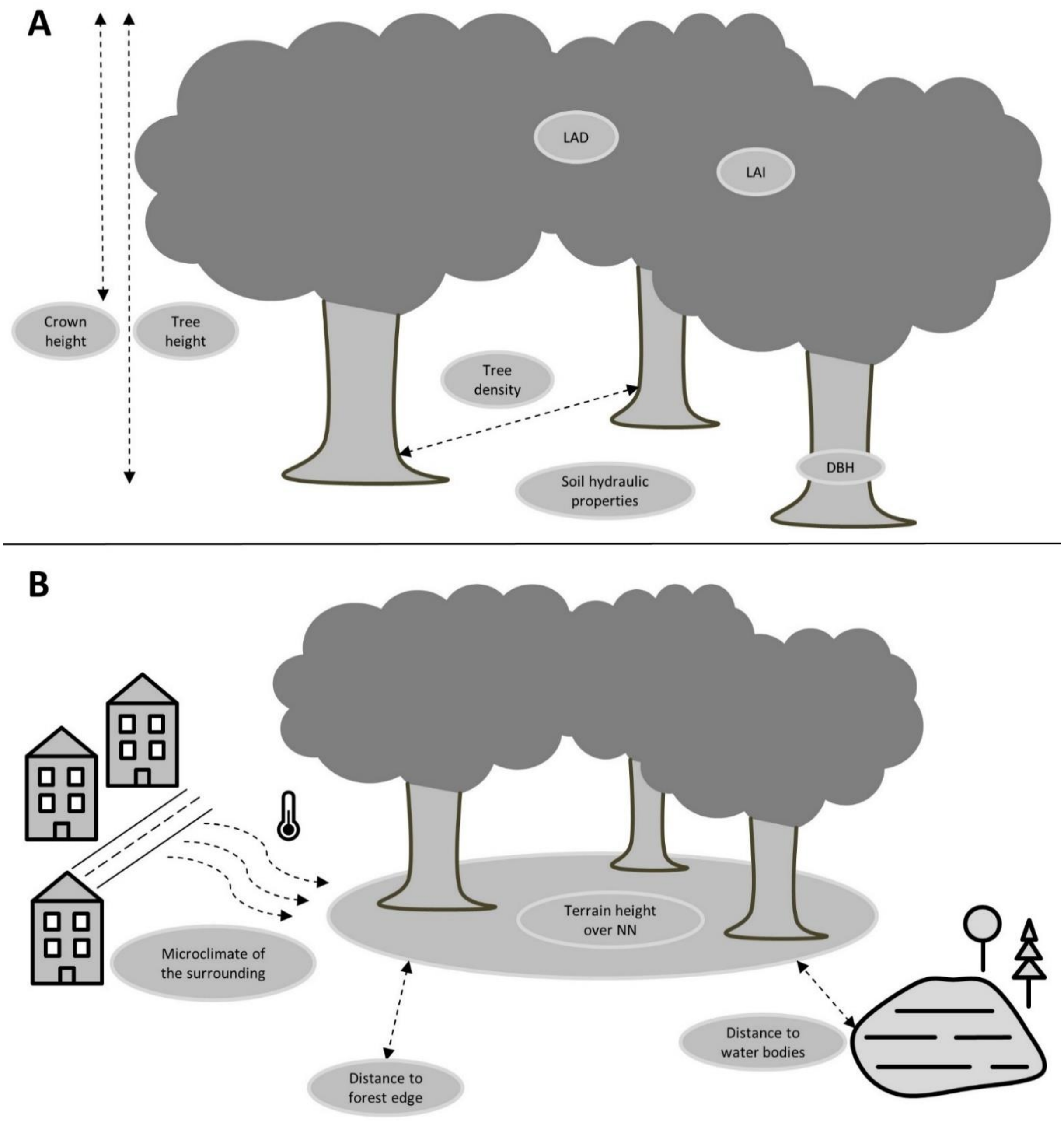


*Figure 36: Factors influencing microclimate of urban forest stands. A= endogenous factors, B = exogenous factors.*

The aim of the experiment was i) to identify, describe, and quantify the cooling function of the different urban forest types, and ii) to analyze, which characteristics of the forest stands itself and of the surrounding are responsible for the cooling effects. To quantify cooling effects, the forest temperatures at the monitoring sites were compared with temperatures measured at an urban climate station. The temperature deviation between the forest and the urban temperature is used as indicator for cooling effects. The temperature and humidity deviation between the forest sites and the urban site were correlated with endogenous and exogenous factors (Figure 36). Endogenous factors are the stand-specific characteristics of the living biomass (LAI, LAD, tree density, DBH, tree height, and canopy height), and soil properties giving information about the amount and accessibility of soil water for evapotranspiration (bulk density, proportion of coarse soil, rooting depth, soil carbon content, plant available water, mean volumetric water content) (Figure 36). Exogenous factors are the characteristics of the urban surrounding, such as closest distance to the forest edge or to open water bodies, the terrain height over NN, and the microclimate of the surrounding.

# 3 Materials and methods

## 3.1 Air temperature and humidity measurements

Air temperature and relative humidity were measured with iButton Hygrochron Temperature/Humidity Data Loggers (DS1923) (maxim integrated, San Jose, California, USA) in an interval of 30 minutes between April 2018 and Mai 2020.

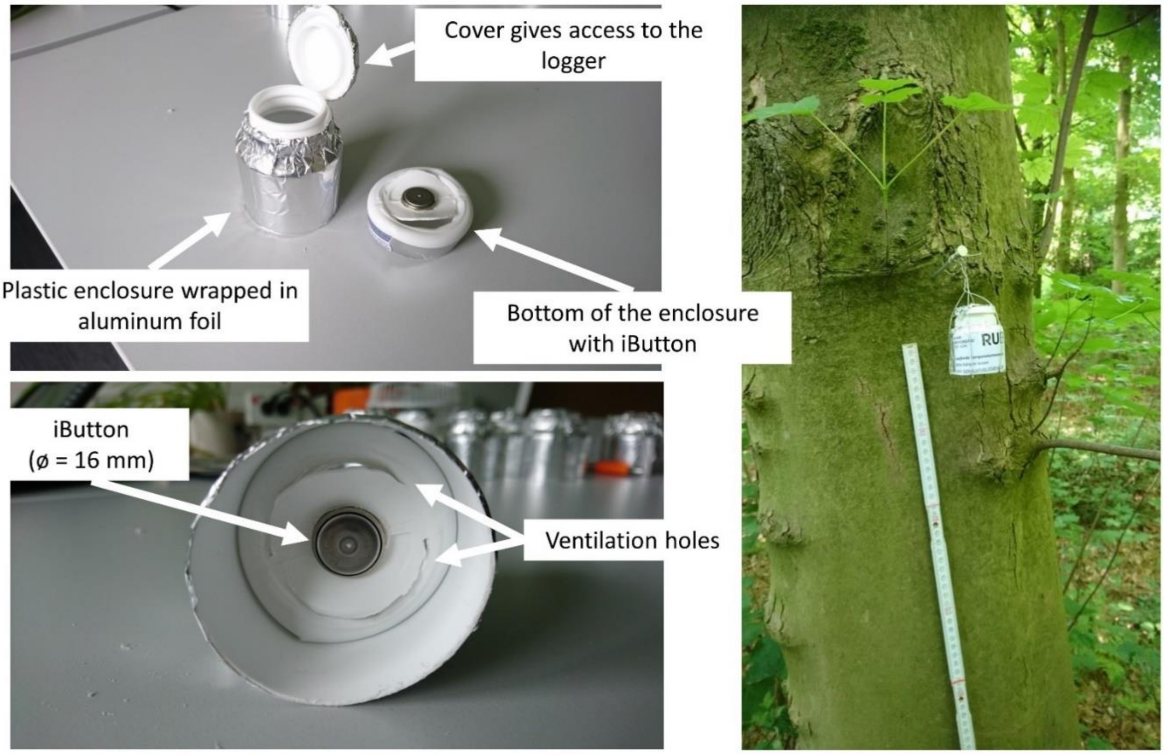


*Figure 37: iButton with enclosure and example from installation.*

The applicability of iButtons for forestry microclimatic studies has been already shown by GREISER et al. (2018), HARDWICK et al. (2015) and GILLNER et al. (2015) amongst others. A self-made plastic enclosure wrapped in aluminum was used to protect the iButton from rain and direct radiation. To prevent heat accumulation, ventilation holes were cut into the bottom of the enclosure. One device (iButton with enclosure) was mounted on a tree in 2 m height in the center of each site. The device was installed at the north side of the tree to widely reduce the influence of direct radiation and to make the measurements comparable between the sites (Figure 37). As reference station, the temperature and relative humidity data from the Ludger-Mintrop Urban Climate Station in Bochum (51° 29' 29.8'' N, 7° 12' 56.6'' E) was used. The urban climate station is located in a small urban gardening area, adjacent to dense residential areas, where the occurrence of the UHI have already been proven (LANUV NRW 2016, 2018b).

## 3.2 Assessment of factors influencing microclimate

### 3.2.1 Endogenous factors

DBH, tree height, tree density, crown height, LAI, LAD, and the soil hydraulic properties (bulk density, proportion of coarse soil, rooting depth, soil carbon content, plant available water, mean volumetric water content) were already examined for the experiment of carbon sequestration (Part II) and the data were also used in this experiment. The methods are described in Part II – chapter 3.

### 3.2.2 Exogenous factors

The shortest distance (Euclidean distance) to the forest edge and to open water bodies were determined with QGIS, version 3.14.10. Terrain height over NN of the site locations were taken from Google Earth information. Information on the surrounding Physiological Equivalent Temperature (PET) at 15:00, nocturnal air temperature at 04:00, and nocturnal cold air flow were taken from the climate analysis of Northrhine-Westphalia (LANUV NRW 2018b). In that study, climate analysis was done with the mesoscale model FITNAH (Groß 1993) for a warm, cloudless summer day without macroclimate pressure gradient air flow on a 100 by 100 m regular grid.

## 3.3 Data analysis

Only days with cloudless sky conditions, no precipitation, and low wind (mean wind speed < 3 m $s^{-1}$) were considered for analysis. On those days, the microclimate is mainly driven by direct and thermal radiation, modified by topography, vegetation, and surfaces. Furthermore, the study was limited to the vegetation period (April - September) to analyze the influence of the vegetation on microclimate. With respect to defined condition, 23 days were considered for analysis. To analyze the diurnal and nocturnal temperature and humidity patterns separately, the diurnal data was split from the nocturnal data by considering daily time of sunrise and sunset. For every time step, the air temperature and relative humidity measured at the forest sites was subtracted from the air temperature and relative humidity measured at the urban climate reference station. Finally, the daily diurnal and nocturnal mean of temperature deviation was used to quantify daily cooling effects.

Because the data was normally distributed, analysis of variance (ANOVA, FISHER 1973) was used to analyze significant differences between the single sites and the forest types. Significant level was defined by $p = 0.05$. Pearson correlations were performed to investigate the main factors influencing daily cooling effects. For significant factors, a stepwise both-sided regression analysis was performed to calculate functions explaining the air temperature and humidity deviation. All statistical analysis were performed with Rstudio, version 1.4.1103, running with R version 4.0.3 (R Core Team 2020).

# 4 Results

## 4.1 Endogenous and exogenous factors influencing microclimate

All sites are plain in topography to avoid impacts on the microclimate by slope and orientation. The forests and soil characteristics of the monitoring sites used for this experiment as endogenous factors are already shown in the $CO_2$ sequestration experiment. The characteristics can be seen in Part II – chapter 4.1 and 4.5.1 and will not be shown here again. The microclimate of the surrounding is represented by the climate analysis of Northrhine-Westphalia (LANUV NRW 2018b) (Table 16). Slight to moderate diurnal heat stress is expected at the sites in Herten (Hert-Bu, Hert-Ei) and at ZeHa, where the PET is smaller than 35 °C, while at BlueHa extreme heat stress (> 41 °C) is expected (VDI 2004, OKE et al. 2017). All other sites are influenced by strong heat stress (35 - 40.9 °C). The climate analysis of Northrhine-Westphalia (LANUV NRW 2018b) defined a nocturnal air temperature at 04:00 > 17 °C as threshold for identification of an urban heat island. Therefore, only the sites Hert-Bu, Hert-Ei, WePa, and ZeHa are located in a weak urban heat island, while at all other sites the nocturnal air temperature is below 17 °C. At Hert-Ei and Hert-Bu, low nocturnal cold air flow is expected (≤ 300 $m^3 s^{-1}$), while all other sites are influenced by moderate cold air flow (> 300 - 1,500 $m^3 s^{-1}$). Most sites are located relatively close to the forest edge (20 - 112 m), which is typical for small urban forest stands in the region. Sites where the microclimate may be influenced by water bodies are WePa, ZeHa, Hert-Ei, and Hert-Bu. The semi-natural forests are more located in lower terrain (< 50 m over NN), while sites on heaps (ReHa-Ah, BlueHa, WePa) are located > 75 m over NN.

*Table 16: Exogenous factors at the monitoring sites. PET (Physiological Equivalent Temperature) at 15:00, nocturnal air temperature at 04:00, and nocturnal cold air flow were taken from climate analysis of Northrhine-Westphalia (LANUV NRW 2018b).*

| Urban forest type | ID | PET at 15:00 | Nocturnal air temperature at 04:00 | Nocturnal cold air flow | Closest distance to forest edge | Closest distance to water bodies | Terrain height over NN |
|---|---|---|---|---|---|---|---|
| | | [°C] | [°C] | [m$^3$ s$^{-1}$] | [m] | [m] | [m] |
| Semi-natural forests | RePa-Bu | 39.8 | 16.6 | 1,189.3 | 82 | 150 | 47 |
| | RePa-Ah | 39.8 | 16.6 | 1,189.3 | 112 | 178 | 47 |
| | Hert-Bu | 28.2 | 18.3 | 201.3 | 145 | 65 | 46 |
| | Hert-Ei | 29.4 | 17.2 | 295.5 | 186 | 32 | 45 |
| Urban greening forests | ReHa-Ah | 39.0 | 16.6 | 1,057.7 | 31 | 276 | 77 |
| | ReHa-Ei | 39.0 | 16.6 | 1,057.7 | 65 | 137 | 58 |
| | BlueHa | 41.3 | 15.8 | 1,451.6 | 105 | 154 | 79 |
| | BoHo | 38.8 | 16.7 | 1,458.5 | 36 | 770 | 72 |
| Succession forests | WePa | 38.4 | 17.7 | 864.8 | 20 | 52 | 85 |
| | ZeHa | 33.3 | 17.1 | 1,271.1 | 32 | 51 | 48 |
| | ReHa-Bi | 39.0 | 16.6 | 1,057.7 | 35 | 496 | 53 |

## 4.2 Temporal differences between the forest and urban climate

For all sites, similar patterns in hourly air temperature deviation between the forest stands and the urban climate station were observed (Figure 38). At night, forest air temperature was mostly higher as those at the urban climate station. At sunrise around 06:00, the deviation turned negative, and largest temperature deviation was found around 09:00 with -4 K at the semi-natural forests and between -4 and -2 K at the urban greening and succession forests. During the day, the forest temperature equilibrated with the urban temperature but still in the afternoon, when heat stress was strongest (15:00), the forest air temperature was on average 1 K lower than at the urban site. Depending on site, the range of afternoon cooling (15:00) in the forest stands was between -2.2 ± 1.6 K at BoHo and -0.2 ± 1.3 K at ReHa-Ei, and only at ZeHa, the average forest air temperature was slightly higher (0.7 ± 0.9 K). Finally, around sunset (20:00), the forest air temperature became again higher as the urban air temperature at all sites.

For the relative humidity deviation, a similar pattern can be observed (Figure 39). The deviation from urban humidity was smallest during night, increased at sunrise and decreased slowly over the course of the day. At the semi-natural sites, the relative humidity was always higher compared to the urban measurements, while the humidity at the urban greening and succession sites was closer to the urban site. Because of these differences between day and night, the diurnal and nocturnal deviations were separately analysed in further analysis.

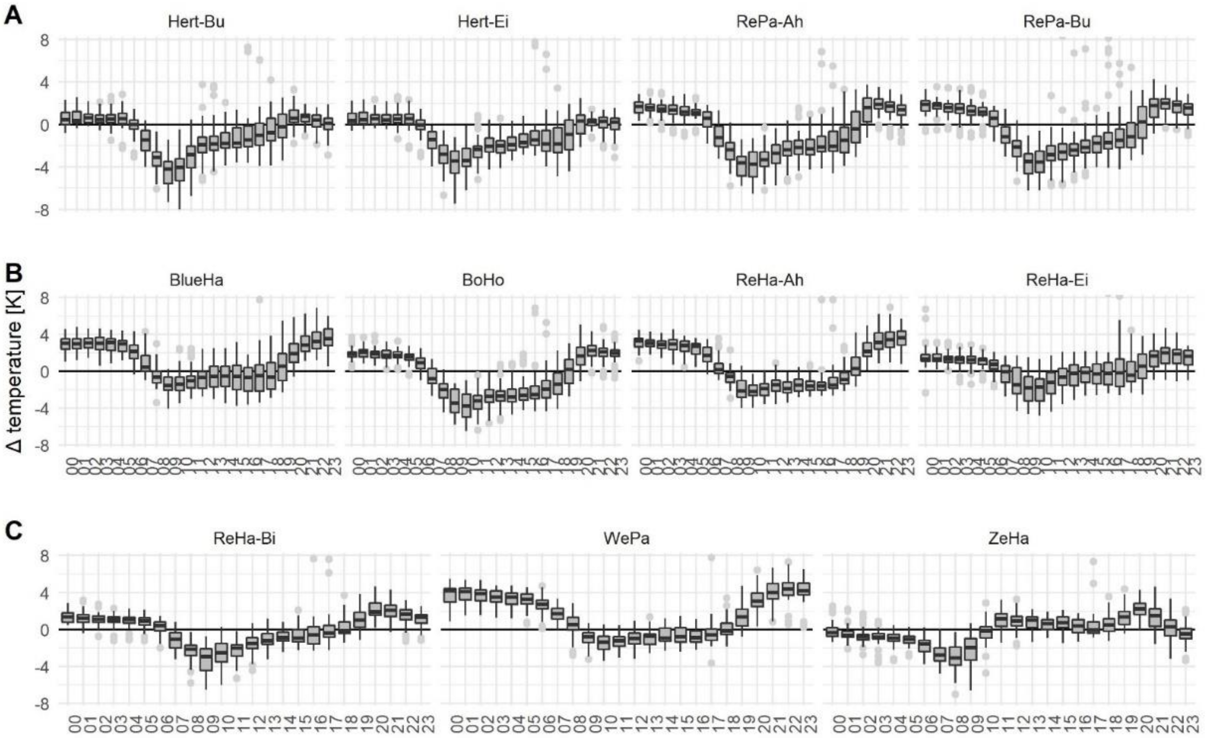


*Figure 38: Hourly air temperature deviation between the monitoring sites and the urban climate station for the 23 sample days. A = semi-natural forests, B = urban greening forests, C = succession forests.*

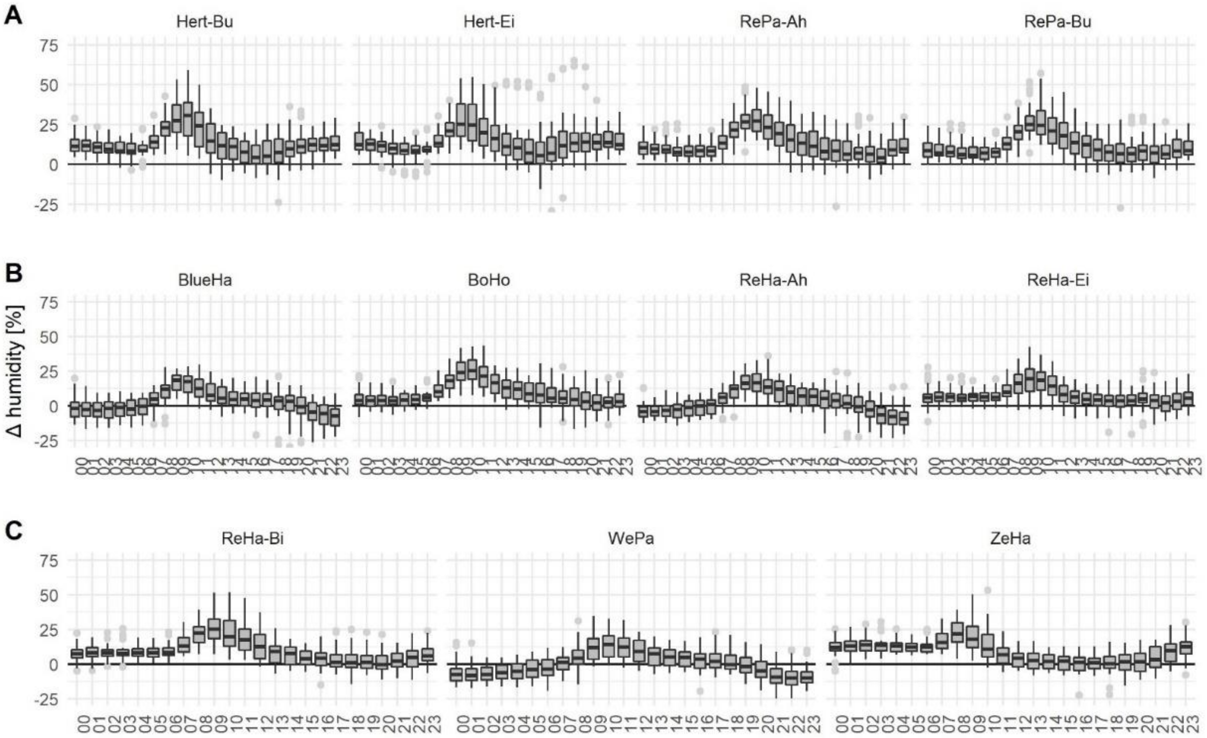


*Figure 39: Hourly humidity deviation between the monitoring sites and the urban climate station for the 23 sample days. A = semi-natural forests, B = urban greening forests, C = succession forests.*

## 4.3 Diurnal situation

The mean diurnal air temperature deviation ranges between -1.75 ± 0.76 K at RePa-Ah and 0.13 ± 0.96 K at WePa, considering the means over all sample days (Figure 40). The whiskers of the boxplots indicated a broader range, where at some days mean air temperature deviation were between -3 K and 1.5 K depending on forest site. A clear diurnal air cooling was observed at the semi-natural (-1.63 ± 0.9 K) and urban greening forests (-0.69 ± 0.9 K). In contrast, the diurnal air temperature deviation for the succession forest (-0.22 ± 0.7 K) is closer to the temperatures measured at the urban climate station.

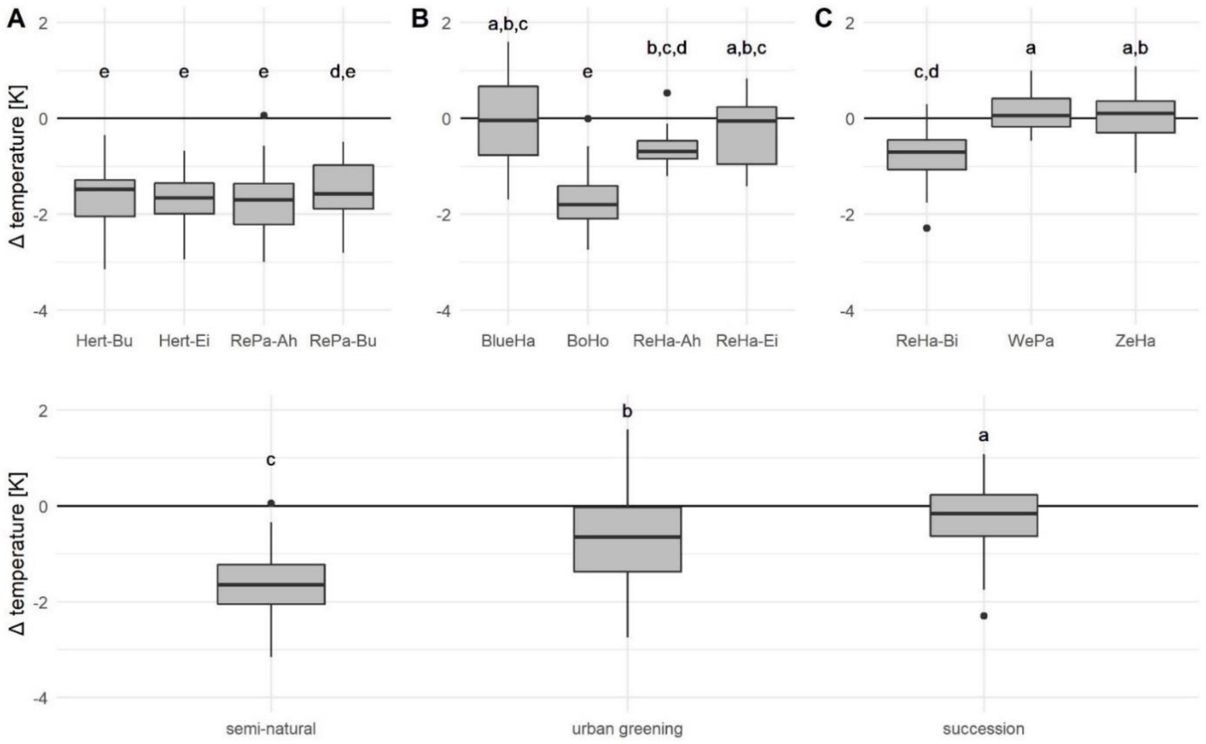


*Figure 40: Diurnal temperature deviation for the single monitoring sites and aggregated for the forest types. A = semi-natural forests, B = urban greening forests, C = succession forests. Letters show significant differences (ANOVA).*

At all sites, the forest mean diurnal relative humidity was larger than at the urban site and ranged between 16 ± 9 and 4.4 ± 6.8 % (Figure 41). At some days, a humidity deviation between 30 and -5 % were detectable, as indicated by the whiskers of the boxplots. Significantly largest diurnal humidity deviation was recorded for the semi-natural forests (14.9 ± 7.1 %), while there were no significant differences between urban greening (8.4 ± 6.2 %) and succession forests (7.3 ± 6.5 %).

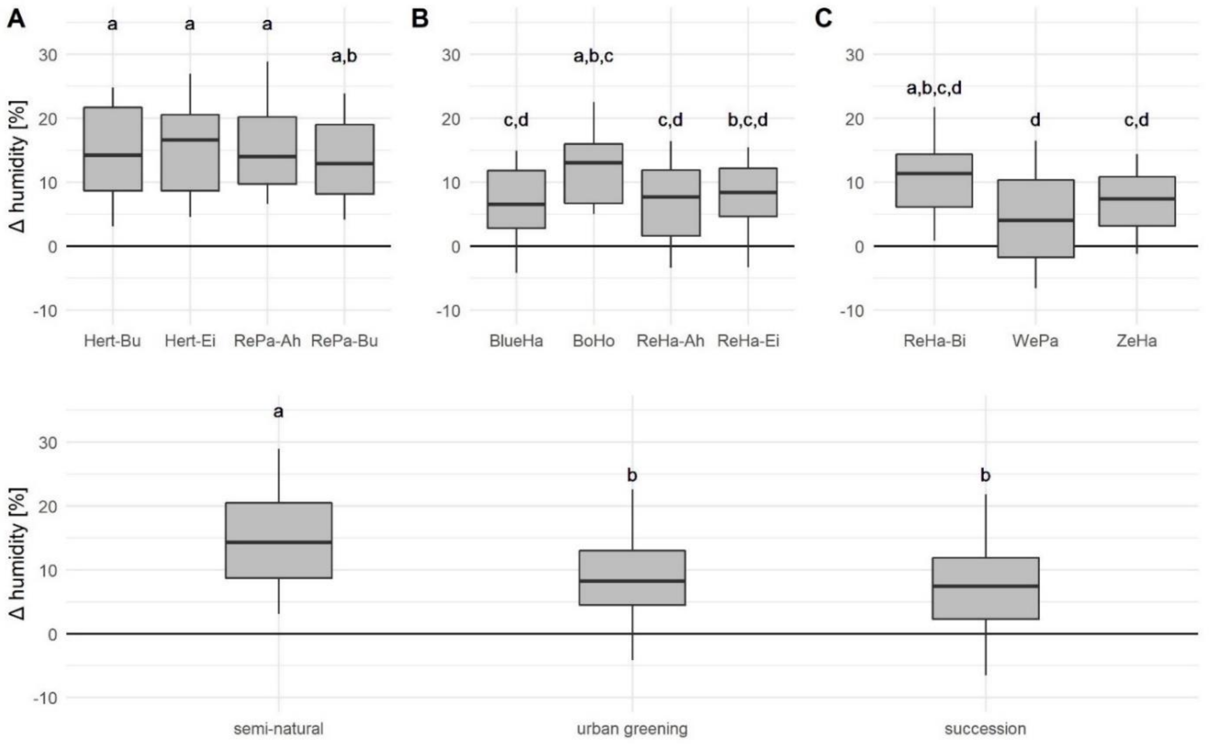


*Figure 41: Diurnal humidity deviation for the single monitoring sites and aggregated for the forest types. A = semi-natural forests, B = urban greening forests, C = succession forests. Letters indicate significant differences (ANOVA).*

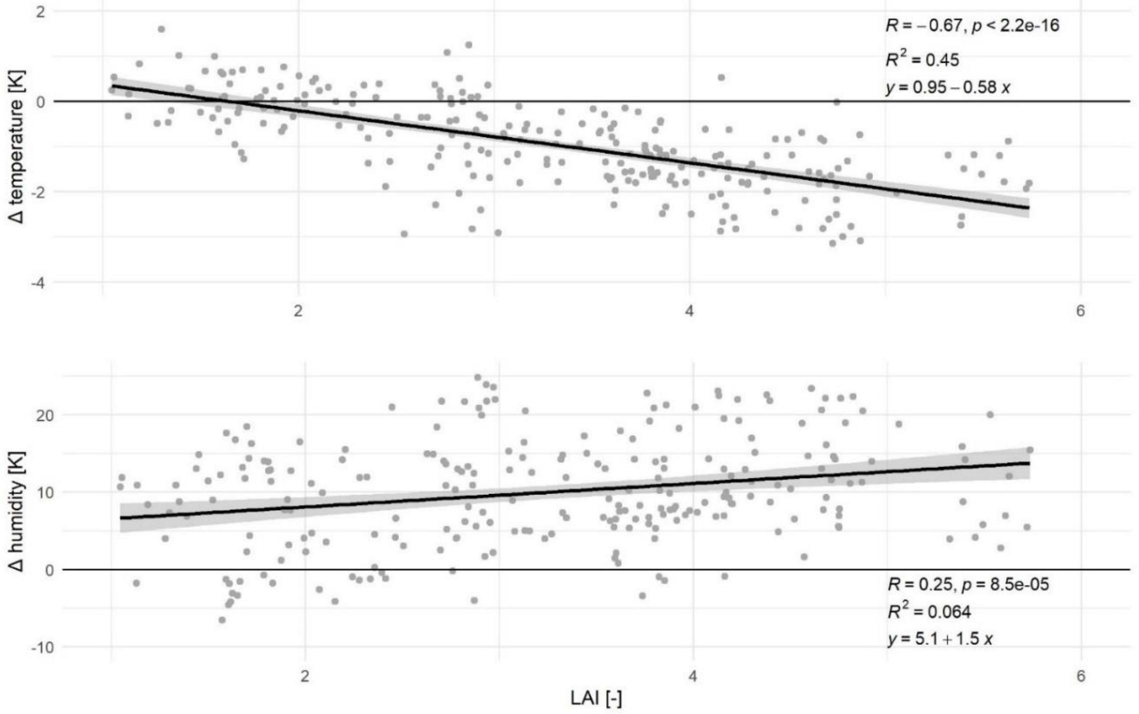


*Figure 42: Influence of LAI on daily diurnal temperature and humidity deviation for the 23 sampling days over all sites (n=253).*

Figure 42 shows a regression analysis for the mean diurnal air temperature and humidity deviation and the LAI for the specific sampling day over all sites. A clear negative correlation between LAI and mean diurnal air temperature deviation ($r = -0.67$, $p < 0.0001$) was found. It can be also seen that the temperature deviation turns negative at a minimum LAI of 2. Consequently, in urban forests with LAI lower than 2 air temperature is comparable to urban sites or are even warmer than those. Vice versa, forests diurnal air cooling occurs only for forests with an LAI larger than 2 and the cooling ability increases with increasing LAI. For humidity, only a weak positive correlation ($r = 0.25$, $p < 0.0001$) was found, indicating that the relative humidity is in general larger in urban forest stands in comparison to urban areas and slightly increases compared to the urban station with larger LAI.

*Table 17: Results from the correlation analysis between the mean diurnal air temperature and humidity deviation and the endogenous and exogenous factors potentially influencing microclimate. Bold letters indicate significant relationships with * = low significance ($p < 0.05$), ** = medium significance ($p < 0.01$), *** = high significance ($p < 0.001$).*

| **Factors** | **Measures** | **Δ mean temperature [K]** | | | **Δ mean humidity [%]** | | |
|---|---|---|---|---|---|---|---|
| | | **r** | **$R^2$** | **p** | **r** | **$R^2$** | **p** |
| **Endogenous factors** | | | | | | | |
| Mean LAI | [-] | **-0.82** | **0.67** | **0.002**** | **0.66** | **0.43** | **0.028*** |
| Bulk density (B-horizon) | [kg cm$^{-3}$] | **-0.74** | **0.55** | **0.009**** | **0.69** | **0.48** | **0.018*** |
| Proportion of coarse soil (> 2 mm) (B-horizon) | [%] | **0.72** | **0.52** | **0.013*** | **-0.65** | **0.42** | **0.03*** |
| Mean canopy height | [m] | **-0.72** | **0.51** | **0.013*** | **0.79** | **0.63** | **0.0037**** |
| Mean tree height | [m] | **-0.71** | **0.50** | **0.015*** | **0.73** | **0.53** | **0.011*** |
| Mean DBH | [cm] | **-0.7** | **0.49** | **0.017*** | **0.79** | **0.63** | **0.0037**** |
| Proportion of coarse soil (> 2 mm) (A-horizon) | [%] | **0.61** | **0.37** | **0.047*** | -0.49 | 0.24 | 0.13 |
| Tree density | [tree number m$^{-2}$] | 0.45 | 0.2 | 0.17 | **-0.62** | **0.38** | **0.043*** |
| **Exogenous factors** | | | | | | | |
| Closest distance to forest edge | [m] | -0.59 | 0.35 | 0.056 | **0.7** | **0.49** | **0.016*** |
| Terrain height over NN | [m] | 0.53 | 0.29 | 0.091 | **-0.73** | **0.54** | **0.01*** |

Table 17 shows the results from the single correlation analysis. In comparison to the regression analysis in Figure 42, the correlation and regression analysis were performed for the site-specific mean air temperature and humidity deviation at the monitoring sites over all sample days ($n = 11$), because all factors, except LAI, considered in the analysis are static and did not change during the experimental period significantly. The mean air temperature deviation was negatively correlated with mean LAI, bulk density of the B-horizon, mean canopy height, mean tree height, and mean DBH, and positively correlated with the proportion of coarse soil of the A- and B-horizon. No significant correlations were found for the exogenous factors. The mean diurnal humidity deviation was positively correlated with LAI, bulk density of the B-horizon, mean canopy height, mean tree height, mean DBH, and the closest distance to forest edge and negatively correlated with the proportion of coarse soil of the B-horizon, the tree density, and the terrain height over NN. All other factors were not significantly correlated with the temperature and humidity deviation.

*Table 18: Results from stepwise both-sided regression for the significant factors. * = low significance ($p < 0.05$), ** = medium significance ($p < 0.01$), *** = high significance ($p < 0.001$).*

| **Coefficients** | **Measures** | **Estimate** | **Std. Error** | **t value** | **p** |
|---|---|---|---|---|---|
| ***Diurnal temperature deviation*** | | | | | |
| (Intercept) | [-] | 1.565872 | 0.380764 | 4.196 | 0.003** |
| LAI | [-] | -0.567664 | 0.119969 | -4.845 | 0.001** |
| Mean DBH | [cm] | -0.025625 | 0.007607 | -3.501 | 0.098** |
| ***Diurnal humidity deviation*** | | | | | |
| (Intercept) | [-] | 12.33379 | 3.54284 | 3.643 | 0.013* |
| LAI | [-] | 2.08050 | 0.70592 | 3.132 | 0.026* |
| Terrain height over NN | [m] | -0.09846 | 0.03771 | -2.734 | 0.040* |
| Tree density | [trees 100 $m^{-2}$] | -15.24463 | 5.85306 | -2.803 | 0.040* |
| Proportion of coarse soil (> 2 mm) (B-horizon) | [%] | -0.03801 | 0.02581 | -1.651 | 0.1913 |

Results from the stepwise regression analysis are given in Table 18. Mean diurnal temperature deviation can be explained by:

$$\Delta T = y + a * LAI + b * DBH \qquad [8]$$

where *ΔT* is the mean diurnal air temperature deviation [K], *y* is the intercept shown in Table 18, *a* and *b* are coefficients shown in Table 18, *LAI* is the leaf area index [-] and *DBH* is the diameter at breast height [cm].

Mean diurnal air humidity deviation can be explained by:

$$\Delta H = y + a * LAI + b * TN + c * TD + \ d * B.coarse.soil \qquad [9]$$

where *ΔH* is the diurnal air humidity deviation [%], *y* is the intercept shown in Table 18, *a, b, c*, and *d* are coefficients shown in Table 18, *LAI* is the leaf area index [-], *TN* is the terrain height over NN [m], *TD* is the tree density [trees 100 $m^{-2}$] and *B.coarse.soil* is the proportion of coarse soil (> 2 mm) in the B-horizon [%].

For evaluation of the stepwise regression, the measured mean diurnal air temperature and mean humidity deviation of the sample days were plotted along with the estimated diurnal deviations in Figure 43. For the mean air temperature deviation, the mean error is -0.002 K and the 25 %- and 75 %-quantile is -0.45 K and 0.45 K, respectively. Therefore, the air temperature model [Eq. 8] can predict the diurnal air cooling of urban forest ecosystems with a standard deviation of 0.68 K. The model for the humidity deviation [Eq. 9] has a mean error of 0.01 % and a 25 %- and 75 %-quantile of -5.94 % and 5.59 %, respectively, yielding a precision of 6.7 %.

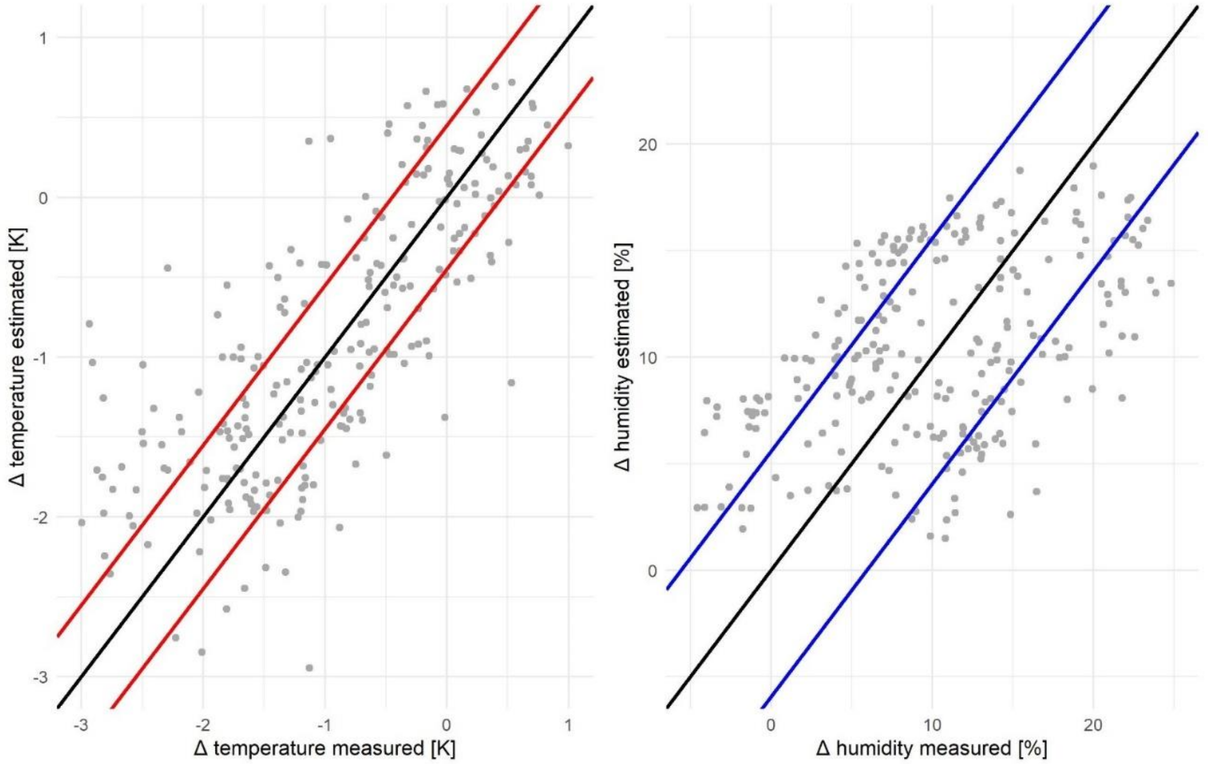


*Figure 43: Evaluation of the models from the stepwise regression, where the measured diurnal temperature and humidity deviation is plotted against the estimated deviations. The red and blue lines show the 25 %- and 75 %-quantile of the variation, the black line is the line through the origin.*

## 4.4 Nocturnal situation

Except of ZeHa (-0.53 ± 0.5 K), nocturnal forest air temperatures were higher compared to the urban temperatures at all investigated sites and ranged between 0.4 ± 0.5 K at Hert-Ei and 3.71 ± 0.8 K at WePa (Figure 44). The whiskers of the boxplots show that a broader range (-2 - 5 K) is possible. Significant differences were found between the forest types, where semi-natural forests (0.96 ± 0.7 K) were closest to the urban temperatures, followed by the succession forests (1.44 ± 1.9 K), while the urban greening forests had the largest deviation from urban temperatures (2.28 ± 0.9 K).

For humidity, at nearly all sites higher nocturnal relative humidity were found in comparison to the urban relative humidity (Figure 45). The deviations ranged between -6.38 ± 5.6 % at WePa and 11.82 ± 3.9 % at Hert-Ei. Considering the whiskers of the boxplots, nocturnal humidity deviations between -15 to 20 % were possible. The forest types showed significantly different deviations, where the deviation is largest at the semi-natural forests (10.29 ± 4.1 %) and smallest at the urban greening forests (0.72 ± 6.5 %).

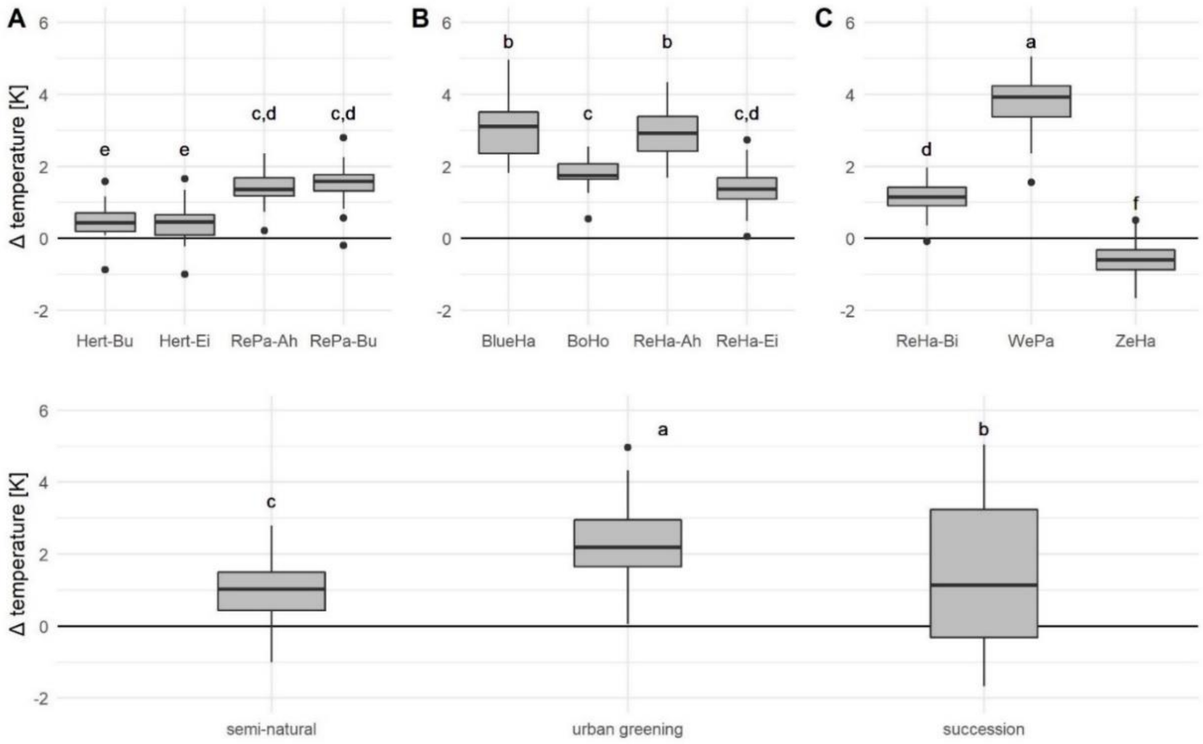


*Figure 44: Daily nocturnal air temperature deviation for the single monitoring sites and aggregated for the forest types. A = semi-natural forests, B = urban greening forests, C = succession forests. Letters indicate significant differences (ANOVA).*

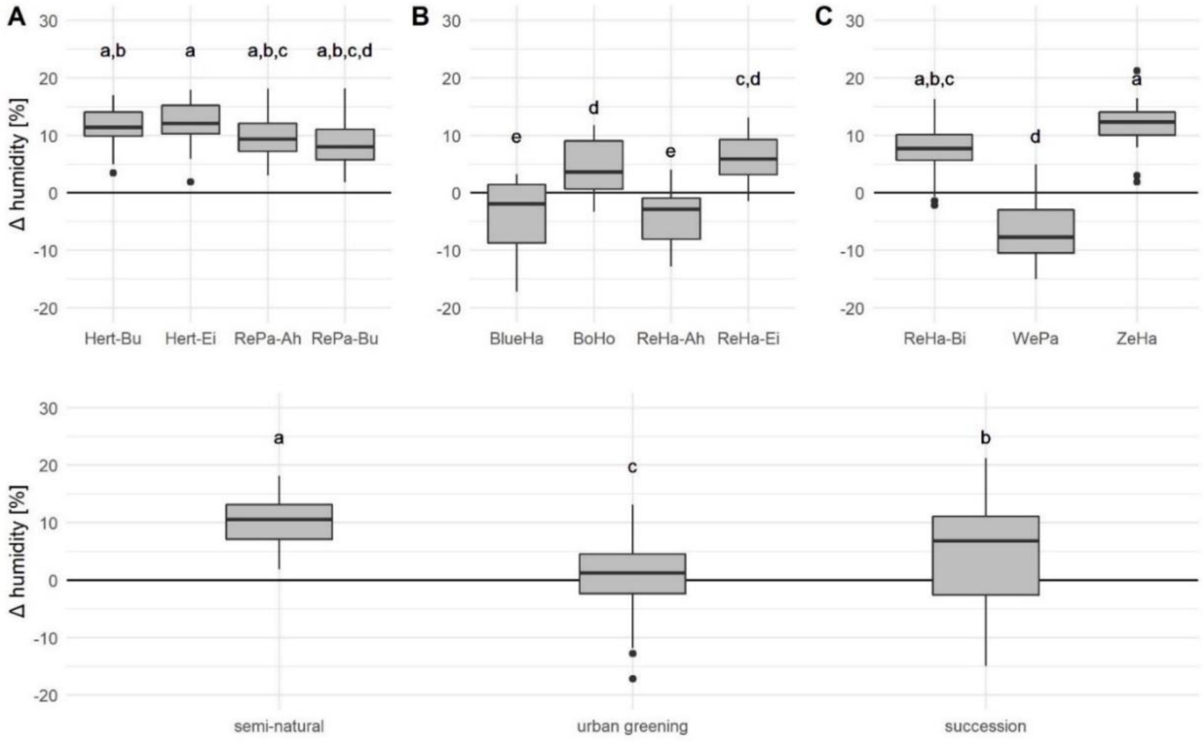


*Figure 45: Daily nocturnal relative humidity deviation for the single monitoring sites and aggregated for the forest types. A = semi-natural forests, B = urban greening forests, C = succession forests. Letters indicate significant differences (ANOVA).*

Like for the diurnal situation, the site-specific mean nocturnal air temperature and humidity deviation over all 23 sample days (n = 11) were correlated with endogenous and exogenous factors influencing the forest microclimate and only strong statistical relationships with the terrain height over NN were found (Figure 46). Therefore, it can be concluded that the nocturnal forest air was even warmer (r = 0.89, p < 0.001) and drier (r = -0.96, p < 0.001), the higher the urban forests were located in the terrain. Furthermore, a weak but significant relationship was found between PET at 15:00 and the nocturnal temperature deviation, where the nocturnal forest air was even warmer, the warmer the PET at 15:00 was. For all other endogenous and exogenous factors no significant relationship was found.

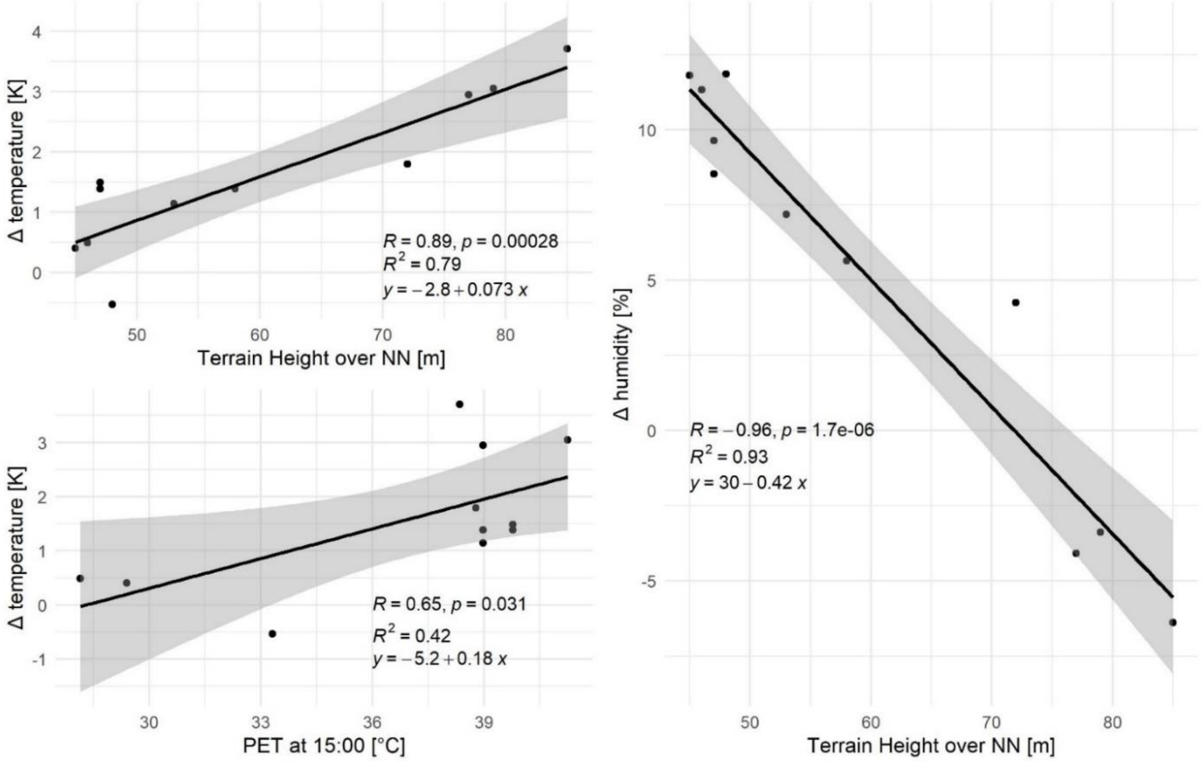


*Figure 46: Correlation analysis for the site-specific mean nocturnal air temperature and humidity deviation.*

# 5 Discussion

## 5.1 Cooling function of urban forests

In general, the air temperature at the investigated urban forest stands was cooler at daytime and warmer at night in comparison to the urban reference temperatures. Therefore, it can be concluded that the cooling effect of urban forest stands occurs only over the day, while the cooling function of open urban spaces, like at the urban climate station, is stronger during night. Strongest diurnal cooling occurred between sunrise and noon with -4 to -2 K. In the afternoon, the forest temperature slowly equilibrated with the urban temperature (-1 K), but depending on the forest site, the cooling effect can be still up to -2.2 K (15:00) in comparison to the urban site. Over the entire day, diurnal cooling is strongest at the semi-natural forests (-

1.6 K), followed by the urban greening forests (-0.7 K), and the succession forests (-0.2 K). For interpretation, it has to be noted, that the Ludger-Mintrop Urban Climate Station is located in a small urban gardening area with a high sky view factor, where the occurrence of the UHI is proven (LANUV NRW 2016, 2018b). However, the cooling ability of urban forest stands might be even larger, if the forest air temperatures would be compared with air temperatures measured at a highly sealed area close to roadsides or buildings.

The measured air temperature deviations are caused by different radiation and thermal conditions in forests in comparison to open urban sites (Figure 47). During the day, short-wave radiation is mostly absorbed and reflected in the upper layers of the forest canopy and only a small proportion access the trunk space (LARCHER 2003). In European beech stands, for example, only 8 % of the irradiance reaches the forest floor under a fully developed canopy (KIESE 1972, ELLENBERG et al. 1982). In consequence, the diurnal air temperatures in the trunk space are significantly lower than in the canopy, leading to a diurnal stable stratification (inversion), where air exchange is only possible by wind gusts (FOKEN 2017).

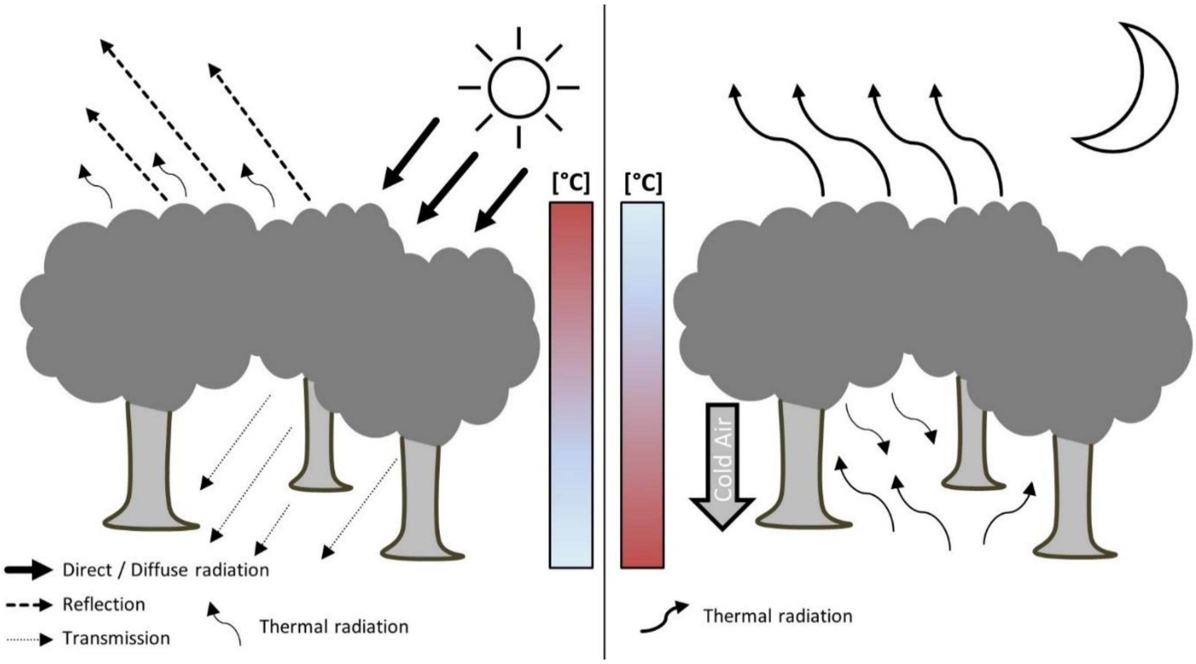


*Figure 47: Schematic radiation patterns and idealized vertical temperature gradients in forests for the diurnal (left) and nocturnal (right) situation. Radiation intensity is indicated by the thickness of the arrows. Blue colours indicating colder air temperatures and red colours warmer air temperatures.*

During night, the stratification gets unstable. The air of the canopy layer cools down and will sink to the forest floor (KIESE 1972). However, this cooling effect is partly compensated by long wave radiation, which is reflected by the trees and the canopy, and therefore, warm air is trapped in the trunk space, leading to balmier nocturnal air temperatures under the canopy compared to those at open (urban) spaces (FOKEN 2017). At the urban climate reference station, the opposite radiation and thermal conditions occur, where the stratification is unstable during daytime and stable during night with unhindered thermal radiation. Therefore, forest air temperature is more balanced between day and night compared to urban air temperatures, and it can be concluded, that the cooling function of urban forest stands occurs only at daytime, while at night cooling is stronger at urban open spaces. Those mechanisms were also recently found for street trees, indicating, that those effects in forest stands are also valid for isolated trees in urban environments (WUJESKA-KLAUSE & PFAUTSCH 2020).

Recent research from remote sensing confirms the daytime cooling and nighttime warming of forest canopies (MEIER et al. 2019). Daytime cooling is explained by higher surface roughness of the tree canopies, leading to larger turbulent heat flux and evaporative cooling in comparison to open spaces (LEE et al. 2011, DAVIN & de NOBLET-DUCOUDRÉ 2010). In contrast, the mechanism of nighttime warming is not yet fully understood (MEIER et al. 2019). Reasons for the warming can be a stronger green house gas effect due to a moister boundary layer or higher aerosol loading over forests (VANDEN BROUCKE et al. 2015). SCHULTZ et al. (2017) and MEIER et al. (2018) explain the warming effect with heat storage in the biomass, where heat is stored in the biomass over the day and at night, this heat is slowly released. Because forest canopies have more biomass and larger surfaces than open urban spaces, more heat might be accumulated over the day and finally released during night. These mechanisms might be also valid for the monitoring sites. However, these effects are confounded by other processes because in the presented study the strength of nocturnal warming was explained only by the terrain height of the sites and not by the attributes of the forest canopy.

## 5.2 Factors influencing diurnal cooling und nocturnal warming

The diurnal cooling was significantly different between the examined forest sites and types, whereby semi-natural forests showed strongest diurnal cooling in comparison to urban greening and succession forests. Correlation and stepwise regression analysis indicated that the diurnal cooling can be explained by forest structure: Diurnal cooling increased with larger LAI, canopy height, tree height, and DBH. The reason for that can be found in the fact that forests with denser canopies have larger surfaces for blocking direct radiation. The larger the tree crown is, the more direct radiation will be absorbed and reflected while passing through the canopy, and less irradiation will reach the trunk space. Diurnal relative humidity deviation was influenced by nearly the same factors as the diurnal cooling. Results from the stepwise regression showed that it is possible to estimate the diurnal cooling of urban forest stands with the stand-specific LAI and DBH with a standard deviation of 0.68 K quite precisely. The regression analysis for the LAI of the specific day indicated that a diurnal cooling occurs only at a LAI larger than 2 and the cooling increased with increasing LAI.

Furthermore, diurnal cooling was even stronger for forests with high bulk density in the B-horizon and low proportion of coarse soil material in the soils. However, these relationships can be indirect as both factors are also correlated with LAI ($r = 0.68$, $p = 0.02$; $r = -0.68$, $p =$

0.038). Consequently, at sites with loose and coarse soils forests were found with more open canopies, where more radiation can access the trunk space. For that, two relationships are likely: i) coarse and loose soils provide unfavorable growth conditions for forests because the soils can store only low amounts of water and dry out relatively quickly, especially under drought conditions like during the experimental period (compare findings from carbon sequestration in Part II). Under water stress, the trees react by dropping their leaves, leading to lower mean LAI values as shown by the carbon sequestration experiment (Part II). Or ii) the coarse and loose soils were only found at the *Betula pendula* succession forests. In those stands, the canopy is typically more open than at other deciduous tree stands (e.g., URI et al. 2007, JOHANSSON 1999). Therefore, the low LAI values were not a consequence of unfavorable growth conditions rather than being caused by the tree biology. The second explanation might be more realistic, because no correlations was found between other soil properties (soil carbon content, soil depth) or soil water content (plant available water, mean volumetric water content) and the air temperature deviation. However, due to water shortage during the experimental period, it can be concluded that the investigated cooling effects were mainly caused by shading from the tree canopies rather than by evapotranspiration. All forest soils were highly unsaturated (dry). Therefore, evapotranspiration should have been quite low for both years and had no or only little effect on the forest air cooling. Based on this, it can be hypothesized that diurnal cooling might be greater if the soils are wetter and the trees are able to transpire water to the atmosphere.

No correlations were found between the diurnal cooling and the exogenous factors, indicating that the investigated urban forest stands were closed systems during the day and that the urban air temperatures were modified by the vegetation to such extend, that the forest air temperature was not influenced by the urban surrounding, even though the relative air humidity increased with distance to the forest edges and at sites located in lower terrain. That changed during night. Unlike during the day, the results from correlation analysis indicated that the nocturnal warming was not influenced by the stand-specific characteristics. But nocturnal warming was mainly influenced by the terrain height of the single sites. The lower the sites were located in the terrain, the smaller was the nocturnal warming. Catabatic winds might be an explanation for these patterns (GRUDZIELANEK & CERMAK 2018, 2015). In forests which are located at low terrain levels, more cold air accumulates from the surrounding than at forests at the tip of heaps, where cold air is generated and drains into the surrounding by following the slopes. Heap tips stay longer above the nocturnal inversion compared to locations in lower terrain height (RVR 2013). Relative humidity was also mainly influenced by the terrain height, as the cold air inflow lowered the forest air temperatures and increased relative humidity. Beside these two effects, a weak positive correlation was found between the nocturnal warming and the PET at 15:00 of the surrounding, and therefore, nocturnal warming was even larger at forests where strong thermal heat stress was expected.

A comparable study examined the land surface temperature of urban forests by Landsat-7 satellite images and correlated the urban cool island (UCI) intensity with different forest structural attributes and their spatial location (REN et al. 2018). The authors also found that the UCI intensity was impacted by LAI, crown closure, tree height, and basal area, but also by the distance to buildings and distance to open water bodies. The findings from the monitoring sites are therefore in line with those from the remote sensing approach. However, no correlation between the cooling ability and the distance to the forest edge or to open water bodies was found.

## 6 Conclusion

The aim of the experiment was i) to identify, describe, and quantify the cooling function of the different urban forest types, and ii) to analyze which characteristics of the forest stands itself and of the surrounding are responsible for the cooling effects. Microclimatic cooling depends on the size of the tree traits, and especially on size and the density of the canopy. Therefore, the diurnal cooling is strongest in semi-natural forests, which are characterized by the largest tree traits. In urban greening and succession forests, the canopy is smaller and more open, leading to temperatures comparable to open sites. Exogenous factors did not influence the diurnal temperature deviation in urban forests compared to open urban sites. Microclimatic cooling in urban forests only occurs during daytime, while during nighttime, forests are warmer than open urban sites. The warming is even higher, the higher the forest is located in the terrain due to the exposition of catabatic winds.

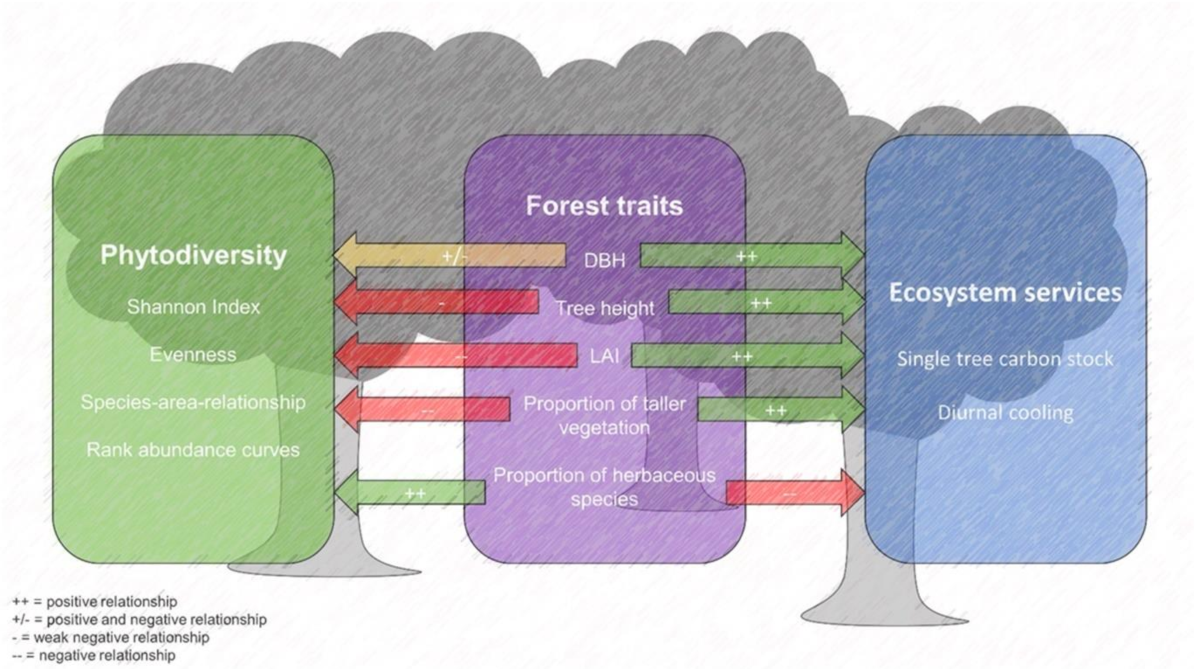
## 7 Implications for urban forestry and planning

The experiment showed that urban forest stands provide an effective shelter for urban dwellers to escape diurnal heat stress, where the mean diurnal cooling ability can be up to -4 K in comparison to open urban spaces. The cooling effect strongly depends on the size and density of the canopy, where a minimum LAI of 2 is needed to provide effective cooling and the cooling increases with increasing LAI. However, nocturnal air temperatures of forests are warmer than those of open urban sites, and therefore, it can be concluded that open urban spaces are better in generating and draining cold air to overheated urban areas. For urban planners, who are considering green infrastructure to cool cities, following implications can be made based on the presented study:

- By planning urban green infrastructure to improve cooling effects, planners should consider both, forested areas, which provide shelter against heat stress over the day, and open spaces such as lawns, which are warmer over day but are able to cool cities at night.

- Diurnal cooling effects are greater in old-grown semi-natural forests than in newly planted urban greening or in succession birch forest stands. Therefore, especially semi-natural forests in urban areas should be preserved against other uses.
- The size and density of the forest canopy mainly determine the cooling function of urban forests. Planners should enable good soil condition when establishing new forests, so the trees can grow old and build up large, vital, and dense canopies. Additionally, site conditions should be considered for tree selection.
- Scientists and urban planners can use Eq. 8 and 9 to estimate cooling ability of existing or planned deciduous urban forest stands for climate adaption planning in cities.

# Part IV – Phytodiversity and Phytodiversity-ES relationship



# 1 Theoretical background

In scientific and policy literature it is often postulated that there is strong correlation between phytodiversity, and the provision of ES and that a consideration of green infrastructure in planning leads to an improvement in both, biodiversity, and ES simultaneously (DG ENVIRONMENT 2012, SCHWARZ et al. 2017). However, empirical evidence of those relationships is often lacking (SCHWARZ et al. 2017, KNAPP et al. 2018, KABISCH et al. 2016, ZITER 2016). The relationship between biodiversity and the provision of ES is often complex, and generalized evidence cannot be drawn. A main reason for that lack of understanding is that both concepts are very broad and consider many different aspects. ES include manifold services ranging from the provision of food and other natural resources, over socio-cultural and regulating services, to services for ecosystem functioning (CONSTANZA et al. 1997, DE GROOT et al. 2002). Biodiversity can be described on different scales, from the diversity of habitats (KALLIMANIS et al. 2008), over the species richness of or in communities (HALPERN & SPIES 1995), to the genetic diversity (VELLEND 2004). It can include the complete biodiversity of an area (PAILLET et al. 2010), or the diversity in single species groups, like insects (HUMPHREY et al. 1999), plants (BARTHLOTT et al. 2007), birds (MELLES et al. 2003), mammals (MOHD 2006), or others. Furthermore, the diversity of different plant traits, functional groups, or structural components of a habitat can describe biodiversity (ZITER 2016). Relationships that can be found in specific habitat types or biomes are not general and can therefore hardly be transferred to other habitat types or biomes as they might differ in their conditions. Therefore, it is crucial that case studies investigating the relationship between biodiversity and ES clearly state which aspects of biodiversity and which ES have been considered in the analysis.

Many studies indicate that the provision of ES is more related to the diversity of species traits rather than to taxonomic diversity (MCGILL et al. 2006, DÍAZ & CABIDO 2001, DE BELLO et al. 2010, LAVOREL 2013, Moretti et al. 2013). Therefore, trait-based approaches are more commonly used to analyze the relationship between biodiversity and ES and functions (RICOTTA & MORETTI 2011). Two main theories are used to explain the relationships: i) the ‘mass-ratio theory’ by GRIME (1998) and ii) the ‘niche complementary theory’ firstly published by TILMAN et al. (1996). Both theories do not contradict each other. Rather they represent the two sides of functional traits: dominance and divergence (ALI & YAN 2017). In general, the mass-ratio theory states that the ES provision is mainly controlled by the traits of a few dominant species in a community, whereas the species richness of subordinates and transient species have no or only little effect on ES provision (GRIME 1998). In contrast, the niche complementary hypothesis states, that a high variation of species traits (as a proxy of species richness) leads to a more extensive and efficient usage of resources, which generates a higher productivity and sustainability of the plant community (HOOPER & DUKES 2004, DÍAZ et al. 2007, TILMAN et al. 1997). This theory had been proven especially for grasslands, where the plant productivity and nitrogen utilization increased with increasing plant species richness, while nitrogen leaching decreased (TILMAN et al. 1996). In forests, the mass-ratio theory tends to be more common, especially for biomass and carbon storage estimations where the above-ground biomass and C stock is determined by the dominant tree species and not by the understory vegetation (ALI & YAN 2017, FOTIS et al. 2018). However, there are also studies for forest stands documenting the validity of the niche complementary theory (NIKLAUS et al. 2017, MORIN et al. 2011. MENSAH et al. 2018) and studies in which both theories are supported (HAO et al. 2020, GARCÍA-PALACIOS 2017). MADRIGAL-GONZÁLEZ et al. (2016) showed that it depends on tree size and climatic conditions whether the mass-ratio or complementarity effects explains the ES provision.

Complementarity effects in forests occur when mixed stands are compared with monocultures. Different studies imply that forest productivity is about 24 % higher in mixed stands compared to monocultures and forest productivity increases with increasing number and evenness of tree species (ZHANG et al. 2012, VILÀ et al. 2013, GAMFELDT et al. 2012). Those complementarity effects are explained by i) facilitation and ii) competitive reduction. Facilitation occurs when one species improves the availability of resources for other species (e.g., symbiotic nitrogen fixation) and competitive reduction occurs when the intense intra-specific competition in monocultures is replaced by less intense inter-specific competition in mixed stands (FORRESTER & BAUHUS 2016). Inter-specific competition is less intense because the single species differ in tree structure, growth, phenology, and utilization of resources, and therefore, more spatial, temporal, chemical, and ecological niches can be filled, leading to a more extensive and efficient usage of resources, compared to monocultures. However, little is known about how the complete species richness and evenness, including the shrub and herbaceous layer, supports forest productivity and other ES.

All studies mentioned so far were performed in rural areas, while investigations in urban areas are sparsely found (SCHWARZ et al. 2017). ES provision by urban spontaneous vegetation were already investigated by ROBINSON & LUNDHOLM (2012). However, only tall forbs and grasses as urban spontaneous vegetation were considered, while urban forests were not assigned to urban spontaneous vegetation. Furthermore, diversity effects were often analyzed in terms of forest productivity and not for the provision of regulating ES for urban dwellers such as cooling effects of forests. Therefore, no evidence of whether complementarity effects or mass-ratio effects are valid for urban forest systems regarding the biodiversity-ES relationship.

## 2 Aim and design of the experiment

To address these knowledge gaps, the relationship between the phytodiversity and the provision of regulating ES were analyzed for the urban forests found in the study areas already introduced in chapter 2.5 of the introduction. Phytodiversity is defined here as the species richness and evenness of vascular plants in forest habitats. The aim of the experiment was i) to analyze the differences in forest traits, phytodiversity, and ES provision between the different forest types introduced in chapter 2.2 of the introduction, and ii) to analyze the influence of forest traits on phytodiversity and ES provision to find out whether complementarity of mass-ratio effects can describe the phytodiversity-ES relationship in urban forests. Following research questions were addressed:

- How does the species composition and the forest traits (DBH, tree height, and LAI) differ between the forest types?
- Which forest type provides the highest phytodiversity and ES provision?
- How are the forest traits related to phytodiversity and ES provision?
- Do forests with high ES provision are also characterized by a high phytodiversity?
- Is the provision of ES more driven by complementarity or mass-ratio effects?

To assess the phytodiversity, a complete species and biotope mapping were performed considering all urban forests found in the study areas. As regulating ES, single tree carbon stocks and diurnal cooling were considered. Both services can act as a proxy for the regulating ES in forest stands because both services are associated with forest traits such as tree size, canopy size, and canopy density, which also control other regulating ES such as air pollution filtering and water retention (compare chapter 2.3 in the introduction). The findings from the carbon sequestration and microclimatic cooling experiments on the eleven monitoring sites shown in Part II and III were used to assess the carbon storage and microclimatic cooling of each biotope. Finally, both assessments were brought together. To analyze the phytodiversity-ES relationship, a trait-based approach was used. The influence of the forest traits on phytodiversity and ES provision were analyzed separately and compared if forest traits influence both ES and phytodiversity.

# 3 Materials and methods

## 3.1 Biotope and species mapping

At the study areas (introduction – chapter 2.5), all forest biotopes were mapped between June and September 2017. The forest biotopes were recognized by homogenous tree species composition and tree sizes (DBH and tree height) and were delineated against adjacent forest biotopes with different species composition and tree sizes. Because urban forests in the region have variable sizes and even small forest patches are defined as those (compare the definition of urban forests in Part I - chapter 1), a minimum biotope size was not defined. The biotopes were named following the reference list of biotopes provided by LANUV NRW (2018a) and each biotope was assigned to one of the three forest types following the definitions in chapter 2.2 of the introduction.

All vascular plant species growing in each biotope were listed and their coverage and distribution were rated in the following abundance classes (LANUV NRW 2019):

- *d* (dominant): species is distributed uniformly and covers more than 25 % of the biotope area
- *dl* (dominant-local): species is distributed partly and covers more than 25 % of the biotope area
- *f* (frequent): species is distributed uniformly and covers less than 25 % of the biotope area
- *fl* (frequent-local): species is distributed partly and covers less than 25 % of the biotope area
- *r* (rare): small number of individuals which only cover less than 1 % of the biotope area

Furthermore, DBH was categorized for each biotope in following DBH classes, according to LANUV NRW (2019):

- < 7cm
- 7-14 cm
- 14-38 cm
- 38-50 cm
- 50-80 cm
- 80-100 cm

The biotope and species mapping were supported by HOMM (2018) and SEILING (2018).

## 3.2 Calculation of tree height and LAI

Mean tree height of each biotope was obtained from the digital normalized surface model sampled in 2018 by GEOBASIS NRW. The surface model provides object heights with a precision of ± 5 dm. Zonal statistics in QGIS 3.16.6 were used to calculate mean tree height in each biotope using the digitalized biotope boundaries.

LAI during the vegetation period 2018 (April-September) was calculated from 19 cloudless and suitable satellite images from the Sentinel-2 platform. The images were resampled and resized on the study areas, and the LAI was computed using the Biophysical Processor (WEISS & BARET 2016) integrated in the Sentinel Application Platform (ESA SNAP Version 7.0). To calculate the mean LAI for each biotope, the QGIS (3.16.6) zonal statistics tool was used. As already shown in the carbon sequestration experiment, the algorithm of WEISS & BARET (2016) underestimates the LAI from remote sensing in forests in comparison to field measurements (compare Part II – chapter 4.5.1), the LAI was corrected using the formula from Part II – chapter 4.5.1 for the different forest types:

$$LAI_{corr} = (a * LAI + b)^2 \quad [10]$$

where $LAI_{corr}$ is the leaf area index after the correction, $LAI$ is the leaf area index obtained from the satellite images, and $a$ and $b$ are parameters listed in Table 19 for different forest types.

*Table 19: Parameters for the LAI correction introduced by the carbon sequestration experiment (Part II – chapter 4.5.1) for the different forest types.*

| **Forest type** | **a [-]** | **b [-]** |
|---|---|---|
| Semi-natural forests | 0.442 | 0.842 |
| Succession forests | 0.477 | 0.589 |
| Urban greening forests | 0.437 | 0.694 |

## 3.3 Floristic and ecological characterization of the forest biotopes

To measure how far the species composition differs between the forest types, the turnover rate was calculated. The turnover rate measures the changes in species composition typically along spatial or temporal gradients (TREMP 2005). Therefore, it is a measure for the dissimilarity of species composition and the opposite of the similarity index of SØRENSEN (1948). The turnover rate was calculated by

$$TR = \frac{A_{n,m} + B_{n,m}}{n+m} \quad [11]$$

where $TR$ is the turnover rate, $A_{n,m}$ is the number of species gained by the transition from group 1 to 2, $B_{n,m}$ is the number of species lost by the transition from group 1 to group 2, $n$ is the number of species in group 1, and $m$ is the number of species in group 2.

To analyze how the vegetation differs between the forest types, following floristic and ecological attributes were used:

- ELLENBERG'S indicator values for light (*L*), temperature (*T*), soil moisture (*F*), soil acidity (*R*), and soil nitrogen content (*N*) (ELLENBERG et al. 1992)
- forest affinity of species according to SCHMIDT et al. (2011)
- leaf anatomy according to KLOTZ et al. (2002)
- urbanity according to WITTIG et al. (1985)
- life forms according to RAUNKIAER (1910)
- competitive strategy types according to GRIME (1979)

The indicator values by ELLENBERG et al. (1992) rank the species in Central Europe by their ecological optimum, regarding different environmental factors such as light availability, temperature, moisture, acidity, or nitrogen in an ordinal scale between 1 and 9. With those indicators, the shaping of the environmental factors can be estimated by assigning the indicator values to the single species found in the biotope and calculating the mean over all indicator values for each biotope.

SCHMIDT et al. (2011) assessed a wide range of plant species, mosses, and lichens in Germany by their affinity to forests. They differentiated between three groups: i) species largely restricted to forests, ii) species occur in forest and open lands, and iii) open land species. The groups are divided in subgroups, where in the first group species were differentiated between i-i) species largely restricted to forests and i-ii) species with preference to forest edges and clearings. In the second group, species were differentiated between ii-i) species occurring in forest and open land and ii-ii) species may occur in forests but prefer open land. To visualize the spectrum of forest affinity at the forest types, the species were ranked by their forest affinity following Table 20.

*Table 20: Ranking of forest affinity (SCHMIDT et al. 2011).*

| **Groups of forest affinity** | **Subgroups** | **Forest affinity** | **Rank** |
|---|---|---|---|
| Species largely restricted to forests | Species largely restricted to forests | very strong | 1 |
| | Species with preference to forest edges and clearings | strong | 2 |
| Species occur in forest and open lands | Species occurring in forest and open land | moderate | 3 |
| | Species may occur in forests but prefer open land | weak | 4 |
| Open land species | Open land species | no affinity | 5 |

Leaf anatomy is categorized in i) heliomorphic leaves with adaptions to open water or high soil moisture in swamp or marshes, ii) hygromorphic leaves with adaptions to high humidity, iii) skleromorphic leaves with the ability to restrict transpiration as an adaption to areas with (partly) low water availability and high solar radiation, and iv) succulent leaves with structures to retain water as an adaption to dry conditions. Leaves with no special adaptions to water shortage and surplus are defined as v) mesomorphic leaves. Transitions between the leaf anatomy types are possible and common.

WITTIG et al. (1985) assessed a wide range of species in Germany to their affinity to urban habitats. They differentiate between i) urbanophobic species only found in rural areas, ii) moderate urbanophobic species mostly found in rural areas, iii) urbanneutral species found in urban habitats as well as in rural areas, iv) moderate urbanophilic species found mostly in urban areas, and v) urbanophilic species only found in urban areas.

The life forms by RAUNKIAER (1910) give information on species persistence in periods with unfavorable conditions (e.g., during winter) based on the location of the renewal organs relatively to the ground. Annual plants which persist as seeds are called therophytes, geophytes persist in rhizomes or in bulbs in the soil, hemicryptophytes persist near the soil surface, chamaephytes are small shrubs smaller than 50 cm, nanophanerophytes are shrubs and small trees between 0.5 to 5 m, and phanerophytes are trees larger than 5 m.

The competitive strategy types give information about how species reproduce and maintain in the landscape against other species. The concept by GRIME (1979) considers three types: i) competitors (C) are characterized by high growth rate, productivity, and flexibility in morphology with an optimal usage of the given resources. This type is typical for trees and shrubs but also for numerous herbs specialized to habitats with low stress conditions and low disturbance. ii) Stress tolerators (S) are species with adaption to habitats with unfavorable conditions such as extreme pH levels or high salt content. iii) Ruderals (R) are adapted to habitats with high intensity of disturbance. They are often annuals with a short life cycle and a high seed production. Not all species can be categorized to only one strategy type and transition types (CR, SR, CS) are common. Many species also have attributes from all three types. These are categorized as intermediary type (CSR).

The attributes were assigned to the species and the proportion of each category was calculated for the species community found in each forest type.

## 3.4 Measuring phytodiversity

To measure phytodiversity, following indicators were used:

- Shannon-Index (*H’*) (SHANNON & WEAVER 1949)
- Evenness (*E*) (PIELOU 1974)
- Rank abundance curves (WHITTAKER 1972)
- Species-area-relationships

Originally developed from information theory, the Shannon Index (*H’*) has been raised up to one of the most prominent diversity indices in ecology. *H’* measures the uncertainty in occurrence of one specific species in a random sample (MÜHLENBERG 1993). High uncertainty is equated to high diversity because the more species in a habitat occur, the higher is the uncertainty to sample a specific species. If a community consist only of one species, no uncertainty exists and *H’* would be zero. *H’* increases with the number of species sampled, but for biological systems *H’* normally does not exceed a value of five (KREBS 2014). The main advantage of *H’* is that the measure considers both, species number and species abundance, in one indicator. *H’* was calculated by

$$H' = \sum_{i=1}^{s} \frac{n_i}{n} * \ln \frac{n_i}{n} \quad [12]$$

where $H'$ is the Shannon Index, $s$ is the number of species in a sample, $n_i$ is the abundance ranking of species $i$ in a sample, and $n$ is the sum of abundance ranking of all species in a sample. To rate the abundance of species, the abundance classes from the biotope and species mapping (compare chapter 2.2) were ranked following Table 21.

*Table 21: Ranking of the abundance classes used to calculate H‘.*

| Abundance classes | Distribution | Coverage | Ranking |
|---|---|---|---|
| dominant (*d*) | uniformly | > 25 % | 5 |
| dominant-local (*dl*) | local | > 25 % | 4 |
| frequent (*f*) | uniformly | < 25 % | 3 |
| frequent-local (*fl*) | local | < 25 % | 2 |
| rare (*r*) | local or uniformly | < 1 % | 1 |

Species diversity has two components: species richness and species evenness (MARGALEF 1958). While $H'$ is a measure for species richness, species evenness ($E$) was calculated additionally by comparing $H'$ with the maximum value of $H'$ (PIELOU 1974, LENGDREN & LENGDREN 1998):

$$E = \frac{H'}{H'_{max}} \quad [13]$$

where $E$ is the evenness, $H'$ is the Shannon Index of the biotope, and $H'_{max}$ is the maximum Shannon Index when all species are equally represented.

Results for $H'$ and $E$ were aggregated for the forest types and as the data was normally distributed, ANOVA (FISHER 1973) and post-hoc test by TUKEY (1957) were performed in Rstudio version 1.4.1103 running with R version 4.0.3 (R Core Team 2020) to identify significant differences between the forest types. Significant level was defined by $p = 0.05$.

Species richness and species evenness for the forest types were additionally visualized in rank abundance curves (WHITTAKER 1972). To do so, the species occurring in the forest types were ordinated in a scatter plot with the species rank on the abscissa and the $\log_{10}$ of the relative abundance in the specific forest type on the ordinate. For the abscissa, the species were ranked by their descending frequency in the specific forest type. The rank abundance curves were calculated via Loess regression (CLEVELAND 1979) in Rstudio. Species richness is represented by the number of species on the abscissa, whereas species evenness is represented by the slope of the curves. A shallow gradient indicates that the species community is relatively even, where less dominant species are present in the forest type. Vice versa, a steep gradient indicates that the species community is relatively uneven, where a small number of species dominate the community (KREBS 2014).

Theoretically, the number of species tend to increase with the area sampled (KREBS 2014). However, *H’, E*, and the rank abundance curves measures phytodiversity regardless of the area occupied by the community. To analyze how the species number changes with increasing biotope area, the species-area-relationship was analyzed graphically by plotting the biotopes by their area on the abscissa and their number of species on the ordinate. Log regressions were calculated in Rstudio to create species-area-curves (KREBS 2014) for the forest types to compare the number of species at a given biotope size.

## 3.5 Estimating mean single tree carbon storage and diurnal cooling

The findings from the carbon sequestration experiment (Part II) and the microclimatic cooling experiment (Part III) were used to estimate tree carbon stocks and diurnal cooling. As shown in Part II – chapter 4.3, mean single tree carbon stock increases with increasing DBH and tree height. Two equations were introduced, one for DBH and one for tree height. Both equations were used to calculate single tree carbon stock and the mean of both were calculated by:

$$C = \frac{(0.0235411*DBH-0.0731494)^2+(0.031253*TH-0.197250)^2}{2} \qquad [14]$$

where *C* is the mean single tree carbon stock in the living biomass [t], *DBH* is mean diameter at breast height [cm], and *TH* is the mean tree height [m]. Because the DBH was sampled in DBH classes, mean DBH of each class were used for calculation.

Diurnal cooling, defined as the mean temperature difference at daytime between the forest and the urban climate, increases with increasing LAI and DBH, as shown in Part III – chapter 4.3. To estimate diurnal cooling, the formula introduced in Part III – chapter 4.3 was used:

$$\Delta T = 1.565872 - 0.567664 * LAI - 0.025625 * DBH \qquad [15]$$

where *ΔT* is the mean diurnal cooling [K], *LAI* is the mean leaf area Index, and *DBH* is mean diameter at breast height [cm].

Like for the results in phytodiversity, results for single tree carbon stock and diurnal cooling were aggregated for the forest types and significant differences were analyzed by ANOVA and Tuckey-post-hoc test (significant level of *p* = 0.05) (TUCKEY 1957).

## 3.6 Analyzing phytodiversity-ES relationship

To analyze phytodiversity-ES relationships, a trait-based approach was used. The forest traits DBH, tree height, and LAI were correlated with the ES provision (mean single tree carbon stock and diurnal cooling), as well as with the phytodiversity measures. Because the forest traits were used to calculate ES, it is hypothesized that ES provision will increase with enlarging forest traits. If both, ES and phytodiversity, increases with enlarging forest traits, complementarity effects are assumed because by enlarging forest traits, improvement in both, phytodiversity and ES, are expected. Vice versa, if phytodiversity do not increase with enlarging forest traits or no correlation can be found, the forest traits do not improve or have no influence on phytodiversity. In that case, mass-ratio effects are assumed, where ES provision is mainly controlled by single dominant tree species, regardless of the whole forest phytodiversity.

# 4 Results

## 4.1 Biotope and species mapping

In total, 338 forest biotopes were mapped at the study areas, whereby 92 biotopes were semi-natural, 145 were succession, and 101 were urban greening forests, respectively (Figure 48).

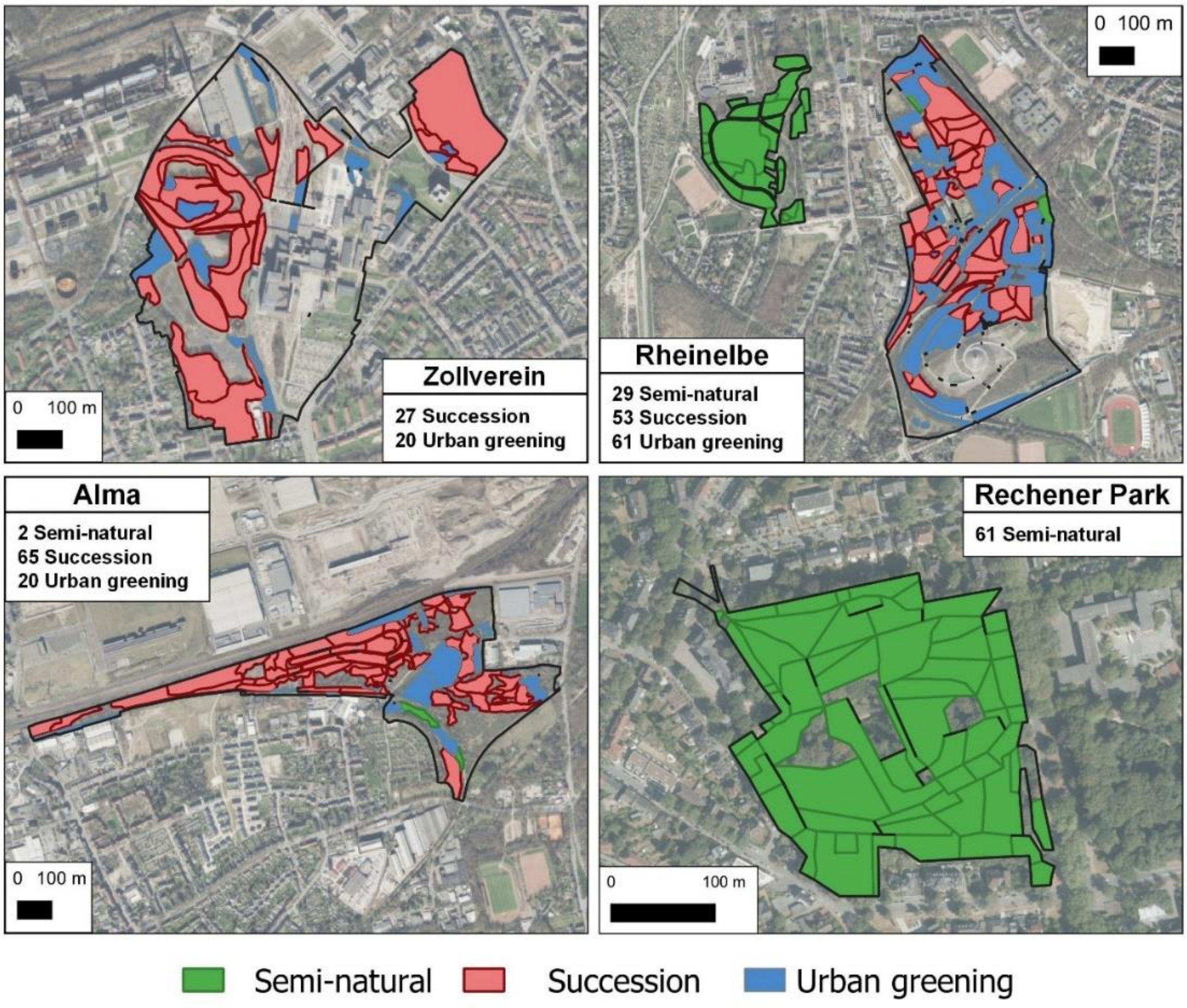


*Figure 48: Locations and sample sizes of the forest types in the study areas. Base map is provided by GEOBASIS NRW.*

*Table 22: Endangered species found in the different study areas and forest types, following Raabe et al. 2010.*
*Red lists: RL NRW = Red list of vascular plants for Northrhine-Westphalia, RL WB/WT = Regional red list for the Westphalian Lowland and Westphalian Bay, RL BRG = Regional red list for the Ruhr Metropolitan Region.*
*Red lists categories: * = not endangered, V = near threatened, 3 = vulnerable, 2 = endangered, R = rare, D = data deficient, - = not evaluated.*
*Study areas: RP = Rheinelbe Park, RecPa = Rechener Park, AL = Zeche Alma, RE = Halde Rheinelbe, ZZ = Zollverein.*

| **Species** | **Regional red lists** | | | **Forest types and study areas** | | |
|---|---|---|---|---|---|---|
| | **RL NRW** | **RL WB/WT** | **RL BRG** | **Semi-natural** | **Urban greening** | **Succession** |
| *Allium ursinum* | * | * | 3 | RP / RecPa | | |
| *Aquilegia vulgaris* agg. | 3 | 2 | - | RecPa | | |
| *Corydalis solida* | * | * | 3 | RP | | |
| *Equisetum sylvaticum* | * | 3 | 3 | RP | | |
| *Hyacinthoides non-scripta* | 3 | - | - | RP | | |
| *Ulmus glabra* | 3 | - | D | RP | | |
| *Polystichum aculeatum* | * | D | * | | AL | |
| *Ulmus minor* | 3 | - | D | | RE | |
| *Centaurium erythraea* | V | * | 3 | | | AL |
| *Dianthus armeria* | 3 | 3 | 3 | | | AL / ZZ |
| *Salix cinerea* agg. | * | * | D | RecPa | AL / RE | AL |
| *Taxus baccata* | 3 | - | - | RecPa / RP | RE | AL / ZZ / RE |
| *Cornus mas* | R | - | - | RecPa | ZZ | |
| *Convallaria majalis* | * | * | 3 | | RE | ZZ |
| *Festuca filiformis* | V | * | 3 | | ZZ | ZZ |
| *Galium saxatile* | * | * | 3 | | AL | AL |
| *Lotus corniculatus* | V | * | * | | AL | AL |
| | | | **Total** | **9** | **9** | **8** |

The autochthonous succession forests were characterized mostly by *Betula pendula* but also Willow- and Robinia-forests (*Salix spec., Robinia pseudoacacia*) were found. The investigated urban greening forests were characterized by different intermediate tree species (*Acer pseudoplatanus, A. platanoides, Alnus glutinosa, Fraxinus excelsior*, and others) which grew on anthropogenic soils. Forests on native soils characterized by *Fagus sylvatica, Quercus robur, Carpinus betulus, Acer pseudoplatanus*, and *Fraxinus excelsior* were categorized as semi-natural forests. At Rheinelbe Park and Rechener Park only semi-natural forests were found, while at Halde Rheinelbe, Alma, and Zollverein only a few biotopes could be described as semi-natural forests. At the brownfields mainly succession and urban greening forests occurred. The number of succession and urban greening forests at Halde Rheinelbe and Zollverein were more equal, while at Alma mainly succession forests were found.

In total, 327 different plant species were found. Most of them occurred in urban greening forests (225), followed by succession forests (209), and semi-natural forests (145). Mean sampled species number per biotope was largest at the succession forests (27), followed by urban greening (22), and semi-natural forests (21). Additionally, 17 endangered species were sampled, whereas nine were found at the semi-natural and succession forests and eight were found in the urban greening forests (Table 22). The origin of some endangered species might be from cultivations in adjacent gardens, where they escaped from garden waste dumped into the forests. This should be the case especially for *Aquilegia vulgaris agg., Hyacinthoides non-scripta*, and *Polystichum aculeatum*. Additionally, some trees and shrubs were cultivated in the forests for ornamental reasons (e.g., *Ulmus glabra, U. minor, Taxus baccata, Cornus mas*).

## 4.2 Floristic and ecological characterization

The urban greening and succession forests had a high similarity in species composition (Figure 49). 82 % of species were found in both forest types, whereas 18 % of species were found either in urban greening or in succession forests. However, the transitions from urban greening or succession forests to semi-natural forests are associated by large changes in species composition. Only half of the species found in semi-natural forests were also found in urban greening (47 %) and succession forests (49 %).

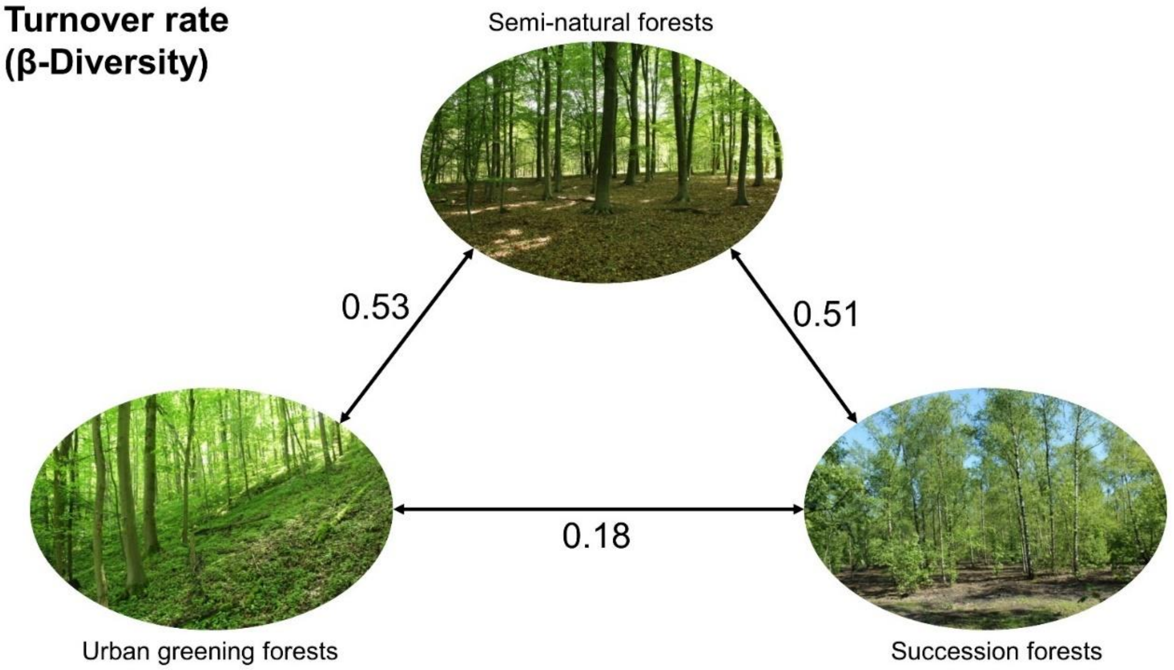


*Figure 49: Turnover rate (dissimilarity) between the different investigated forest types.*

Because of the similarity of urban greening and succession forests, no large differences were detected in the floristic and ecological characterization. Overall, these forest types were more characterized by open land or forest edge species (e.g., *Anagallis arvensis, Agrimonia eupatoria, Centaurium erythraea*) in comparison to semi-natural forests, where a higher proportion of typical forest species (e.g. *Anemone nemorosa, Allium ursinum, Polygonatum multiflorum*) were present (Figure 50, A). In detail, 37 % of species found in the semi-natural forests were largely restricted to forests and only 7 % of species were open land species. In comparison, 19 and 21 % of species were largely restricted to forests and 22 and 27 % of species were open land species in the succession and urban greening forests, respectively. Consequently, the indicator values for light and temperature differed between the forest types: Semi-natural forests were more characterized by species specialized to partly shaded and partly warm conditions (L = 5.5, T = 5) and succession and urban greening forests by partly sun-exposed and warm conditions (L = 7, T = 6). No differences were found for the indicator values of moisture, acidity, and nitrogen between the forest types. Depending on species composition, the investigated forests were characterized by mesophilic soil moisture (F = 5), weak basophilic (R = 7), and weak nitrophilic (N = 6) soils. However, a small peak for the succession and urban greening forests can be seen at a soil moisture value of 4, which indicate partly dry conditions (Figure 50, D).

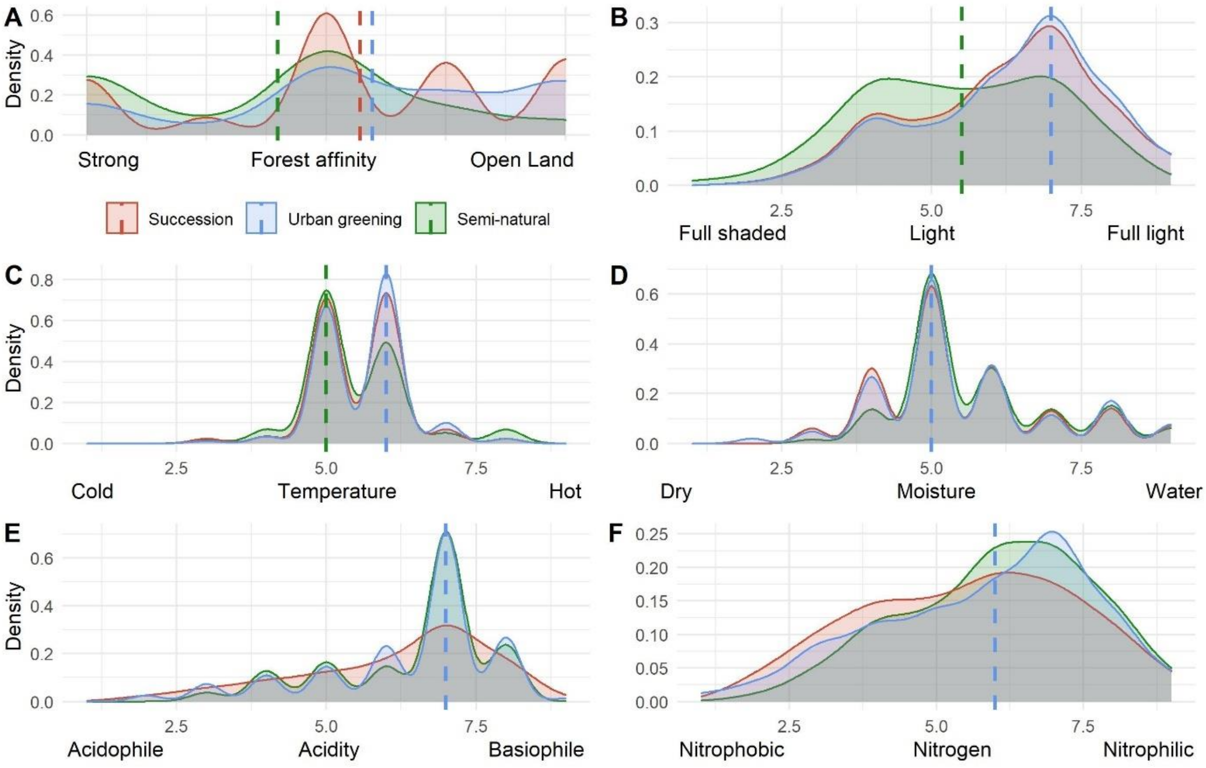


*Figure 50: Spectrum of forest affinity (SCHMIDT et al. 2011) (A) and of the ELLENBERG indicator values (ELLENBERG et al. 1992) for temperature (A), moisture (B), acidity (C), nitrogen (D), and light (E) aggregated for the different forest types. Dashed lines indicate the location of the means.*

The species attributes are shown in Figure 51. As can be seen, no great differences between the succession and urban greening forests in species composition regarding leaf anatomy, urbanity, life forms, and competitive strategies can be detected. Most species had mesomorphic leaves (49 - 51 %) showing no anatomical adaption on water deficiency or surplus, but more species had hygromorphic (25 - 27 %) than skleromorphic (18 - 20%) leaves, whereas only a small proportion of species had succulent (< 1%) and heliomorphic (4 - 6 %) leaves. More urbanophobic (53 - 56 %) than urbanophilic (8 - 10 %) species were found. However, 36 to 38 % of the species were urbanoneutral and had no preference for rural or urban habitats. Hemicryptophytes (42 - 43 %) were the most common lifeform in succession and urban greening forests, followed by phanerophytes (14 - 16 %), nanophanerophytes (15 - 16 %), and therophytes (13 - 15 %), whereas only a small proportion were geophytes (9 - 10 %), and chamaephytes (3 %). Competitors (70 - 74 %) were most common, whereas stress tolerators, and ruderals (7 - 8 %) were rarely present. 19 - 23 % of species belong to the intermediary strategy type (CSR).

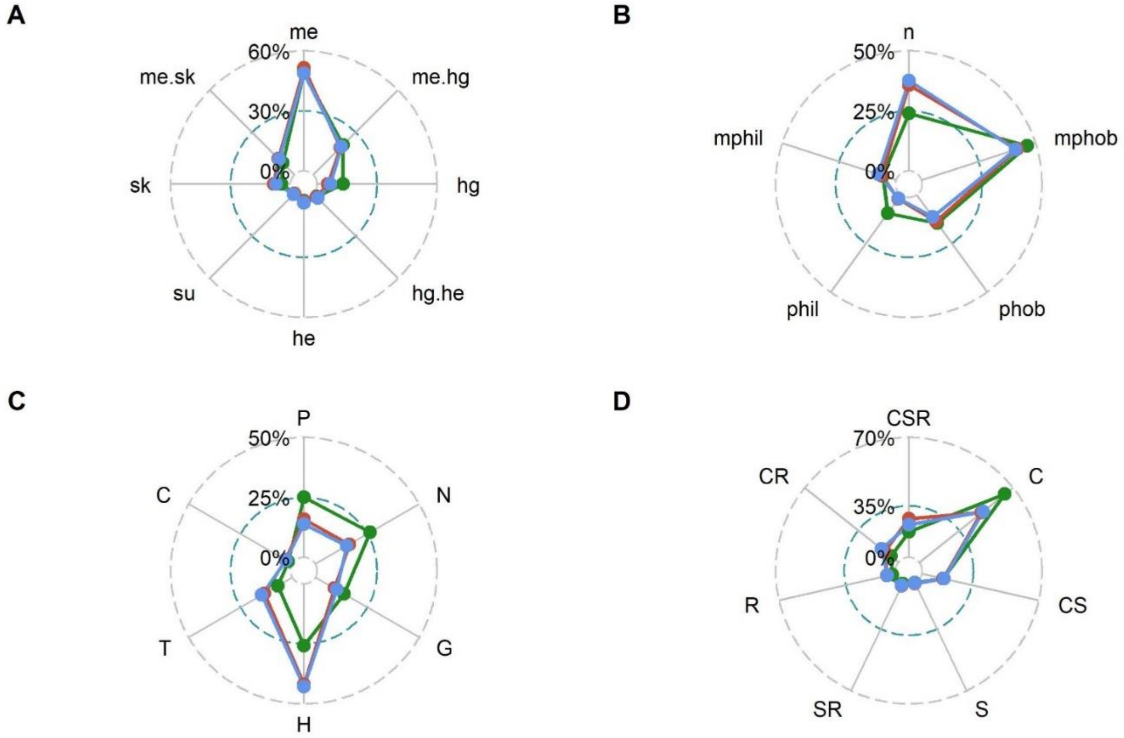


*Figure 51: Leaf anatomy, urbanity, life forms, and competitive strategies of species found in the different forest types. Green = semi-natural forests, red = succession forest, blue = urban greening forests.*
*A = leaf anatomy by Klotz et al. (2002), where me = mesomorphic, hg = hygromorphic, he = heliomorphic, su = succulence, sk = skleromorphic.*
*B = urbanity by Wittig et al. (1985), where n = neutral, mphob = moderate urbanophobic, phob = urbanophobic, phil = urbanophilic, mphil = moderate urbanophilic.*
*C = lifeforms by Raunkiaer (1910), where P = Phanerophyte, N = Nanophanerophyte, G = Geophyte, H = Hemicryptophyte, T = Therophyte, C = Chamaephyte.*
*D = competitive strategy types by Grime (1979), where C = competitors, S = stress tolerators, R = ruderals.*

Compared to the succession and urban greening forests, semi-natural forests were characterized by more species with hygromorphic leaves (34 %) and less species with skleropmorphic leaves (13 %), more urbanophobic (61 %) and urbanophilic (15 %) species, more phanerophytes (25 %), nanophanerophytes (26 %), and geophytes (14 %) and less hemicryptophytes (26 %) and therophytes (7 %). More than 50 % of species located in the semi-natural forests were phanerophytes or nanophanerophytes. Furthermore, competitors were more common (82 %), whereas stress tolerators and ruderals (3 %) are less frequent, compared to succession and urban greening forests.

## 4.3 Phytodiversity

All measures shown in Figure 52 indicate that the succession forests were characterized by the highest phytodiversity. The succession forests had a significantly higher *H'* and *E* (3.1 ± 0.04 and 0.54 ± 0.01) compared to the other types, whereas no significant differences for *H'* and *E* can be seen between the semi-natural (2.9 ± 0.03 and 0.5 ± 0.01) and urban greening forests (2.8 ± 0.06 and 0.49 ± 0.01). The species-area-relationship also indicates a higher species richness per area for the succession compared to the other forest types. Looking at the curves in Figure 52 C, the urban greening and semi-natural forests show the same pattern, whereas the succession forests is parallel above the other ones with an offset of about 5 species per m².

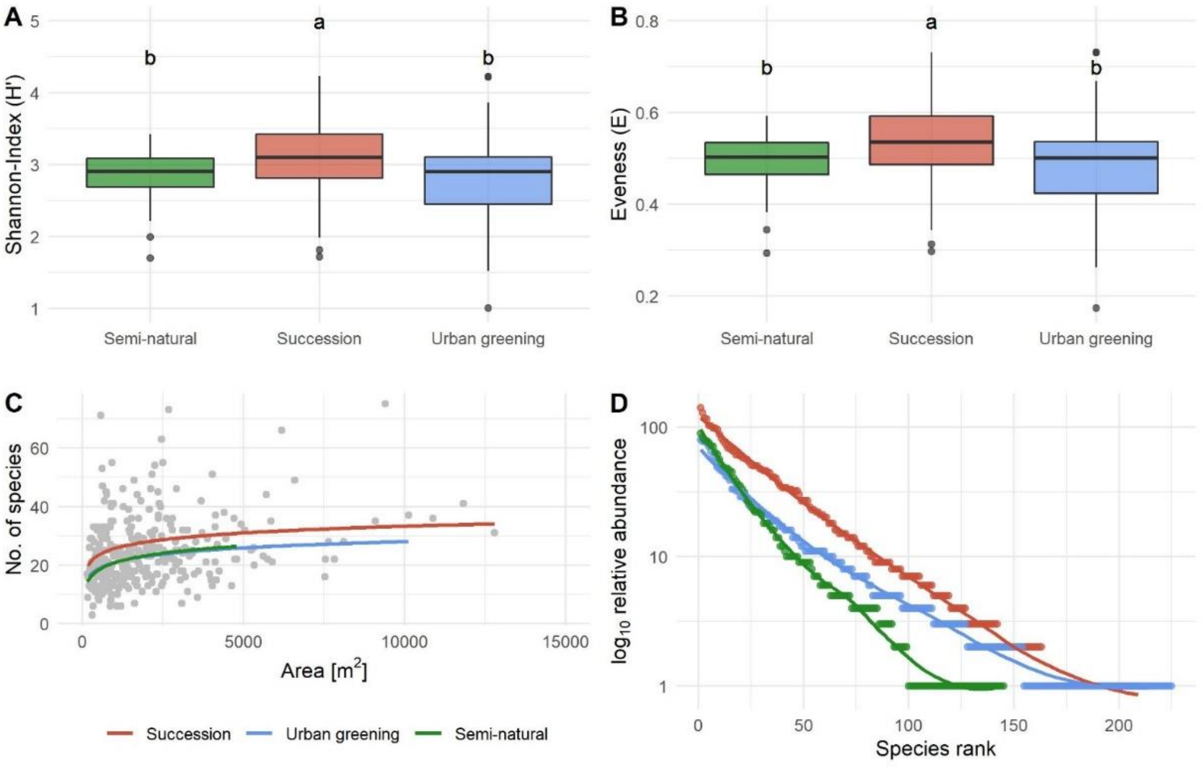


*Figure 52: Shannon-Index (H') (A), Evenness (E) (B), species-area-relationship (C), and rank abundance curves (D) for the investigated urban forest types. Small letters show significant differences (ANOVA).*

The rank abundance curves in Figure 52 (D) show largest species number for the urban greening forests but the gradient of the curve representing the succession forests is shallower, indicating that there were less dominant species and that the species composition was more even compared to the urban greening forests. The semi-natural forests were characterized by the lowest species richness and evenness. The slope of the gradient for the semi-natural forest is steep compared to the others, indicating that in semi-natural forests more dominant species occurred.

Because the results of all measures showed the same ranking for phytodiversity, only *H'* were used for further analysis. To investigate the influence of species composition on phytodiversity, *H'* of each biotope was correlated with the different species attributes in Figure 53. As can be seen, *H'* was positively correlated with the proportion of species with weak and no forest affinity and the proportion of geophytes, therophytes, and hemicryptophytes. Vice versa, *H'* was negatively correlated with the proportion of species with very strong to moderate forest affinity and with the proportion of phanerophytes, nanophanerophytes, and chamaephytes. All other attributes showed no correlation with *H'*.

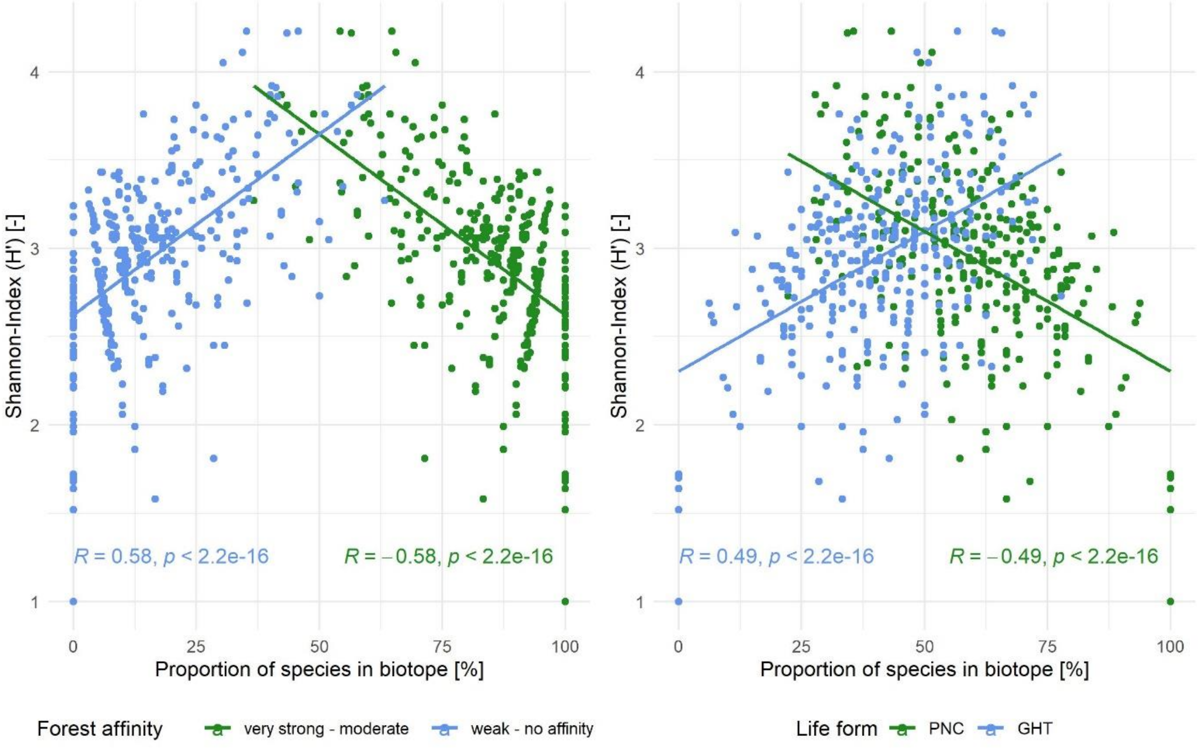


*Figure 53: Correlation analysis between H‘ and the composition of forest affinity and life forms considering all investigated biotopes. PNC = Phanerophytes, Nanophanerophytes, and Chamaephytes. GHT = Geophytes, Hemicryptophytes, and Therophytes.*

## 4.4 Forest traits and provision of ES

The semi-natural forests were characterized by the significantly largest tree height (22.4 ± 0.5 m) and LAI (5.0 ± 0.1), while there were no significant differences in tree heights between the urban greening (12.2 ± 0.4 m) and succession forests (12.8 ± 0.3 m) (Figure 54). The LAI was slightly larger at the urban greening (3.8 ± 0.1) compared to the succession forests (3.7 ± 0.1).

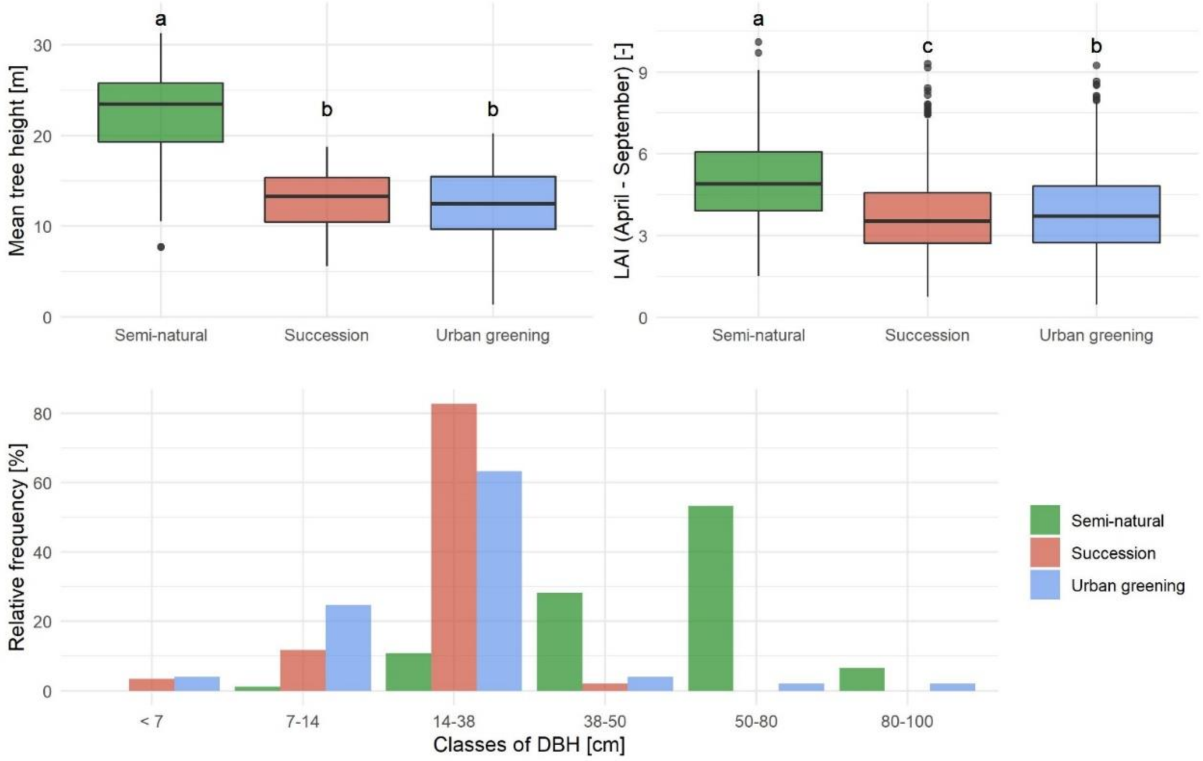


*Figure 54: Tree height, LAI during the vegetation period, and DBH found for the investigated forest types. Small letters show significant differences (ANOVA).*

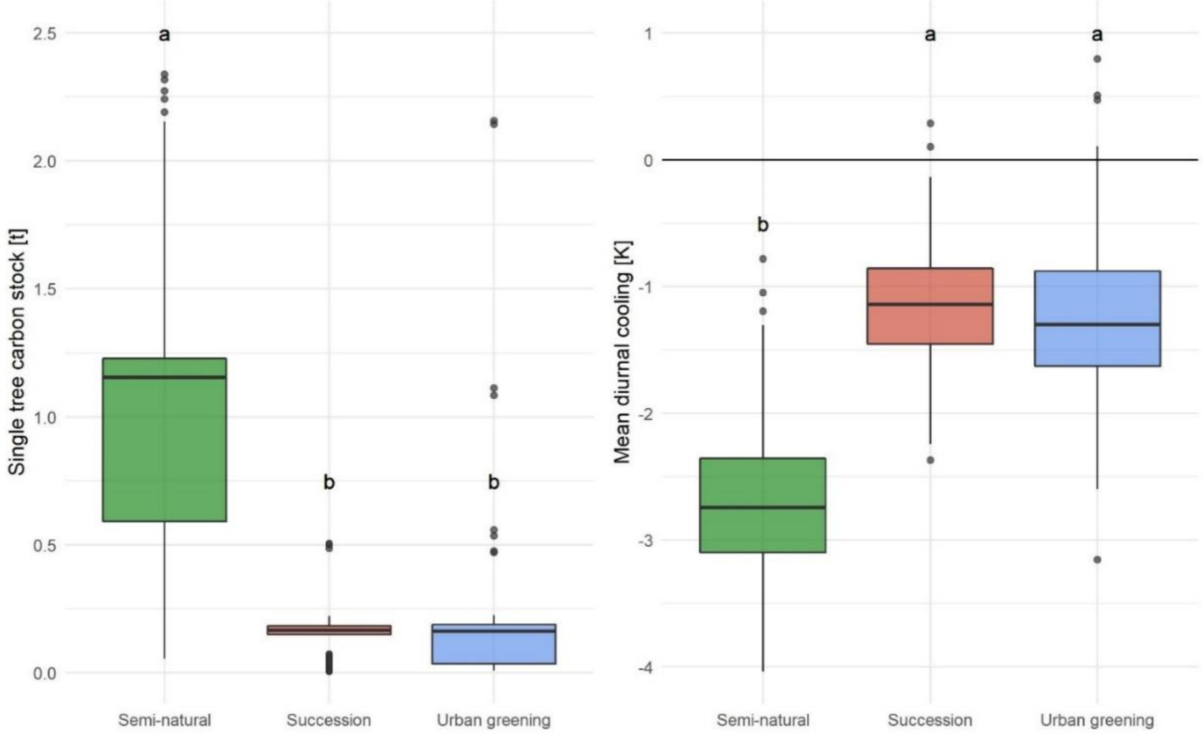


*Figure 55: Single tree carbon stock and mean diurnal cooling of the investigated forest types. Small letters show significant differences (ANOVA).*

60 % of the urban greening and 80 % of the succession forests belonged to the DBH class of 14 - 38 cm. Succession forests with a DBH greater than 50 cm were not part of the sample. Compared to the other forest types, the DBH of the semi-natural forests was noticeably larger.

Over 50 % of the semi-natural forests belonged to the DBH class of 50 - 80 cm, whereas semi-natural forests with DBH < 7 cm were not found during the sampling. Independently from the forest type, the sizes of the forest traits increased with increasing proportion of taller vegetation (phanerophytes, nanophanerophytes, and chamaephytes) found in the biotopes (not shown).

Tree size and leaf area shaped the ES provision. Consequently, semi-natural forests had the largest mean single tree carbon stock (0.99 ± 0.05 t C) and showed strongest diurnal cooling (-2.69 ± 0.07 K), while there were no significant differences in single tree carbon stock and diurnal cooling between the urban greening (0.2 ± 0.03 t C and -1.22 ± 0.06 K) and succession forests (0.16 ± 0.01 t C and -1.14 ± 0.04 K) (Figure 55).

## 4.5 Phytodiversity-ES relationships

In Figure 56, the influence of the different forest traits on phytodiversity and on the provision of ES is shown. As it was to be expected, the tree carbon stocks and diurnal cooling increased significantly with increasing DBH classes. Furthermore, tree carbon stocks and diurnal cooling was positively correlated with mean tree height and mean LAI. However, an opposite trend can be detected for the influence of the forest traits on *H’*. Hereby, lowest *H’* were found at DBH classes of < 14 cm and 80 - 100 cm, and *H’* is highest in forests with DBH between 14 - 38 cm. Furthermore, *H’* tend to decrease with increasing tree height and LAI.

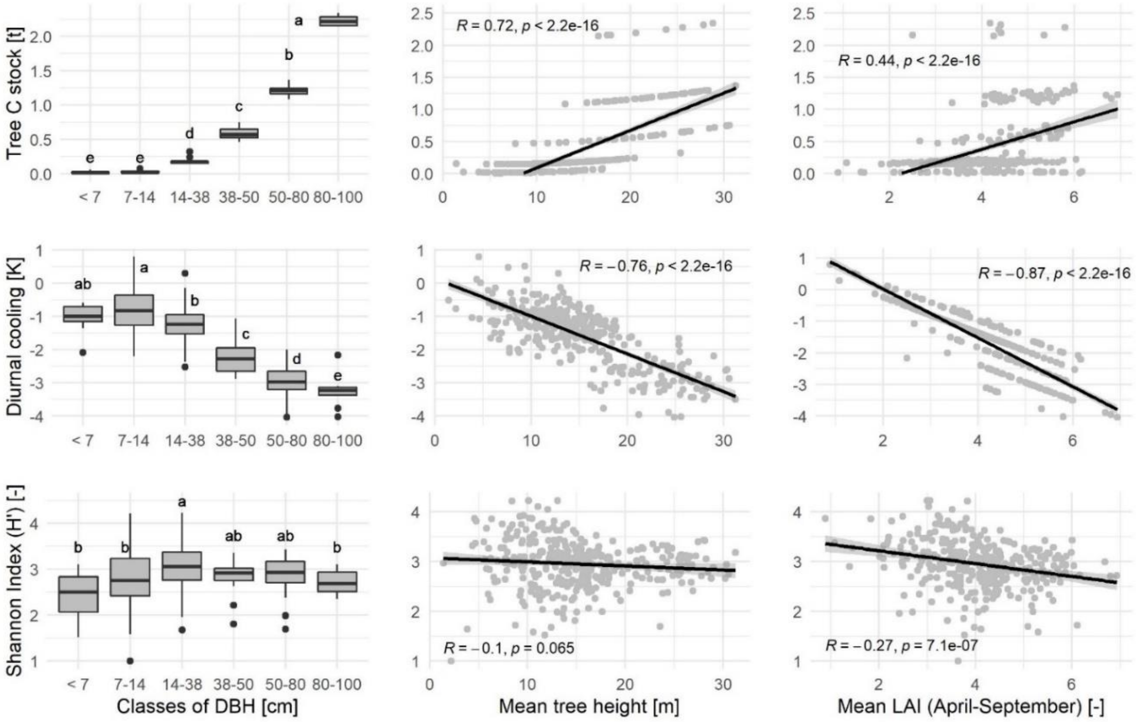


*Figure 56: Analysis of influences of forest traits (DBH, Tree height, LAI) on the provision of ES and on phytodiversity. Small letters show significant differences (ANOVA).*

The relationship between the forest traits and *H’* is analyzed in more detail in Figure 57. Overall, lowest *H’* were found in forests with smallest DBH classes (< 14 cm) and highest *H’* were reached in the DBH class 14 - 38 cm. However, in larger DBH classes (> 38 cm), *H’* slightly decreased compared to the highest class at DBH 14 - 38 cm. Nevertheless, second highest *H’* were found in the classes 38 - 50 and 50 - 80 cm, whereas forests belonging to the DBH class 80 - 100 cm showed *H’* again as low as in the smallest DBH classes < 14 cm. In Figure 57 C, the relationship of DBH and *H’* was analyzed in detail for the forest types. For the succession and urban greening forests, *H’* firstly increased over the DBH classes < 7, 7 - 14, and 14 - 38 cm, but decreased with larger DBH classes, as described earlier. This general pattern can also be detected for the semi-natural forests, where *H’* increased over the DBH classes 7 - 14, 14 - 38, 38 - 50, and 50 - 80 cm but started to decrease at larger DBH classes of 80 - 100 cm. Irrespectively of this pattern, no significant differences were found for *H’* for tree height classes (not shown). For LAI, *H’* was highest in small LAI classes (< 3.5) and decreased with larger LAI classes. In Figure 57 D, *H’* of the single biotopes was correlated separately with the mean LAI for the investigated forest types. Weak negative correlations were found for the succession and urban greening forests, indicating that *H’* decreased with increasing LAI. In contrast, *H’* increased with larger LAI in semi-natural forests but this weak relationship was not significant.

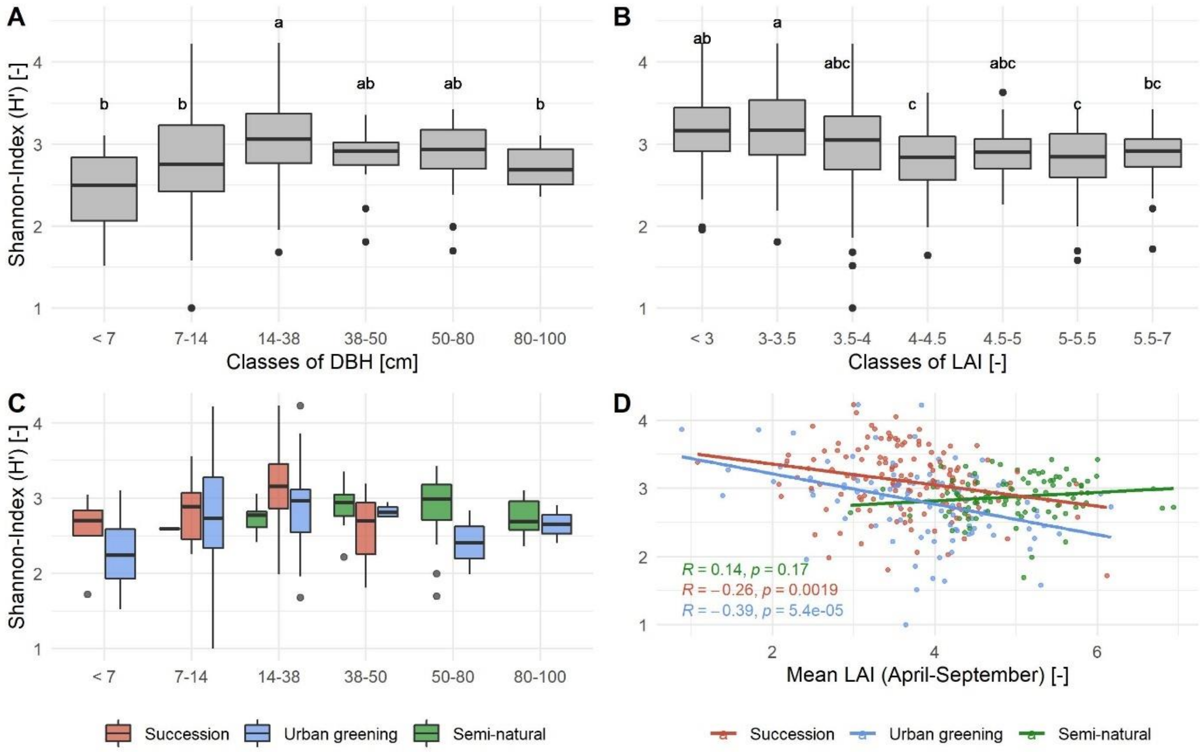


*Figure 57: Analysis of the phytodiversity-ES relationship in urban forests. Small letters show significant differences (ANOVA).*

# 5 Discussion

## 5.1 Differences in species composition between forest types

The calculated turnover rate indicated that the species composition between the urban greening and succession forests were relatively similar, but half of all species grown in the semi-natural forests were not found in the other forest types. Following main differences in species composition between semi-natural forests and the other forest types were found: i) more than half of the species (51 %) at the semi-natural forests were tall and small trees (phanerophytes and nanophanerophytes), whereas in the other forest types only 30 % of species belonged to these life forms, and ii) 40 % of the species found in the semi-natural forests were largely restricted to forests and only 7 % were open land species. At the other forest types, only 20 % are largely restricted to forests, whereas around 25 % are open land species. The high proportion of tall and small trees caused large and dense canopies in semi-natural forests, indicated by large LAI values, leading to low light conditions (SERCU et al. 2017) and a cooler and humid microclimate for the understory vegetation, indicated by a strong diurnal cooling. These features supported forest species adapted to low light and humid conditions and suppressed the development of open land species (VOCKENHUBER et al. 2011, BURKE et al. 2008). Consequently, more species with hygromorphic leaves were found in the semi-natural forests compared to the other forest types. Vice versa, less phanerophytes and nanophanerophytes grew in the urban greening and succession forests causing smaller and more open canopies, indicated by smaller LAI values. These features led to more light supply and a warmer and partly drier microclimate for the understory vegetation, indicated by a weaker diurnal cooling. Under these conditions, open land species were more supported, and species largely restricted to forests were suppressed. The higher proportion of open land species is reflected by the leaf anatomy, where more species had scleromorphic leaves, compared to the semi-natural forests, as well as by the ELLENBERG indicator value for light consumption.

Another explanation for the similarity in species composition between urban greening and succession forests and the dissimilarity between semi-natural and the other forest types might be that the semi-natural forests were not located in the same study areas, whereas urban greening and succession forests are located in the same areas. In general, each study area has its individual soil seed bank, and therefore, it is likely that the same species occur in both biotopes located close to each other compared to those located in different study areas.

Geophytes were more common in semi-natural compared to succession and urban greening forests. Most of them are vernal geophytes such as *Allium ursinum, Anemone nemorosa, Corydalis solida, Ficaria verna, Arum maculatum*, and others. In temperate climate, this life form is typical for nutrient-rich deciduous forests. Typically, geophytes use the short period between late winter and canopy closure in late spring for their lifecycle (POPOVIC et al. 2006). During that short period (normally in March and April), most of the sunlight reaches the understory vegetation and more nutrients (especially nitrogen) are available. Geophytes can use these resources quite efficient and can compete over hemicryptophytes, which have a longer life cycle (ELLENBERG & LEUCHNER 2010). In urban greening and succession forests, the canopy was more open and even in summer enough light could access the understory vegetation. These conditions were more favorable for hemicryptophytes, and therefore, less geophytes were found in these forest types.

The proportion of urbanophobic and urbanophilic species were larger in semi-natural than in the other forest types. Native forest species are typically not found in urban areas and because the proportion of typical forest species in semi-natural forests was larger, the proportion of urbanophobic species was larger, too. The larger proportion of urbanophilic species can be explained by the high number of shrubs and trees planted in parts of the Rechener Park for ornamental reasons such as *Acer japonicum, A. palmatum, A. saccharinum, Aucumba japonicum, Corylus colurna, Euonymus fortunei*, and others. Those species typically are not found in rural areas, and therefore, they were rated as urbanophilic species.

The composition of lifeforms at the forest types were in good agreement with the investigations of GAUSMANN (2012) and HETZEL (2012). GAUSMANN (2012) found, that succession forests are composed by 40 % hemicryptophytes and 32 % phanerophytes and nanophanerophytes, which is nearly equal to the findings of the presented study. For semi-natural urban forests, Hetzel (2012) found with 41 - 66 % even a larger proportion of nanophanerophytes and phanerophytes in semi-natural forests compared to the findings from the presented study, and a smaller proportion of hemicryptophytes (18 - 36 %).

## 5.2 Phytodiversity-ES relationship

As ES, only carbon storage and diurnal cooling were considered. However, other services like filtering of air pollution and rainwater retention are also associated with the forest structure described by DBH, tree height, and LAI (compare chapter 2.3 of the introduction). Therefore, it can be expected, that the provision of other regulating services will differ in the same manner shown for carbon storage and diurnal cooling and both services can act as proxys for the provision of regulating ES in urban forest systems.

The phytodiversity indicators clearly showed, that highest phytodiversity were found in the succession forests, whereas no significant differences were found between the urban greening and semi-natural forests. Only the rank abundance curves indicated that the urban greening forests had a higher species richness and evenness compared to semi-natural forests. The correlation of *H’* with the proportion of lifeforms and forest affinity showed that phytodiversity decreased with increasing proportion of taller vegetation (phanerophytes and nanophanerophytes) and with increasing proportion of species largely restricted to forests. Vice versa, forests with a high proportion of species in the herbaceous layer (hemicryptophytes, geophytes, and therophytes) and a high proportion of open land species tended to have a higher phytodiversity. This can be explained by the fact that in temperate deciduous forests most vascular plant species occur in the herbaceous layer (VOCKENHUBER et al. 2011, WHIGHAM 2004, GILLIAM 2007). Consequently, the number of herb species impact the phytodiversity indicators relatively strong. Besides that, also ecological reasons might explain these results. Phanerophytes and nanophanerophytes are competitors and by their growth and dense canopies they limit the resources, especially light availability, for the understory vegetation (GRIME 1979). This is supported by the finding, that the size of the forest traits (DBH, tree height, LAI) increased with the proportion of phanerophytes, nanophanerophytes, and chamaephytes in the biotopes, indicating an unbalanced usage of resources primarily by the taller vegetation. That explains the finding, that a higher proportion of trees in biotopes led to less species found in the understory vegetation and that the evenness of species decreased in the community. Vice versa, in forests with a smaller proportion of taller vegetation, the resources might be better shared between the species leading to a higher species richness and evenness.

But not only the proportion of taller vegetation negatively influenced H’. H’ also tended to decrease with increasing LAI, DBH, and tree height. The dense canopy of the semi-natural forests created low light conditions homogenously in the biotope, where only typical forest species adapted to low light and humid conditions were supported. In contrast, in the urban greening and especially in the succession forests, the canopy was more open and light availability was more heterogenous creating an ecotone, where forest, forest edge and open land species find their niche (VOCKENHUBER et al. 2011). This led to a more even plant community with a higher phytodiversity compared to semi-natural forests. Those findings, that low light conditions due to large canopy cover decreased the species richness in the herbaceous layer, were also found in many temperate forests in different European regions such as for the Hainich National Park in Thurinigia (VOCKENHUBER et al. 2011), for acidophilic beech and oak forests (*Quercetalia roboris*) in Schleswig-Holstein (HÄRDTLE et al. 2003), for oak-dominated forests in Central Bohemia (HOFMEISTER et al. 2009), or for deciduous-coniferous mixed forests in the Őrség region in Western Hungary (TINYA et al. 2009). GAUSMANN (2012) points out that the succession forests of the Ruhr Metropolitan Region are in terms of species composition unsaturated plant communities. The competition between the species is low, so that especially adventive species can niche into the forest community, leading to a high species richness and evenness.

However, the provision of regulating ES was associated to larger tree traits, and those tree traits were even larger, the higher the proportion of taller vegetation was. With larger DBH and tree height, more carbon was stored in the living biomass and the canopy was larger and denser, resulting in a stronger diurnal cooling. In general, trees are the main sinks for carbon in forest systems and only little carbon is stored in the understory vegetation (PAN et al. 2018). Additionally, filtering of air pollution or rainwater retention is even larger in forests with dense and large canopies (JANHÄLL 2015, KERMAVNAR & VILHAR 2017). For these ES, the semi-natural forests seem to be better suited compared to the other types as they are characterized by largest DBH, tree height, LAI, and largest proportion of tall and small trees.

Due to the relationships described before, no clear phytodiversity-ES relationship can be found for the urban forests in the Ruhr Metropolitan Region (Figure 58). Phytodiversity firstly increased with increasing DBH classes but in DBH classes larger than 38 cm phytodiversity decreased again. Phytodiversity was highest in small LAI classes (< 3.5) and decreased with increasing LAI classes. For the succession and urban greening forests, a weak negative correlation was found between H’ and LAI, indicating that phytodiversity decreased with increasing LAI. For the semi-natural forests, this correlation was positive. However, this relationship was not significant. Consequently, the hypothesis that a similarity between phytodiversity and ES provision exists in urban forests needs to be falsified. For the investigated forest ecosystems, it can be concluded that the provision of ES is driven by dominant tree species, while the phytodiversity of the whole stand has no influence or is even lower in forest stands with a high provision of ES. Therefore, mass-ratio theory is more supported than the niche-complementarity theory.

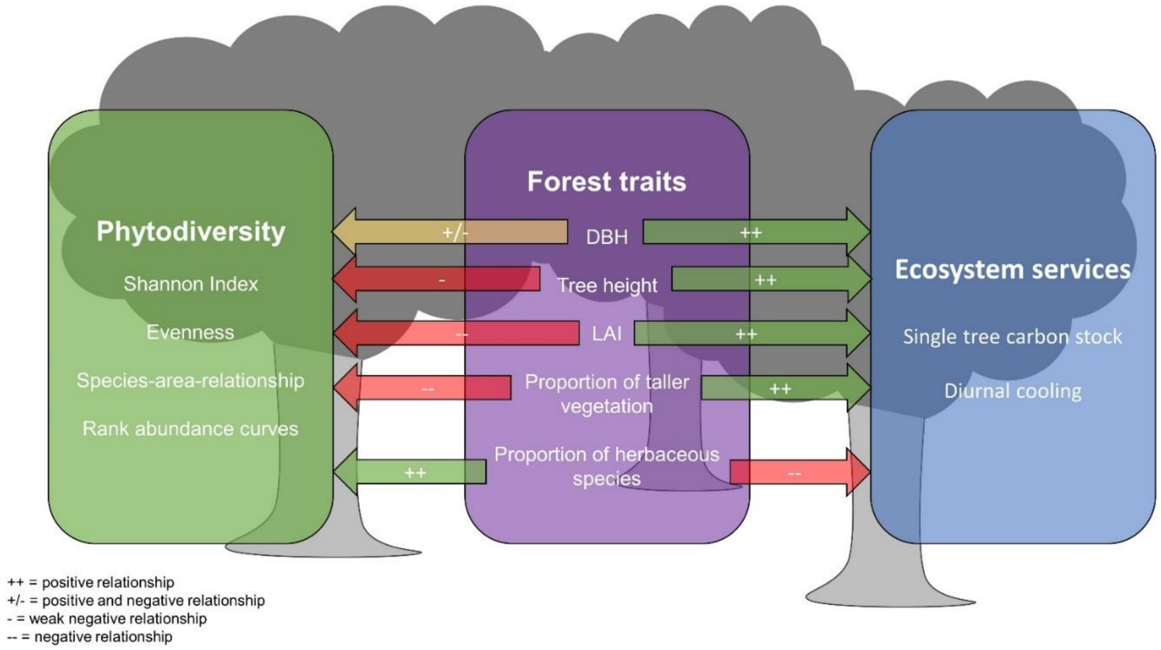


*Figure 58: Relationships between the forest traits and the phytodiversity and the ES provision in urban forests.*

# 6 Conclusion

The aim of the experiment was i) to analyze the differences in forest traits, phytodiversity, and ES provision between the different forest types and ii) to analyze the influence of forest traits on phytodiversity and ES provision to find out whether complementarity or mass-ratio effects can describe the phytodiversity-ES relationship in urban forests.

82 % of species found in the succession forests were also found in the urban greening forests, showing that both forest types did not differ strongly in species composition. However, semi-natural forests and the other forest types share only half of all species mapped during the study. Compared to the succession and urban greening forests, the species composition of semi-natural forests was more characterized by phanerophytes, nanophanerophytes, geophytes, more typical forest species, more hygromorphic species, more urbanophobic and urbanophilic species, and more species which are adapted to low light conditions.

The highest phytodiversity was found in the succession forests compared to the other forest types. No significant differences in the phytodiversity metrics were found between the urban greening and semi-natural forests, but the rank-abundance-curves indicated that the urban greening forests have a higher phytodiversity compared to the semi-natural forests. However, largest forest traits (DBH, tree height, and LAI) were found in the semi-natural forests, leading to strongest diurnal cooling and largest carbon stocks, whereas no significant differences were found in ES provision between urban greening and succession forests.

The size of the forest traits influences the ES provision positive but the phytodiversity negative. Therefore, no complementary effects between phytodiversity and ES provision were found for the investigated urban forests. In conclusion, ES provision in the investigated urban forests depended on mass-ratio effects where a few dominant species with large forest traits provide regulating ES regardless of the entire phytodiversity of the stand.

# 7 Implications for urban forestry and planning

The highest phytodiversity in urban forest ecosystems of the Ruhr Metropolitan Region was found in forests with small LAI values (< 3.5) and with a DBH between 14 and 38 cm. This forest structure tends to provide enough resources for a balanced community between higher vegetation (trees, and shrubs) and herbaceous species, which supports species richness and species evenness. However, this forest structure is not optimal for ES provision. In forests with low LAI values, small DBH, and small tree height diurnal cooling and carbon storage was rather small. Therefore, it can be recommended to preserve both forest types for different forest functions: semi-natural forests primarily for ES provision, and succession and urban greening forests for preservation of phytodiversity in urban areas. Furthermore, it should not be disregarded, that the species composition differed between semi-natural forests and succession, and urban greening forests. Therefore, even if semi-natural forests had a lower phytodiversity, they provide habitats for species not found at the other forest types. Consequently, the phytodiversity of the urban system benefits from all three forest types.

# Conclusion

The aim of the dissertation was to investigate the mechanisms of regulating ES provision of different urban forest types in the Ruhr Metropolitan Region by field experiments. As ES, $CO_2$ sequestration, and microclimatic cooling were considered, whereby both ES can be also used as proxies for other regulating ES. The experiments took place during a two-years lasting drought period. Therefore, it was possible to analyze how the different urban forest types react on the drought and how the ES provision changes under warm temperatures and water shortage. Furthermore, the phytodiversity (species richness and evenness) of the different urban forest types were investigated. In the last step, the influence of the phytodiversity on ES provision were analyzed to investigate if mass-ratio or complementarity effects describe the phytodiversity-ES-relationship in urban forests. In the following, the findings from the investigations are summarized and the suitability of the framework for ES assessment and the advantages and disadvantages of the approach are discussed.

## 1 ES provision of the urban forest types and influence of the drought period

From the findings of the field studies, it can be clearly concluded, that semi-natural forests are best in providing ES. Carbon storage and microclimatic cooling depends on the size of the tree traits (DBH, tree height, LAI, crown height, e.g.) and stand age. Here, the semi-natural forests have a clear advantage compared to urban greening and succession forests, which are typically younger in age with smaller sizes in tree traits.

However, annual $CO_2$ sequestration does not depend strongly on the sizes of the tree traits. In 2018, urban greening forests showed a slightly higher $CO_2$ sequestration compared to the semi-natural forests. This changed in 2019, when the $CO_2$ sequestration in semi-natural forests slightly increased and the urban greening forests drastically decreased the $CO_2$ sequestration compared to 2018. It was possible to identify the water shortage during the experimental period as a driver for these changes because the difference in $CO_2$ sequestration between both years can be explained by the depth of the effective rooting zone, the content of soil organic carbon, and the amount of plant available water. Those soil properties mainly control the amount and the accessibility of soil water. Urban greening and succession forests are limited in rooting due to sealed soil layers and the amount of plant available water and soil organic carbon is smaller which makes them more vulnerable to droughts. In comparison, semi-natural forests have no limitation in rooting and are characterized by higher amounts of soil organic carbon and plant-available water, which makes them more resilient against droughts. In summary, urban greening forests can sequester slightly more $CO_2$ per year compared to semi-natural forests, but semi-natural forests are less affected over longer times if water shortage due to dry spells occur.

On the other hand, not all semi-natural forests were equally resilient against drought. An investigated *Stellario-Carpinetum* with a water logging soil layer turned from a carbon sink in 2018 into a carbon source in 2019. Therefore, it must be specified that semi-natural forests on mesophilic locations are resilient, while semi-natural forests on more extreme locations are also vulnerable against water shortage.

Overall, succession forests provided less ES due to small tree traits, an open canopy, and unfavorable growth conditions on the technosols. Low amounts of $CO_2$ were sequestered by $CO_2$ assimilation and translocation, which faced relatively high $CO_2$ release by respiration due to high soil pH values. The open canopy, mostly build up by *Betula pendula*, blocked only little solar radiation, which led to smaller diurnal cooling effects. Referring to KOWARIK et al. (2017), it can be concluded that the provision of ES is not equal between the forest types.

The drought clearly affected the urban forest vitality and the ES provision. It was shown that nine of eleven forests studied drastically decreased $CO_2$ sequestration in the second drought year compared to the first year. Two forests even turned from carbon sinks into sources in the second drought year. Due to expected climate change, drought periods will increase in length, intensity, and frequency. Urban greening, succession, and semi-natural forests on extreme locations are not well adapted to these changes. How the drought affected microclimatic cooling was not directly investigated, but it was shown that the microclimatic cooling depends, among others, on the number of leaves and the canopy density. The number of leaves was also reduced in the second year at all sites. This implies that also microclimatic cooling will be lower during drought periods. Consequently, it is expected that forests will decrease their provision of ES under future climate. To prevent this scenario, it can be recommended to adapt urban forests to drought periods. Here, good soil conditions are crucial to improve urban forest resilience to periods with water shortage.

## 2 Phytodiversity of urban forest types and phytodiversity-ES relationship

The highest phytodiversity (species richness and evenness) was found in the succession forests, followed by the urban greening forests. Semi-natural forests had the lowest phytodiversity. Species richness and evenness in the investigated forests depended on the ratio between taller vegetation (trees and shrubs) and herbs found in a biotope. Highest phytodiversity was found in biotopes where the ratio between taller vegetation and herbs are even and phytodiversity decreased with increasing proportion of taller vegetation. In addition, the tree traits influenced phytodiversity, where highest phytodiversity was found in forests with open canopies (low LAI) and a DBH between 14 and 38 cm. These characteristics were more common in succession forests, while semi-natural forests are characterized by closed canopies and large DBH. It is likely that the open canopies in succession forests led to more small-scaled habitats and a better sharing of resources, which promotes the occurrence of various specialized herb species. In semi-natural forests, the allocation of resources might be unbalanced due to a small species number of large trees, which restrains the number of herb species. However, only half of the species found in the succession and urban greening forests were also found in the semi-natural forests. This indicates that even if semi-natural forests have a lower phytodiversity, they are characterized by an individual species composition not found in the other forest types. Consequently, the urban system benefits from all three urban forest types.

The sizes of the tree traits increased ES provision but decreased phytodiversity. Therefore, it can be concluded that mass-ratio effects are more suited to describe the phytodiversity-ES-relationship of urban forests, where a small number of trees with large traits provide the regulating ES regardless of the phytodiversity of the entire forest.

## 3 Evaluation of the framework

The framework presented in this dissertation focused on measuring ES in the field rather than using modelling, literature reviews, or indicators. The results of the experimental approach provided reliable results of the ES provision of different urban forest types and the mechanisms behind the provision were identified. Overall, the results presented have implications for urban forestry and urban planning in terms of improving ES and phytodiversity for urban forests especially in the face of the expected changing climate.

The new framework introduced shows clearly advantages as direct field surveys are used to determine ES precisely and factors influencing the ES provision can be analyzed. Based on this approach, it was possible to investigate the effect of the consecutive two years drought period, which led to a decrease in ES provision. As the classical approaches to estimate ES are mainly based on assumptions, such climatic extremes are typically not considered. Therefore, the effects of the drought might not be detected by the classical approaches. Additionally, those simpler approaches are also not suitable to analyze the mechanisms behind ES provision.

A clear disadvantage of the experimental based approach to determine ES is the large effort needed to measure ES in the field. At least one year of observation is required to estimate $CO_2$ sequestration and microclimatic cooling. Additionally, the results obtained are only valid for the specific year and the services can easily change in other years with different environmental conditions. Due to this fact, the results from one year cannot be generalized, and therefore, the field surveys should span over several years to generate reliable data. Furthermore, field measurements are not only time-consuming but also expensive in material and manpower. The basic research on ES is very complex, and therefore, it was not possible to consider all relevant ES provided by urban forests in depth and only a few study areas were analyzed, which restrict generalization. Here, the classical approaches have an advantage because the effort is relatively low. Larger study areas can be processed, and several ES can be considered, but not in their whole complexity and the bias might be higher compared to field surveys.

Scientists and practitioners currently work on implementing ES in urban planning processes and decision-making (GRUNEWALD et al. 2021). When it comes to decision-making, the ES estimation must be as precise as possible, whereby the field study-based approach has a lower bias compared to the classical approaches. On the other hand, the greater financial and technical effort of field studies may hamper its application. Furthermore, models are easier to integrate in planning processes. Therefore, regional field surveys, such as the presented work, should be used to investigate the regional mechanisms of ES provision of urban green and blue infrastructure. From the findings of the regional basic research, regional models can be created, or existing models can be adapted to the region’s specifics. Finally, the regional models can then be used in urban planning and decision-making.

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# Appendix

*Table A.1: Coefficient for predicting bw-w-ratio depending on tree species and DBH.*

| DBH | a | b | DBH | a | b | DBH | a | b | DBH | a | b | DBH | a | b |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| [cm] | [-] | [-] | [cm] | [-] | [-] | [cm] | [-] | [-] | [cm] | [-] | [-] | [cm] | [-] | [-] |
| ***Betula pendula*** | | | ***Alnus glutinosa*** | | | ***Fagus sylvatica*** | | | ***Quercus robur*** | | | ***Other*** | | |
| 7 | 1,0353 | -0,281 | 7 | 10,498 | -1,278 | 6 | 0,9616 | -0,032 | 7 | 3,9405 | -0,863 | 7 | 2,0113 | -0,494 |
| 8 | 3,1753 | -0,838 | 8 | 0,9146 | -0,406 | 7 | 0,8305 | -0,027 | 8 | 5,3119 | -1,048 | 8 | 1,2802 | -0,414 |
| 8,5 | 11,975 | -1,413 | 8,5 | 0,7881 | -0,383 | 8 | 0,8303 | -0,162 | 8,5 | 5,3985 | -1,076 | 8,5 | 2,4743 | -0,689 |
| 9 | 10,976 | -1,348 | 9 | 0,6425 | -0,338 | 9 | 0,8844 | -0,246 | 9 | 6,6654 | -1,187 | 9 | 3,2702 | -0,829 |
| 10 | 3,5666 | -0,916 | 10 | 1,7602 | -0,806 | 10 | 1,5502 | -0,565 | 10 | 7,8494 | -1,3 | 10 | 2,7796 | -0,852 |
| 11 | 5,8024 | -1,148 | 11 | 0,9394 | -0,626 | 11 | 2,4734 | -0,806 | 11 | 5,4079 | -1,161 | 11 | 2,7625 | -0,887 |
| 12 | 11,635 | -1,473 | 12 | 1,7155 | -0,907 | 12 | 3,8641 | -1,01 | 12 | 4,7035 | -1,132 | 12 | 3,6362 | -1,025 |
| 13 | 2,5325 | -0,894 | 13 | 1,1303 | -0,811 | 13 | 5,1714 | -1,147 | 13 | 2,9446 | -0,966 | 13 | 3,692 | -1,052 |
| 14 | 1,6194 | -0,803 | 14 | 2,339 | -1,157 | 14 | 6,2185 | -1,227 | 14 | 2,3053 | -0,913 | 14 | 3,9045 | -1,097 |
| 15 | 1,4727 | -0,718 | 15 | 0,8911 | -0,847 | 15 | 7,8565 | -1,316 | 15 | 1,5627 | -0,79 | 15 | 3,5687 | -1,067 |
| 16 | 1,048 | -0,598 | 16 | 1,0165 | -0,925 | 16 | 8,5711 | -1,345 | 16 | 1,1795 | -0,714 | 16 | 3,0495 | -1,02 |
| 17 | 2,2458 | -0,854 | 17 | 0,4067 | -0,587 | 17 | 9,5932 | -1,38 | 17 | 0,8696 | -0,619 | 17 | 2,3808 | -0,94 |
| 18 | 1,8604 | -0,794 | 18 | 0,3916 | -0,562 | 18 | 8,7991 | -1,35 | 18 | 0,8796 | -0,643 | 18 | 1,9198 | -0,879 |
| 19 | 1,3745 | -0,688 | 19 | 0,722 | -0,749 | 19 | 9,2758 | -1,367 | 19 | 0,9999 | -0,698 | 19 | 1,9873 | -0,897 |
| 20 | 0,299 | -0,189 | 20 | 0,3366 | -0,507 | 20 | 9,2503 | -1,357 | 20 | 1,3511 | -0,82 | 20 | 1,5808 | -0,818 |
| 21 | 0,7201 | -0,483 | 21 | 0,5146 | -0,645 | 21 | 10,01 | -1,385 | 21 | 1,285 | -0,806 | 21 | 1,907 | -0,889 |
| 22 | 0,5832 | -0,419 | 22 | 0,7329 | -0,75 | 22 | 9,4655 | -1,363 | 22 | 1,3172 | -0,821 | 22 | 1,8542 | -0,879 |
| 23 | 0,434 | -0,329 | 23 | 0,7771 | -0,763 | 23 | 8,1341 | -1,305 | 23 | 1,0202 | -0,735 | 23 | 1,9456 | -0,887 |
| 24 | 0,3539 | -0,276 | 24 | 0,5141 | -0,629 | 24 | 7,1585 | -1,258 | 24 | 1,0304 | -0,743 | 24 | 1,4609 | -0,796 |
| 25 | 0,2764 | -0,206 | 25 | 0,7778 | -0,76 | 25 | 5,703 | -1,18 | 25 | 0,9998 | -0,727 | 25 | 1,0102 | -0,678 |
| 26 | 0,2565 | -0,197 | 26 | 0,5566 | -0,654 | 26 | 4,9206 | -1,127 | 26 | 0,8226 | -0,661 | 26 | 0,9741 | -0,663 |
| 27 | 0,3996 | -0,349 | 27 | 0,2379 | -0,385 | 27 | 4,5726 | -1,101 | 27 | 0,8985 | -0,686 | 27 | 1,1938 | -0,726 |
| 28 | 0,2763 | -0,238 | 28 | 0,1341 | -0,209 | 28 | 4,6176 | -1,097 | 28 | 0,9637 | -0,711 | 28 | 1,0636 | -0,687 |
| 29 | 0,2576 | -0,226 | 29 | 0,3917 | -0,543 | 29 | 4,1025 | -1,051 | 29 | 1,1811 | -0,783 | 29 | 1,0101 | -0,668 |
| 30 | 0,2892 | -0,273 | 30 | 0,37 | -0,52 | 30 | 3,9105 | -1,032 | 30 | 1,3987 | -0,842 | 30 | 1,0421 | -0,679 |
| 31 | 0,2789 | -0,259 | 31 | 0,3556 | -0,508 | 31 | 2,8912 | -0,935 | 31 | 1,5862 | -0,882 | 31 | 0,9446 | -0,648 |
| 32 | 0,2793 | -0,261 | 32 | 0,583 | -0,658 | 32 | 3,1815 | -0,959 | 32 | 1,7635 | -0,915 | 32 | 1,0279 | -0,672 |
| 33 | 0,332 | -0,323 | 33 | 0,3912 | -0,534 | 33 | 3,3752 | -0,977 | 33 | 1,8682 | -0,931 | 33 | 1,1763 | -0,721 |
| 34 | 0,2501 | -0,236 | 34 | 0,4914 | -0,606 | 34 | 3,9389 | -1,028 | 34 | 1,6546 | -0,892 | 34 | 1,0954 | -0,701 |
| 35 | 0,1706 | -0,117 | 35 | 1,029 | -0,832 | 35 | 2,7657 | -0,911 | 35 | 1,8024 | -0,92 | 35 | 0,9648 | -0,658 |
| 36 | 0,3414 | -0,335 | 36 | 0,6509 | -0,696 | 36 | 2,4344 | -0,868 | 36 | 1,9166 | -0,942 | 36 | 1,7021 | -0,825 |
| 37 | 0,1641 | -0,1 | 37 | 0,3134 | -0,479 | 37 | 2,9533 | -0,925 | 37 | 1,8694 | -0,938 | 37 | 1,7162 | -0,828 |
| 38 | 0,3152 | -0,311 | 38 | 0,1735 | -0,297 | 38 | 1,9803 | -0,798 | 38 | 1,4184 | -0,86 | 38 | 1,9785 | -0,864 |
| 39 | 0,1709 | -0,114 | 39 | 0,154 | -0,264 | 39 | 1,7931 | -0,764 | 39 | 1,6929 | -0,917 | 39 | 1,9474 | -0,858 |
| 40 | 0,2597 | -0,249 | 40 | 0,144 | -0,231 | 40 | 1,5526 | -0,717 | 40 | 1,808 | -0,939 | 40 | 1,939 | -0,854 |
| 41 | 0,1652 | -0,105 | 41 | 0,1705 | -0,275 | 41 | 0,866 | -0,538 | 41 | 1,7528 | -0,93 | 41 | 0,6671 | -0,534 |
| 42 | 0,1856 | -0,139 | 42 | 0,2425 | -0,364 | 42 | 0,7733 | -0,501 | 42 | 1,4221 | -0,866 | 42 | 0,6406 | -0,519 |
| 43 | 0,1768 | -0,126 | 43 | 0,4085 | -0,498 | 43 | 0,6922 | -0,467 | 43 | 1,5041 | -0,882 | 43 | 0,5628 | -0,481 |

| DBH | a | b | DBH | a | b | DBH | a | b | DBH | a | b | DBH | a | b |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| [cm] | [-] | [-] | [cm] | [-] | [-] | [cm] | [-] | [-] | [cm] | [-] | [-] | [cm] | [-] | [-] |
| 44 | 0,1758 | -0,123 | 44 | 0,5248 | -0,547 | 44 | 1,3849 | -0,673 | 44 | 1,5395 | -0,888 | 44 | 0,5652 | -0,48 |
| 45 | 0,1552 | -0,085 | 45 | 0,8136 | -0,655 | 45 | 1,3791 | -0,671 | 45 | 1,5698 | -0,894 | 45 | 0,5837 | -0,49 |
| | | | | | | 46 | 1,2579 | -0,644 | 46 | 1,7675 | -0,931 | 46 | 0,6886 | -0,533 |
| | | | | | | 47 | 1,1855 | -0,624 | 47 | 1,9749 | -0,966 | 47 | 0,7035 | -0,539 |
| | | | | | | 48 | 1,21 | -0,627 | 48 | 2,1338 | -0,988 | 48 | 0,7215 | -0,544 |
| | | | | | | 49 | 1,1824 | -0,618 | 49 | 2,3454 | -1,016 | 49 | 0,7327 | -0,547 |
| | | | | | | 50 | 1,1089 | -0,598 | 50 | 2,3353 | -1,014 | 50 | 0,7406 | -0,549 |
| | | | | | | 51 | 1,8677 | -0,805 | 51 | 2,1601 | -0,991 | 51 | 1,1138 | -0,723 |
| | | | | | | 52 | 1,991 | -0,823 | 52 | 2,2544 | -1,005 | 52 | 1,1756 | -0,74 |
| | | | | | | 53 | 1,8602 | -0,802 | 53 | 2,5409 | -1,042 | 53 | 1,2257 | -0,753 |
| | | | | | | 54 | 1,4777 | -0,732 | 54 | 3,0521 | -1,097 | 54 | 1,3481 | -0,781 |
| | | | | | | 55 | 1,4668 | -0,731 | 55 | 3,6684 | -1,153 | 55 | 1,7942 | -0,866 |
| | | | | | | 56 | 1,5268 | -0,742 | 56 | 3,5995 | -1,148 | 56 | 2,3049 | -0,943 |
| | | | | | | 57 | 1,2414 | -0,681 | 57 | 3,8278 | -1,167 | 57 | 2,4282 | -0,958 |
| | | | | | | 58 | 1,2286 | -0,68 | 58 | 4,1243 | -1,187 | 58 | 2,657 | -0,985 |
| | | | | | | 59 | 1,2898 | -0,693 | 59 | 4,4534 | -1,209 | 59 | 2,8552 | -1,005 |
| | | | | | | 60 | 1,1835 | -0,666 | 60 | 4,6764 | -1,223 | 60 | 2,9744 | -1,015 |
| | | | | | | 61 | 1,0813 | -0,639 | 61 | 4,2576 | -1,198 | 61 | 2,6675 | -0,985 |
| | | | | | | 62 | 0,9142 | -0,589 | 62 | 4,5462 | -1,218 | 62 | 2,8401 | -1,004 |
| | | | | | | 63 | 1,2544 | -0,681 | 63 | 4,7209 | -1,23 | 63 | 2,9276 | -1,013 |
| | | | | | | 64 | 1,1556 | -0,655 | 64 | 4,9461 | -1,245 | 64 | 3,0536 | -1,025 |
| | | | | | | 65 | 1,1879 | -0,663 | 65 | 5,2441 | -1,264 | 65 | 3,2083 | -1,041 |
| | | | | | | 66 | 1,1293 | -0,648 | 66 | 5,1276 | -1,257 | 66 | 3,1495 | -1,036 |
| | | | | | | 67 | 1,1271 | -0,647 | 67 | 5,6231 | -1,284 | 67 | 3,4522 | -1,062 |
| | | | | | | 68 | 1,2685 | -0,679 | 68 | 5,3941 | -1,273 | 68 | 3,2028 | -1,041 |
| | | | | | | 69 | 1,2642 | -0,677 | 69 | 5,5844 | -1,283 | 69 | 3,4258 | -1,06 |
| | | | | | | 70 | 1,1479 | -0,649 | 70 | 5,868 | -1,297 | 70 | 3,6111 | -1,076 |
| | | | | | | 71 | 1,1885 | -0,66 | 71 | 7,2517 | -1,359 | 71 | 2,2269 | -0,935 |
| | | | | | | 72 | 1,1449 | -0,647 | 72 | 7,7494 | -1,379 | 72 | 2,3893 | -0,955 |
| | | | | | | | | | 73 | 9,0804 | -1,427 | 73 | 12,399 | -1,523 |
| | | | | | | | | | 74 | 9,8528 | -1,453 | 74 | 13,5 | -1,55 |
| | | | | | | | | | 75 | 10,528 | -1,474 | 75 | 14,515 | -1,573 |
| | | | | | | | | | 76 | 11,741 | -1,506 | 76 | 16,305 | -1,607 |
| | | | | | | | | | 77 | 11,682 | -1,504 | 77 | 16,232 | -1,605 |
| | | | | | | | | | 78 | 11,619 | -1,502 | 78 | 16,153 | -1,603 |
| | | | | | | | | | 79 | 12,5 | -1,521 | 79 | 17,477 | -1,624 |
| | | | | | | | | | 80 | 13,49 | -1,543 | 80 | 18,981 | -1,649 |
| | | | | | | | | | 81 | 10,688 | -1,478 | 81 | 14,92 | -1,581 |
| | | | | | | | | | 82 | 11,397 | -1,495 | 82 | 15,985 | -1,6 |
| | | | | | | | | | 83 | 11,501 | -1,497 | 83 | 16,138 | -1,601 |
| | | | | | | | | | 84 | 12,449 | -1,518 | 84 | 17,562 | -1,624 |
| | | | | | | | | | 85 | 13,1 | -1,531 | 85 | 18,597 | -1,639 |
| | | | | | | | | | 86 | 10,245 | -1,46 | 86 | 14,4 | -1,565 |
| | | | | | | | | | 87 | 10,781 | -1,474 | 87 | 15,203 | -1,581 |

| DBH | a | b | DBH | a | b | DBH | a | b | DBH | a | b | DBH | a | b |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| [cm] | [-] | [-] | [cm] | [-] | [-] | [cm] | [-] | [-] | [cm] | [-] | [-] | [cm] | [-] | [-] |
| | | | | | | | | | 88 | 11,318 | -1,487 | 88 | 16,036 | -1,594 |
| | | | | | | | | | 89 | 11,758 | -1,498 | 89 | 16,7 | -1,606 |
| | | | | | | | | | 90 | 12,533 | -1,516 | 90 | 17,899 | -1,626 |
| | | | | | | | | | 91 | 11,888 | -1,504 | 91 | 16,907 | -1,612 |
| | | | | | | | | | 92 | 11,459 | -1,496 | 92 | 16,299 | -1,604 |
| | | | | | | | | | 93 | 11,22 | -1,493 | 93 | 15,967 | -1,602 |
| | | | | | | | | | 94 | 11,192 | -1,496 | 94 | 15,943 | -1,605 |
| | | | | | | | | | 95 | 10,341 | -1,477 | 95 | 14,705 | -1,585 |
| | | | | | | | | | 96 | 7,2098 | -1,373 | 96 | 10,024 | -1,475 |
| | | | | | | | | | 97 | 5,6075 | -1,303 | 97 | 7,6592 | -1,399 |
| | | | | | | | | | 98 | 4,8298 | -1,258 | 98 | 6,5393 | -1,351 |
| | | | | | | | | | 99 | 4,0853 | -1,207 | 99 | 5,4562 | -1,296 |
| | | | | | | | | | 100 | 3,5856 | -1,166 | 100 | 4,727 | -1,252 |

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# List of publications based on the results of the dissertation

The definition of urban forests, the distribution, classification, and characteristics described in Part I were already published in the following journals:

*Scholz, T. (2020): Urbane Wälder im Ruhrgebiet – Klassifikation, Merkmale und Regulationsleistungen. Jahrbuch des Bochumer Botanischen Vereins 11: 339-356.*

*Scholz, T., Schmitt, T.; Schmitt, M. (2021): Urbane Waldnutzung. Keil, P.; Hering, D.; Schmitt, T.; Zepp, H. (Ed.): Positionen zu einer Regionalen Biodiversitätsstrategie Ruhrgebiet. Studie im Rahmen der Offensive Grüne Infrastruktur 2030. Oberhausen, Essen, Bochum: 76-103.*

The data collection, the concept, and the writing were performed by the author of this dissertation. The co-authors of the second publication proof-read the manuscripts and gave suggestions.

# Eidesstattliche Versicherung

Ich versichere an Eides statt, dass ich die eingereichte Dissertation selbstständig und ohne unzulässige fremde Hilfe verfasst, andere als die in ihr angegebene Literatur nicht benutzt und dass ich alle ganz oder annähernd übernommenen Textstellen sowie verwendete Grafiken, Tabellen und Auswertungsprogramme kenntlich gemacht habe. Außerdem versichere ich, dass die vorgelegte elektronische mit der schriftlichen Version der Dissertation übereinstimmt und die Abhandlung in dieser oder ähnlicher Form noch nicht anderweitig als Promotionsleistung vorgelegt und bewertet wurde. Die digitalen Abbildungen der Dissertation enthalten nur die originalen Daten. Sofern Art und Umfang durch eine inhaltsverändernde Bildbearbeitung vorgenommen wurde, wurde dies dokumentiert. Es wurde keine kommerzielle Vermittlung oder Beratung in Anspruch genommen.

Tobias Scholz

Essen, den 17.07.2022

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