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Urban soil respiration patterns: An analysis of *Quercus robur vs. Quercus rubra* in parks and streets of Karlsruhe, Germany

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ABSTRACT

The dynamics of soil respiration (R_s) in natural forest and urban forest ecosystems are already well-researched. However, there is still little research on soil respiration under individual trees in urban areas, particularly regarding the influence of the habitat, different tree species, development stage, and their interplay on soil respiration. In this study, we examined how R_s varies between the two most common urban tree-habitats (park and street), two tree species (*Quercus robur* L. and *Quercus rubra* L.), and tree development stages (a three-tier classification system based on the developmental stages of urban trees which is linked to the time since planting). We hypothesized that R_s values would differ between habitats but not between the native *Quercus robur* and non-native *Quercus rubra*. Additionally, we expected a decrease in R_s with the advancing development stage. The study involved 60 trees (30 per tree species, 20 per development stage) distributed across Karlsruhe, southwest Germany. We measured R_s using a portable soil gas chamber. Alongside R_s, environmental (site and soil attributes), and dendrometric data were collected. Regression analysis revealed an interaction between tree species and development stage with statistically significant effect on R_s. *Quercus robur* displayed contrasting R_s patterns in parks (increase) and streets (initial increase, then decrease) with progressing development stage. In contrast, *Quercus rubra* showed only in streets a clear (negative) trend. While *Quercus rubra* R_s differed between locations only in the last development stage, *Quercus robur* showed higher R_s in streets than in parks in the first development stage, with a reversal in the last stage. Additionally, *Quercus rubra* exhibited overall higher mean R_s values than *Q. robur* (4.20 ± 0.25 μmol m^{-2} s^{-1} *vs.* 3.56 ± 0$^{·18}$ μmol m^{-2} s^{-1}). Soil water content, soil water content × soil temperature interaction, soil pH, and impervious surface cover had also a statistically significant effect on R_s. Parks had higher median soil water content (0.11 ± 0.07 m^3m^{-3}) and soil pH levels (6 ± 0.74) compared to streets (0.06 ± 0.06 m^3m^{-3}; 5.5 ± 0), while streets had a greater median of impervious surface beneath the tree canopy (20 ± 29 % *vs.* 0 ± 0 %). Additionally, lower values for R_s and soil pH, which can also be indicators of soil quality, may suggest poorer soil quality at older sites and street locations. Further investigations could clarify whether this is true and if the varying application of soil amendments by the horticultural department is causing the different trends in R_s rates, as well a_s which other factors further explain the variance in R_s.

1. Introduction

The World Bank reports that over half of the human population on Earth resided in urban areas by 2018, with projections estimating a rise to 66 % by 2050 (Wahba Tadros et al., 2021). However, urban environments differ significantly from rural ones. The urban heat island (UHI) effect exemplifies how urban architecture, materials, and lifestyle choices contribute to higher surface air temperatures in cities compared to surrounding areas (Wang et al., 2020; Sachindra et al., 2016; Khan et al., 2021). The UHI can decrease life expectancy and increase the frequency of physical and mental health issues for urban residents (Yang et al., 2016).

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Beyond urbanization, our lifestyles are altering the global climate, leading to a rise in average surface air temperature and increased extreme weather events like droughts and heavy rain (IPCC, 2007). Climate change is expected to have a more pronounced impact on urban areas due to synergy effects (Wang et al., 2020). Changes in local weather patterns induced by climate change, such as wind speed or cloud cover variations, can exacerbate UHI intensity (Chapman et al., 2017). Furthermore, model simulations suggest that the combined effects of climate change and urbanization lead to a more substantial rise in surface temperatures compared to either factor alone (Chapman et al., 2017).

Several studies demonstrate that urban vegetation, including parks, and street trees can significantly reduce surface temperatures in cities through evapotranspiration and shading (Barradas et al., 2022; Wang et al., 2020). Additionally, urban vegetation contributes to the mental well-being of urban populations (Beckmann-Wübbelt et al., 2021), highlighting the need and importance of plants in the city and thus emphasizing the city as a significant habitat for plants. However, research on urban ecosystems related to soil gas emissions and their interaction with vegetation remains limited, with current understanding often relying on observations from natural or non-urban areas (Karvinen et al., 2024).

The urban environment differs in many ways from the surrounding rural areas and natural landscapes, as do the conditions for plant growth and health (Rebele, 1994; Sukopp and Werner, 1983; Sæbø et al., 2003). Drainage systems and sealed surfaces lead to reduced water availability (Sukopp and Werner, 1983; Sæbø et al., 2003; Savi et al., 2015), while artificial irrigation systems maintain consistently high water availability (Decina et al., 2016; Kaye et al., 2005). Human influences alter the soils, affecting both quality (by deposition of toxic substances, pollution) and composition (natural substrate from parent rock, artificial substrates or mixtures of natural and artificial resulting into urban soil type *Technosol*) within the city area (Rebele, 1994; Sukopp and Werner, 1983). Nutrient cycling is further influenced by removing litter from urban vegetation (Templer et al., 2015; Ivanova et al., 2015) as well as artificial inputs. Utility infrastructure such as sewage pipes limits the available space for the root system (Sæbø et al., 2003). Growth above ground is limited by infrastructure and buildings. The latter also limits the availability of light through shading but can change the phenology of plants through artificial lighting too (Sæbø et al., 2003).

While urban areas differ from rural environments, distinct locations within the city exhibit variations (Sukopp and Werner, 1983). Comprehensively described how site properties influence microclimate, soil, plant growth, and plant vitality. They identified nine key urban locations: residential areas, industrial zones, inner-city districts, green spaces, traffic sites, waste disposal facilities, forests, fields, and water bodies. For instance, green spaces offer microclimates similar to natural ecosystems. However, these areas face external influences. Air deposition and visitor activity can introduce nutrients, potentially leading to eutrophication. Additionally, visitor traffic can cause trampling damage, favoring the establishment of species tolerant of high nutrient levels and trampling. In contrast, traffic routes experience significantly higher temperatures and lower humidity (Karvinen et al., 2024). These areas are also heavily exposed to pollutants like road salt and vehicle emissions (see also Sæbø et al., 2003). Traffic compacts the soil, hindering water uptake and gas exchange (Jim, 1998). Furthermore, street vegetation, particularly street trees, suffers from heavy pruning for traffic safety and limited space due to buildings and infrastructure (Sæbø et al., 2003). These harsh conditions lead to lower plant vitality and shorter lifespans for vegetation along traffic routes (Sukopp and Werner, 1983).

Soil respiration R_s offers valuable insights into total belowground carbon allocation (Ryan and Law, 2005) and short-term changes in soil quality and tree health (Crookston et al., 2023; Rodríguez et al., 2022; Kim and Yoo, 2021). R_s comprises two key components: autotrophic and heterotrophic respiration (Ryan and Law, 2005). Autotrophic respiration is closely linked to plant growth and maintenance. Notably, mycorrhizal activity contributes also to autotrophic respiration through direct carbohydrate uptake from plants (Ryan and Law, 2005). Both processes are tightly coupled with photosynthesis and assimilate translocation (Ryan and Law, 2005; Högberg et al., 2001; Han et al., 2014). Heterotrophic respiration, on the other hand, depends on substrate quality, available carbon, decomposer enzyme activity, and soil fauna (Ryan and Law, 2005; Liu et al., 2018; Borges Pinto et al., 2018; Tian et al., 2019; Li et al., 2018; Şandor et al., 2015).

Several environmental factors modulate autotrophic and heterotrophic respiration. Temperature, moisture, and soil properties play significant roles (Raich and Tufekciogul, 2000). Temperature and moisture influence both photosynthesis and microbial respiration rates (Orchard and Cook, 1983; Högberg et al., 2001; Kelliher et al., 2004; Chen et al., 2014). Soil pH affects nutrient availability for plants and microbes, impacting plant growth and enzyme activity (Li et al., 2018). Lower C/N ratios or higher nitrogen availability enhance both heterotrophic and autotrophic respiration (Chen et al., 2016; Tian et al., 2019; Preece et al., 2020). However, physical properties like soil compaction can hinder R_s by limiting gas diffusion (Yanjun et al., 2020; Cambi et al., 2017; Tian et al., 2019).

Plants influence R_s beyond just photosynthesis. Litter inputs alter soil carbon pools and microbial communities, thereby affecting R_s (Stanek and Stefanowicz, 2019; Woś et al., 2023; Zhang et al., 2020). Litter quantity and quality vary among species (Stanek and Stefanowicz, 2019; Woś et al., 2023). Freschet et al. (2012) demonstrated that plants exert both direct and indirect effects on the decomposition of their own litter and that of other plants through their litter quality and their impact on decomposer communities. However, the impact of litter quality and quantity on R_s might be less significant in urban areas, where up to 60 % of litter C and N can be removed by city management, as shown by Templer et al. (2015) for Boston, Massachusetts, USA.

Intriguingly, R_s can vary within a species based on the developmental stage (Zhang et al., 2020; Kelliher et al., 2004; Zhao et al., 2016). Studies report conflicting observations: some show increasing R_s with age (Wu et al., 2020), while others suggest a peak in young trees due to higher fine root biomass, and microbial activity (Kukumägi et al., 2017; Saiz et al., 2006; Kelliher et al., 2004).

Previous studies have focused more on soil organic carbon stocks than R_s, or have been limited to urban forests, parks, and gardens, rarely addressing street locations. Few studies have examined R_s, with some including street locations. For example, Weissert et al. (2016) observed soil organic carbon stocks and CO_2 efflux in tree- and grass-dominated urban ecosystems but found no significant difference in CO_2 efflux. Vasenev et al. (2018) investigated the spatial and temporal variability of R_s in ecosystems within the Moscow metropolitan area and found higher R_s values for more disturbed sites and lower values for near-natural sites. Decina et al. (2016) observed a strong increase in CO_2 efflux in urban areas during the growing season. They reported CO_2 efflux, emitted from soil respiration, equivalent to 72 % of fossil fuel CO_2 emissions between May and October in the Boston metropolitan area, with respiration rates 2.2 times higher than in adjacent rural areas. Wei et al. (2014) and Wu et al. (2016) studied how sealed surfaces in cities affect R_s. Wei et al. (2014) reported decreasing R_s values due to sealing, attributed to decreased microbial activity, among other factors. However, Wu et al. (2016) observed decreasing R_s with increasing distance from sealed surfaces. Karvinen et al. (2024) studied R_s and soil organic carbon in four common urban green spaces in Helsinki, including parks and street greenery, finding no consistently significant differences in R_s values between sites, but noting sporadic differences due to variations in soil moisture.

Previous studies on R_s in urban areas have shown partly contradictory results. Most research focuses on near-natural sites such as forests and parks, while heavily transformed sites like street locations are rarely considered. Furthermore, no studies to date have investigated the differentiation of R_s under *Quercus robur* L., Fagaceae, and *Quercus rubra* L., Fagaceae, in different development stages in urban environments.

Both *Quercus robur* and *Q. rubra* are well-suited for warm and dry conditions (Hirsch et al., 2023; Gillner et al., 2014), making them popular urban tree species. Urban environments, characterized by surface sealing and dense development, create hot and dry conditions (Yang et al., 2016), which are expected to intensify due to climate change (Wang et al., 2020; Rannow et al., 2010). The IPCC predicts rising surface air temperatures and an increase in extreme weather events such as droughts (IPCC, 2007), highlighting the importance of selecting tree species that are more tolerant to heat and water deficits. *Quercus robur* and *Q. rubra* have been identified as suitable candidates for future climate scenarios (Gillner et al., 2014; Laux et al., 2022). Additionally, they hold cultural significance and provide habitat potential for flora and fauna. Laux et al. (2022) demonstrated that *Quercus robur* supports a high diversity of microhabitats in urban environments of Karlsruhe, Germany (parks and streets), while *Q. rubra* showed fewer signs of decline (crown dieback, defoliation, discolored foliage) than *Q. robur*.

Against this background, this study focuses on R_s in urban areas and investigates differences between park and street locations, *Quercus robur* and *Q. rubra*, and tree development stages. The following working hypotheses are addressed:

1. R_s is lower at street locations than at park locations
2. R_s does not differ between native *Quercus robur* and non-native *Q. rubra*
3. R_s decreases with the progressing tree development stage.

2. Material and methods

We conducted a cross-sectional study using multistage sampling in Karlsruhe, Germany (Dormann, 2013). We selected parks and streets containing *Quercus robur* and *Q. rubra* across all three development stages. The Karlsruhe Horticulture Department provided us with the city tree cadaster (Stadt Karlsruhe, 2022).

2.1. Study area

Karlsruhe lies in southwestern Germany (8° 24’ 18” E, 49° 00’ 54” N), covering 173.49 square kilometers. The city center sits at 114.9 m above sea level. The region’s location in the Upper Rhine Valley is sheltered by the Vosges Mountains and the Black Forest. The long-term annual averages for temperature, precipitation, and sunshine hours during the reference period 1990–2020 were 11.2 °C, 803 mm, and 1782 hours, respectively (Stadt Karlsruhe, 2023). According to the tree cadaster provided by the horticulture department of Karlsruhe, the city has approximately 397,000 single-standing trees within the administrative boundary , out of which 137,000 street and park trees were owned by the horticulture department of the city (Gartenbauamt Karlsruhe, 2022). Trees of the genera Maple, Hornbeam, and Oak dominate the urban tree population in Karlsruhe (30.84 %, 11.12 %, and 10.04 %, respectively) (Stadt Karlsruhe, 2022a).

2.2. Tree selection

We selected 60 trees (“tree” refers to a single standing tree, characterized by a single trunk) with no visible signs of disease or decline (e.g. fungal fruiting bodies, crown dieback) from 8806 *Quercus robur* and *Q. rubra* listed in the tree cadaster. We chose parks and streets containing both species across all development stages (Fig. 1 and Table 1). But *Quercus rubra* scarcity in development stage III limited site selection (“site” refers to contiguous street sections or individual parks). The conditions prevailing at eligible sites (inaccessibility of the soil due to dense, impenetrable vegetation; crown dieback) made it impossible to

Table 1
Number of selected trees based on a factorial combination of species, habitat, and development stage (DS).

Species		Park	Street	**Total**
Quercus robur	DS I	5	5	**30**
	DS II	5	5	
	DS III	5	5	
Quercus rubra	DS I	5	5	**30**
	DS II	5	5	
	DS III	5	5	
Total		**30**	**30**	**60**

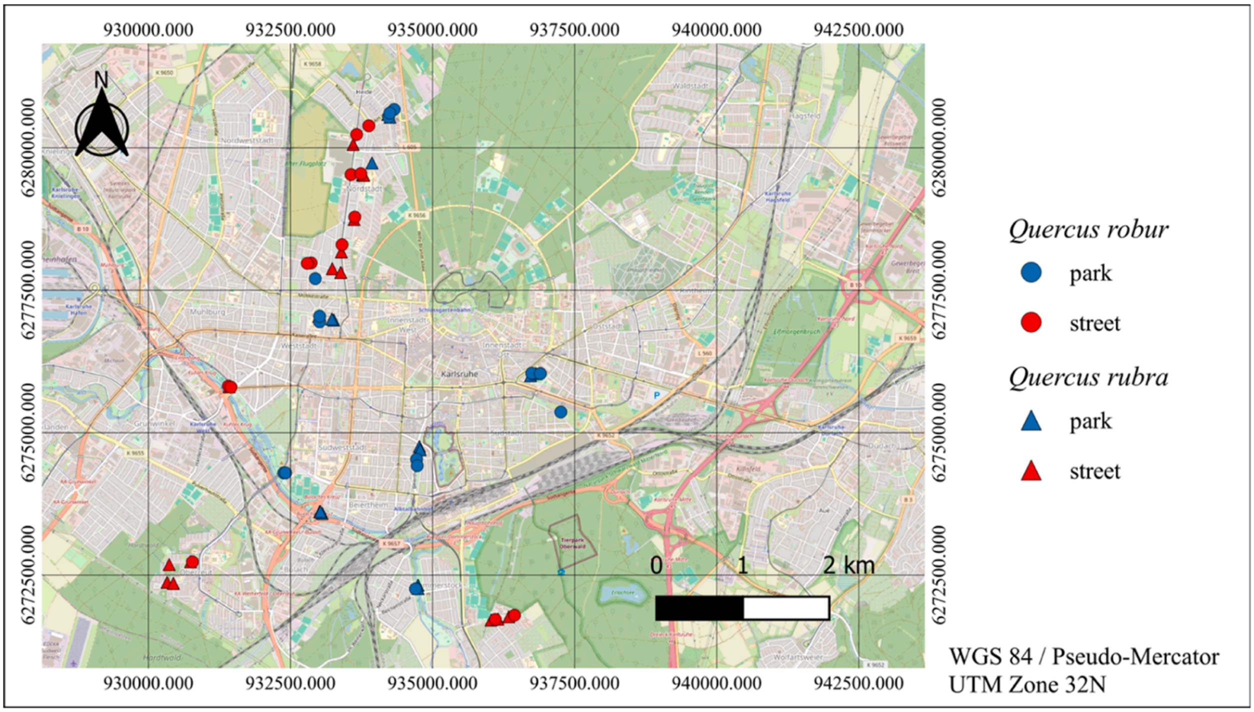

Fig. 1. Map of Karlsruhe, Germany displaying selected park and street trees of *Quercus robur* and *Quercus rubra* (n = 60).

find trees of both species in all three development stages at those sites. Therefore, substitute trees were selected at alternative locations, close to the original locations. Hence, not all sites contained both species across all development stages. We selected locations with similar distances to buildings, neighboring trees, and roads to minimize single-site effects. The Karlsruhe City Department of Gardens and Green Spaces classified trees into park or street trees, and development stages based on the Research Society for Landscape Development and Landscape Design (FLL) guidelines. For these long-lived species, the juvenile stage (development stage I) extends up to 15 years after planting, the maturity stage (development stage II) from 15 to 80 years after planting, and the decay stage (development stage III) from 80 years after planting (FLL, 2020). The exact age of the trees was not determined for this study. Furthermore, the planting age of the trees, and the planting year was not available in the city tree cadaster.

2.3. Data collection

We conducted the fieldwork over 10 days in August 2023, as literature suggests that both sun and shade leaves are still at their peak assimilation rates during this period, before the decreasing assimilation rates could negatively affect root respiration (Ruehr and Buchmann, 2010). According to Schulze et al. (2005) sun leaves reach their maximum assimilation rate in mid-June, but assimilation rate decreases again by August. In contrast, shade leaves reach their maximum assimilation rate in August, which declines shortly thereafter. All measurements were taken between 10 am and 3 pm local time, we visited each of the 60 trees once. All variables collected are shown in Table 2.

2.3.1. Soil respiration R_s

Following manufacturer guidelines, we measured R_s using a LI-COR Smart Chamber and LI-870 CO2/H2O Analyzer (LI-COR, Inc., 2024b) (Fig. 2). The chamber was closed, and the CO_2 flux was measured for 90 seconds. The LI-870 uses non-dispersive infrared technology for CO_2 (0–20,000 ppm) and water vapor measurements. To ensure accurate R_s, we used standardized PVC soil collars instead of placing the chamber directly on the soil (LI-COR, Inc., 2024b). We installed collars a day prior to prevent soil disturbance and consistently placed the collars 0.5 m from the tree trunk, as R_s decreases with distance (Tang and Baldocchi, 2005). The soil surface was not manipulated further for the R_s measurements; vegetation or litter was not removed. We measured each tree twice in succession (north and south, both values were treated as individual values in our subsequent analysis) at least 12 hours after rewetting to avoid inflated R_s due to water displacing soil air (Preece et al., 2020). A Stevens HydraProbe connected via wire to the gas chamber, measured soil temperature (T_s) and soil water content (SWC) at 10 cm depth, simultaneously to respiration measurements.

2.3.2. Environmental data collection

We collected environmental data alongside R_s measurements. These data improve R_s interpretation and reduce uncertainties. A Kestrel 5400 WBGT Heat Stress Tracker & Weather Meter measured air temperature and relative humidity 20 cm north of the stem at breast height (ca. 135 cm). We identified and classified soil vegetation type within the collar according to Nowak (2021). 93 % of the trees had grass (Poaceae, Cyperaceae or Juncaceae family) as ground cover, 5.2 % had herbaceous vegetation (Herbaceous ground cover, other than grass, including agricultural crops), and 1.8 % had bare soil (soil without cover by vegetation or litter, includes naturally occurring sand). After each of the two R_s measurements per tree, we assessed soil compaction within the collars with a penetrometer (Step Systems, 2024). We measured photosynthetic photon flux density (PPFD) with an LI-180 Spectrometer in an open area near the tree, simultaneously to respiration measurements, approximating photosynthetic activity as Ruehr and Buchmann (2010) suggested.

2.3.3. Soil analysis

We collected one soil sample per tree, approximately 1 dm^3 in volume, packed and sealed them in plastic food bags, and stored them under refrigerated conditions until analysis. In the laboratory, we analyzed these samples for pH, nitrate, ammonium, and grain size distribution. Following manufacturer instructions, we used the VISOCOLOR reagent case and a PF-3 compact photometer for analysis (Macherey-Nagel GmbH and KG, 2021).

We sieved samples to remove large particles. We compacted soil in test tubes for grain size distribution, added pyrophosphate to prevent clay particle clumping and water, and then shook the mixture until homogenous. After the mixture settled for 18 seconds in a vertical position, we read the proportion of sand that had been sedimented during this time using the percentage indications on the tube.

We prepared a soil extract using a 0.0125 mol/L calcium chloride solution for nutrient analysis. We mixed 200 mL solution with 50 g soil for five minutes, filtered with an MN 616 1/4 filter, and determined soil pH colorimetrically. We measured nitrate- and ammonium-nitrogen photometrically using a PF-3 photometer and specific reagents. We calculated nitrate and ammonium values based on soil moisture content. Therefore, we spread the samples flat on a surface and air-dried the soil samples at room temperature for at least 24 hours to determine the correction factor.

As the Munsell value class (lightness) can infer organic carbon content (Ferrando Jorge et al., 2021), we classified soil samples using the Munsell color system. We calculated a qualitative C/N ratio by dividing the normalized lightness values by the normalized nitrogen contents.

2.3.4. Dendrometric, health, and neighbouring variables

Following Nowak (2021) (US Department of Agriculture), trees underwent dendrometric measurements (diameter at breast height (DBH), height, crown metrics) and visual assessment of dieback, crown light exposure (number of sides of the tree receiving sunlight, 0–5 sides with top of the tree counted as one side), impervious surface beneath canopy (percent of land area under the canopy drip line that is impervious), proximity and direction to buildings, pruning (number of pruned branches with a diameter above 10 cm). DBH measurements utilized a tape measure, distance and height measurements a laser rangefinder.

2.3.5. Statistical analysis

We used Microsoft Excel and SoilFluxPro Software v5 (LI-COR, Inc., 2024a) for data preparation. We performed statistical analyses using R version 4.3.2 (R Core Team, 2023), specifically linear mixed-effects models. Mixed effects models are a versatile analytical tool that enables researchers to investigate the condition of interest while accounting for variability both within and across individuals and items at the same time (Brown, 2021).

We assessed R_s distribution using the *fitdistr* function from the “MASS” package (Venables and Ripley, 2002) and a Shapiro-Wilk normality test (R Core Team, 2023). R_s data required logarithmic and root transformations for normality. We used Levene’s test from the “car” package (Fox and Weisberg, 2019) to confirm homogeneity of variance.

Given non-normal predictors and categorical variables, we determined collinearity using cluster analysis (*varclus* function, “Hmisc” package (Harrell, 2024)). We excluded one variable from each group of highly correlated variables (Spearman $\rho^2 > 0.5$) (Dormann, 2013). Cluster analysis revealed no collinearity among core variables (habitat, tree species, development stage). DBH, total height, and crown base height showed a high correlation, as did soil temperature and air temperature (Spearman $\rho^2 > 0.5$). We retained T_s but excluded DBH due to its relation to the development stage.

We constructed linear mixed-effects models to identify variables explaining statistically significant R_s variance. Models included interactions and non-interactions (Dormann, 2013; R Core Team, 2023), optimized using Restricted Maximum Likelihood Criterion. We assessed variance inflation using the vif function from the “car” package (Fox and

Table 2

Summary table showing median and median absolute deviation for measured environmental and dendrometric variables, including soil water content (SWC), soil Temperature (T soil).

Habitat	Tree species	Develop-ment stage	Compaction	Sand	Lightness	Nitrate	Ammonium	pH	SWC	T soil	DBH	Number of pruned branches	Dieback	Tree height	Imper-vious beneath canopy	Crown light exposure
			PSI	%		mg/L	mg/L		m^3 / m^3	°C	cm		%	m	%	
Park	*Quercus robur*	I	210 ± 96	75 ± 16	3 ± 0	6.36 ± 4.54	0.33 ± 0.09	6.5 ± 0	0.1 ± 0.08	26 ± 5.7	15 ± 6	2 ± 1.48	0 ± 0	7 ± 2	0 ± 0	4 ± 1.48
		II	225 ± 37	81 ± 0	2 ± 0	9.84 ± 7.26	2.06 ± 0.62	6 ± 0.74	0.15 ± 0.03	24.8 ± 5.7	91 ± 80	8 ± 7.41	0 ± 0	18 ± 6	5 ± 7.41	4 ± 1.48
		III	250 ± 52	86 ± 6	3 ± 0.74	6.96 ± 4.82	0.64 ± 0.21	5.5 ± 0.74	0.09 ± 0.03	21.8 ± 4.2	121 ± 13	4 ± 0	0 ± 0	20 ± 5	0 ± 0	4 ± 1.48
	Quercus rubra	I	165 ± 96	88 ± 1	3 ± 1.48	2.86 ± 1.3	0.66 ± 0.6	6.5 ± 0.74	0.11 ± 0.06	21.9 ± 3.1	8 ± 1	0 ± 0	0 ± 0	6 ± 2	0 ± 0	4 ± 1.48
		II	175 ± 37	86 ± 13	2.5 ± 0	7.48 ± 6.36	0.43 ± 0.35	5.5 ± 0	0.14 ± 0.08	25.8 ± 5.7	57 ± 12	3 ± 4.45	0 ± 0	16 ± 4	0 ± 0	3 ± 1.48
		III	200 ± 74	80 ± 9	3 ± 0	3.39 ± 1.51	0.72 ± 0.38	5.5 ± 0	0.07 ± 0.06	26.1 ± 3.2	133 ± 6	3 ± 1.48	0 ± 0	23 ± 3	0 ± 0	3 ± 1.48
Street	*Quercus robur*	I	140 ± 7	82 ± 4	3 ± 0.74	6.96 ± 1.65	0.62 ± 0.56	6 ± 0.74	0.11 ± 0.05	22.5 ± 2.2	12 ± 6	2 ± 2.97	0 ± 0	7 ± 1	10 ± 14.83	5 ± 0
		II	245 ± 56	78 ± 4	3 ± 0.74	5.56 ± 0.89	0.89 ± 0.41	5.5 ± 0	0.05 ± 0.07	25.3 ± 3.2	34 ± 14	6 ± 7.41	0 ± 0	11 ± 5	30 ± 25.95	5 ± 0
		III	225 ± 0	85 ± 6	2.5 ± 0	4.98 ± 3.87	1.24 ± 0.58	5.5 ± 0	0.02 ± 0.02	29.8 ± 0.9	95 ± 4	8 ± 2.97	0 ± 0	18 ± 2	50 ± 14.83	4 ± 0
	Quercus rubra	I	150 ± 22	86 ± 0	3 ± 0	2.37 ± 1.99	0.5 ± 0.38	6.5 ± 0.74	0.07 ± 0.05	25.4 ± 3.1	7 ± 1	0 ± 0	0 ± 0	5 ± 0	0 ± 0	3 ± 1.48
		II	200 ± 74	81 ± 9	3 ± 0	5.24 ± 4.57	0.55 ± 0.2	5.5 ± 0.74	0.08 ± 0.05	26.3 ± 5.9	72 ± 16	7 ± 1.48	0 ± 0	14 ± 4	45 ± 7.41	5 ± 0
		III	288 ± 111	87 ± 2	3 ± 0	4.59 ± 2.21	0.86 ± 0.27	5.5 ± 0	0.03 ± 0.03	26.1 ± 4.2	108 ± 4	7.5 ± 8.9	0 ± 0	19 ± 3	45 ± 7.41	5 ± 0

Fig. 2. The complete soil respiration field survey setup. A yellow LI-870 CO2/H2O Analyzer is next to the white Smart Chamber, sitting on the green PVC soil collar. The Stephens HydraProbe is installed adjacent to the soil gas chamber in the ground to measure soil temperature and soil water content. (Photo: Somidh Saha).

Weisberg, 2019), disqualifying models with GVIF > 10 or adjusted GVIF > 3.16 (Dormann, 2013). We checked for residual normality with Shapiro-Wilk tests (Dormann, 2013). We identified influential points using *influence.measures* (R Core Team, 2023; Dormann, 2013), investigating points with statistically significant Cook's distance for anomalies. We identified seven influential data points of which two were excluded. After model optimization, the final model included habitat, tree species, development stage, tree species × development stage interaction, soil pH, impervious surface beneath canopy, T_s, SWC, SWC × T_s interaction as fixed effects (see Table 3). The model also included date of measurement, and tree individual as random effect, but neither of them had a statistically significant impact.

To identify variables that significantly explain the variance in R_s and to examine differences between groups, we used the ANOVA function and Tukey HSD tests from the "stats" package (R Core Team, 2023) for post hoc comparisons (Dormann, 2013). For variables identified through mixed-effects model analysis and ANOVA as having a significant influence on R_s, we conducted further analysis using the Kruskal-Wallis test and, for normally distributed variables, the *t*-test (R Core Team, 2023; Dormann, 2013). These tests were used to investigate whether habitat, tree species, or development stage had a statistically significant impact on R_s or whether soil pH, impervious surface beneath canopy, T_s, SWC, SWC × T_s interaction differed across these groups.

We utilized the "ggplot2" package for data visualisation (Wickham, 2016), with *interact_plot* for continuous variable interactions (Long, 2019). Mean values are presented with standard error, while variables with a large number of outliers are reported as median with median absolute deviation, as this is more robust to outliers (Dormann, 2013).

3. Results

3.1. Influence of environmental, dendrometric, and neighbouring variables on soil respiration in combination with habitat, species identity, and development stage

The ANOVA of the optimized mixed effects model indicated that development stage, and tree species × development stage interaction explain variance in R_s. Furthermore, soil pH, SWC, T_s × SWC, and impervious surface beneath canopy also explain variance in R_s (Table 3). The mixed effects model analysis revealed that tree species, development stage, tree species × development stage, soil pH, SWC, T_s × SWC are predictors with an statistically significant effect on R_s (Table 4, Fig. 3 and Fig. 4). *Quercus robur* showed increasing median R_s values in park locations with increasing development stage but followed an initial increase and subsequent decrease in street locations. In contrast, *Quercus rubra* displayed a consistent decrease in median R_s in street locations but no clear trend in park locations. While *Quercus rubra* exhibited differences in median R_s between park and street locations only in the third development stage, *Q. robur* showed distinct patterns in both early and later development stages. In the first development stage, street locations

Table 3
F-values and p-values of the analysis of variance of the restricted maximum likelihood criterion optimized mixed effects model including environmental and dendrometric data to analyse which variables explain a statistically significant amount of variance of R_s.

Predictors	F-value	p-value
Habitat	3.24	0.096
Tree Species	2.40	0.124
Development Stage	5.21	**0.007**
Tree Species × Development Stage	3.39	**0.038**
Soil pH	4.57	**0.035**
Soil Temperature	0.56	0.468
Soil Water Content	5.63	**0.034**
Impervious Surface Beneath Canopy	10.23	**0.002**
Soil Temperature × Soil Water Content	6.19	**0.020**
Observations	115	

Table 4
Coefficient estimates, standard errors, and p-values of the restricted maximum likelihood criterion optimized mixed effects model parameters including environmental and dendrometric data to quantify the predictors effect on R_s.

Parameter	Estimate	Std. Error	p-value
(Intercept)	0.788	0.286	**0.008**
Habitat: Street	0.095	0.053	0.096
Development Stage II	0.139	0.047	**0.004**
Development Stage III	0.204	0.050	**< 0.001**
Tree Species: *Quercus rubra*	0.121	0.044	**0.007**
Soil pH	0.057	0.027	**0.035**
Soil Water Content	−3.831	1.614	**0.025**
Soil Temperature	−0.006	0.008	0.468
Impervious Surface Beneath Canopy	−0.003	0.001	**0.002**
Development Stage II × *Quercus rubra*	−0.081	0.063	0.201
Development Stage III × *Quercus rubra*	−0.161	0.062	**0.011**
Soil Water Content × Soil Temperature	0.175	0.070	**0.020**

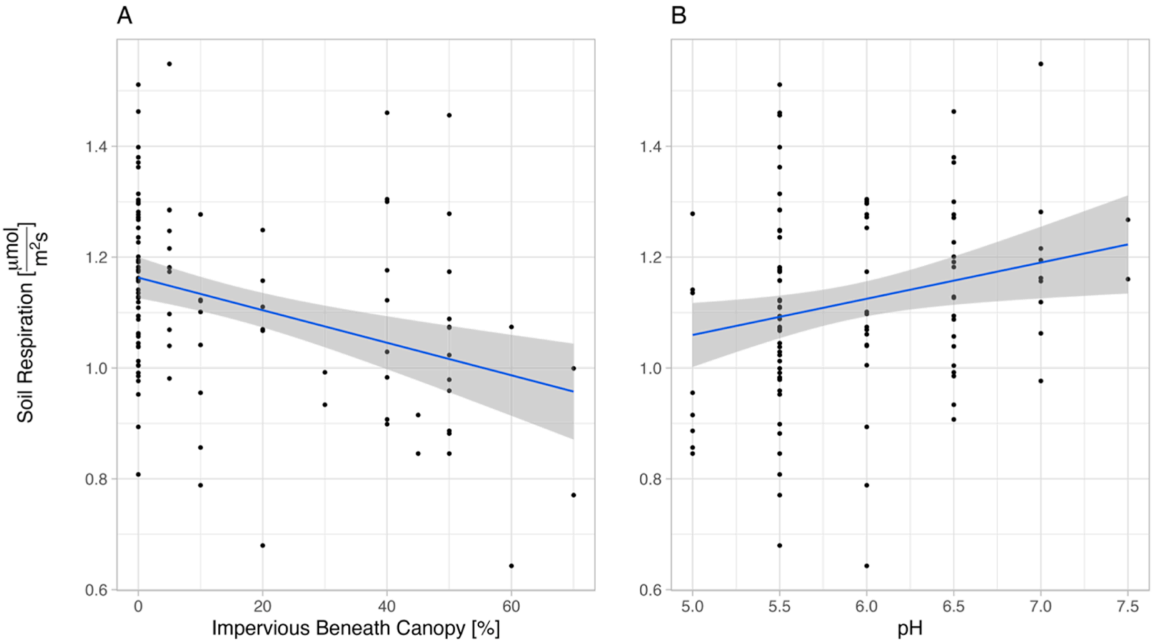

Fig. 3. Relationships between soil respiration and A) impervious surface beneath canopy and B) soil pH which explained a statistically significant proportion of the variance in soil respiration. Blue lines represent linear regression lines, and the blue shading indicates the 95 % confidence interval. Impervious surface beneath canopy: $R^2 = 0.12$ * ** , $\rho =$ -0.35 * ** . pH: $R^2 = 0.04$ * , $\rho = 0.24$ * . * = p-value < 0.05 * * = p-value < 0.01 * ** = p-value < 0.001 * ** * = p-value < 0.0001.

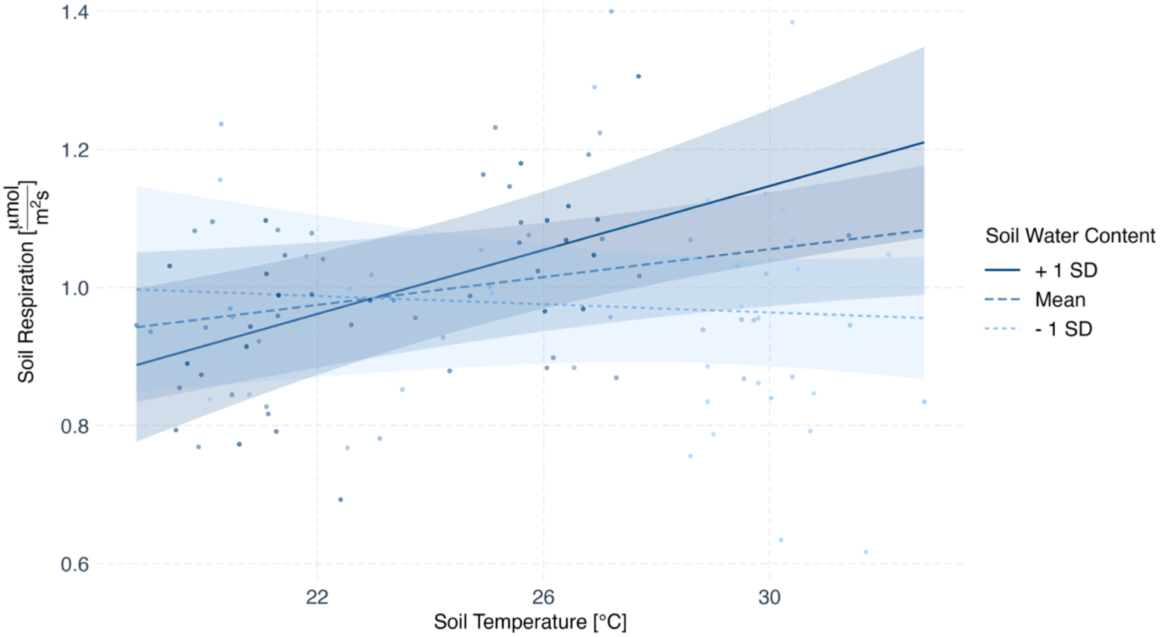

Fig. 4. Relationship of soil temperature on soil respiration depending on soil water content (SWC). Blue (dotted) lines represent linear regression lines. The blue shading indicates the 95 % confidence interval. The blue dots represent individual data points, and their color represents the soil water content present at the time of measurement. Data points with a lighter blue experienced SWC values less than the mean SWC minus one standard deviation, while data points with dark blue shade experienced SWC values greater than the mean SWC plus one standard deviation. $R^2_{lowSWC} =$ -0.3^{2,} $\rho_{lowSWC} =$ -0.3, $R^2_{highSWC} = 0.46_{*}$, $\rho_{highSWC}= 0.55$. * = p.value < 0.05 * * = p-value < 0.01 * ** = p-value < 0.001 * ** * = p-value < 0.0001.

had higher median R_s values than park, but this trend reversed in the later development stage (see also Fig. 5). Fig. 6 depicts model analysis findings, showing that *Quercus rubra* overall had higher median R_s values than *Q. robur*.

Environmental variables which explain a statistically significant share of variance of R_s differed partly between the habitat and development stages but not between species. Median SWC, soil pH, and impervious surface beneath canopy differed between park and street. Median SWC and soil pH were higher for parks than for streets, while the median percentage share of impervious surfaces under the crown projection area was greater at street locations (Fig. 7). Median SWC, soil pH, and impervious surface beneath canopy differed between development stages. Median SWC were lowest for development stage III, median soil pH of development stage II and III were decreased from development

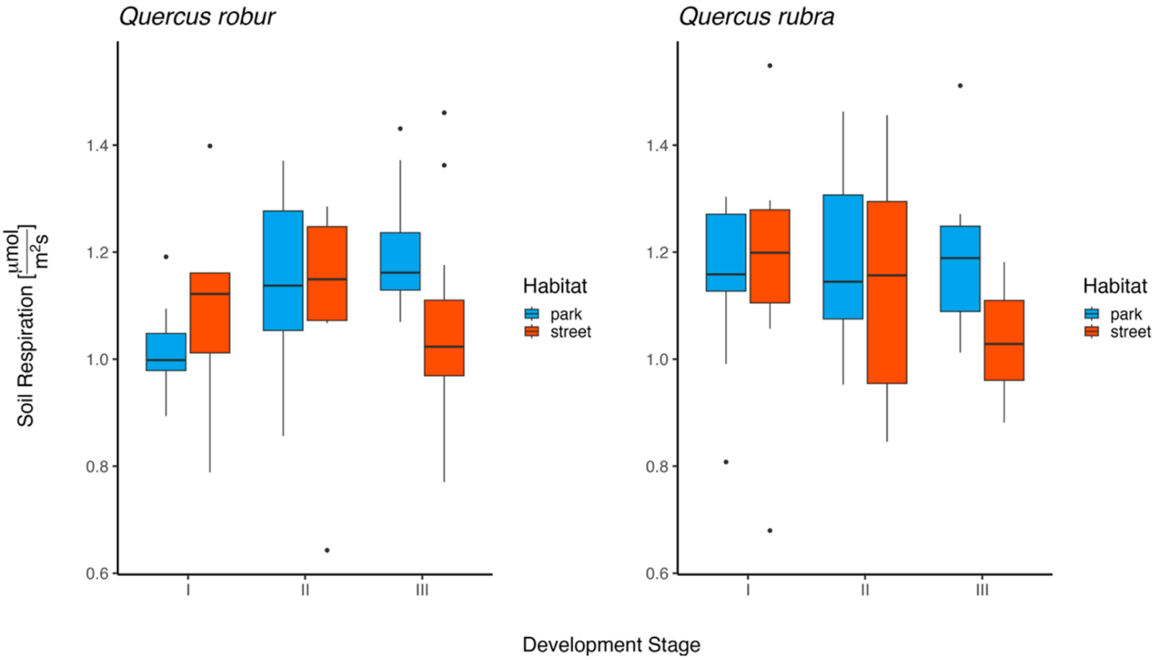

Fig. 5. Breakdown of median soil respiration differences between tree groups of *Quercus robur* and *Q. rubra*, across street and park locations, and different development stages. Outliers are shown as individual data points.

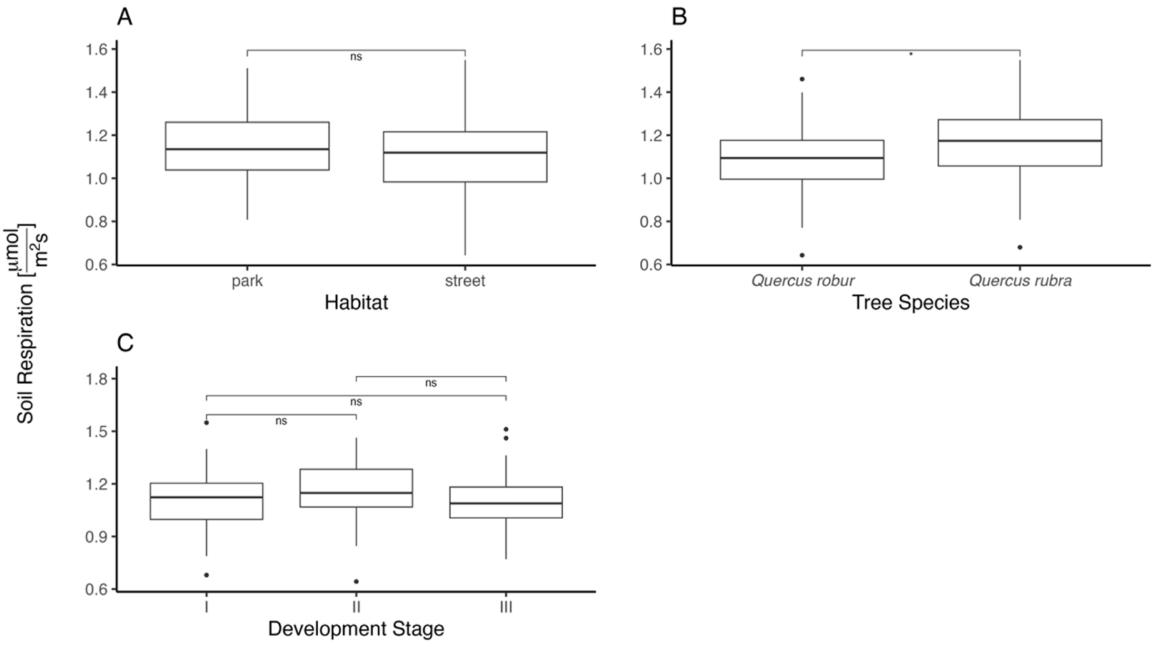

Fig. 6. Grouped differences in median soil respiration between (A) park and street trees, (B) *Quercus robur* and *Quercus rubra*, and (C) development stages I to III. Outliers are shown as individual data points. Statistically significant differences ($p < 0.05$) between groups are marked with an asterisk.

stage I, median impervious surface beneath canopy for development stage II and III differed from development stage I (Fig. 8). Mean SWC peaked on days following rain (Aug 15–16, mean SWC = 0.22 ± 0.01 m^3/m^3). Conversely, mean SWC was lower on days with high air temperature and PPFD (Aug 22–23, mean SWC = 0.02 ± 0.01 m^3/m^3). Mean T_s mirrored air temperature, peaking when air temperature and PPFD reached their peak (Aug 22–23, mean $T_s = 30.2 \pm 0.3$ °C).

A Tukey's HSD test revealed a statistically significant increase in average DBH with the development stage. To assess if DBH might better predict R_s than the development stage, a model replacing the development stage with DBH was created. However, the development stage model had a better fit and neither the development stage nor DBH explained a statistically significant amount of variance.

4. Discussion

4.1. Influence of habitat, Quercus robur, Quercus rubra, and development stage on soil respiration

Contrary to expectations, initial analyses of habitat, tree species, and development stage revealed no differences in R_s. However, linear mixed-effects model analysis indicated statistically significant interactions

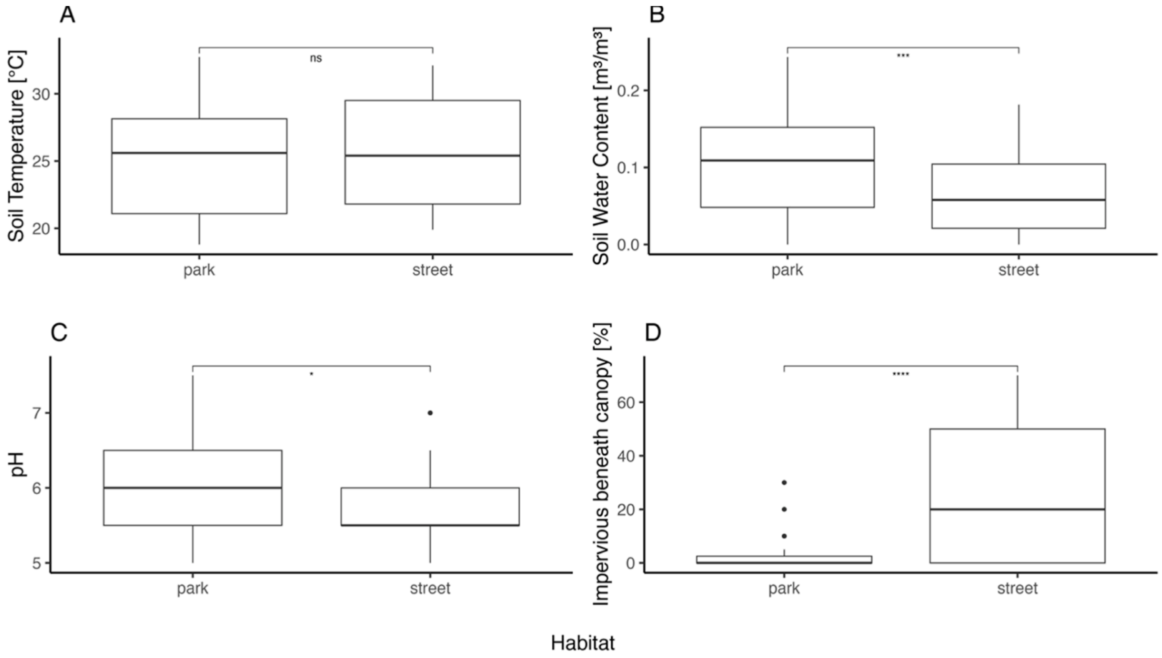

Fig. 7. Differences in A) soil temperature, B) soil water content, C) soil pH, and D) impervious surface area between park and street. These variables were identified as statistically significant predictors of soil respiration. Outliers are shown as individual data points. Statistically significant differences between groups are marked with an asterisk. * = p-value < 0.05 * ** = p-value < 0.001 * ** * = p-value < 0.0001.

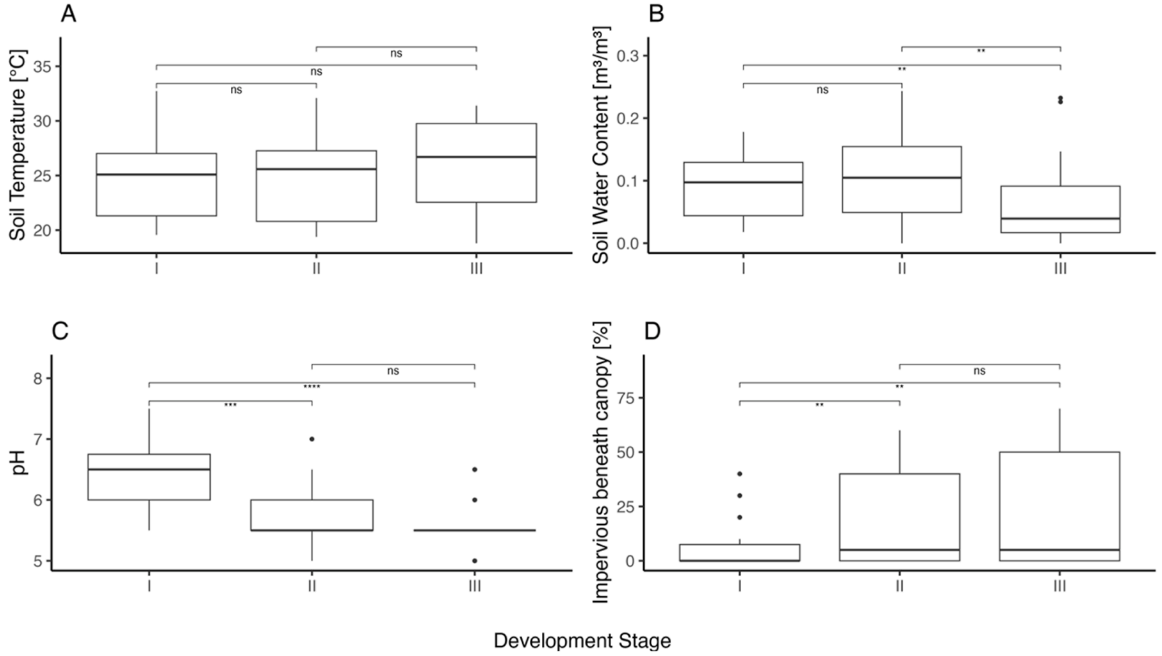

Fig. 8. Differences in A) soil temperature, B) soil water content, C) soil pH, and D) impervious surface area between development stages I to III. These variables were identified as statistically significant predictors of soil respiration. Outliers are shown as individual data points. Statistically significant differences between groups are marked with an asterisk. * = p-value < 0.05 * ** = p-value < 0.001 * ** * = p-value < 0.0001.

between these variables, with distinct R_s patterns across park and street sites among different development stages. The decrease of R_s became more significant with the progressing development stage for street trees of both species, confirming hypotheses 1 and 3 regarding street trees. Park R_s increased with the development stage for *Quercus robur* and stagnated for *Q. rubra*, contradicting hypothesis 3. This inconsistency aligns with conflicting literature: Saiz et al. (2006) reported peak R_s in young stands, plateauing after that, while Wu et al. (2020) found consistent R_s increases with progressing development stage and Kukumägi et al. (2017) identified no development stage differences. Similarly, studies on habitat contrast: Kukumägi et al. (2017) found no park–street R_s disparity, while Wu et al. (2016) reported decreasing R_s with increasing distance from sealed surfaces.

Kim and Yoo (2021) demonstrated that R_s in urban areas was an indicator of soil quality and had high predictive power for the growth of street trees. While the growth rate of the trees was not investigated in our study, the generally lower R_s values at street locations could also indicate poorer soil quality that may deteriorate further over time. This assumption is supported by the fact that the median soil pH values (also an indicator of soil quality (Kim and Yoo, 2021)) at street locations are lower for both *Quercus robur* and *Q. rubra* in development stages II (*Q. robur*: ΔpH $= -0.5 \pm 0.74$; *Q. rubra*: ΔpH $= -1 \pm 1.06$) and III (*Q. robur*: ΔpH $= -0.5 \pm 0.74$; *Q. rubra*: ΔpH $= -1 \pm 0.74$) compared to the median soil pH values at development stage I (*Q. robur*: pH $= 6 \pm 0.74$; *Q. rubra*: pH $= 6.5 \pm 0.74$). This could explain why the decrease in R_s values becomes increasingly significant for street trees as the development stage progresses since the differences in statistically significant predictors for R_s grow stronger with the advancing development stage (see Fig. 8).

We expected no differences in R_s between *Quercus robur* and *Q. rubra* due to litter removal by city management, potentially eliminating a major influence on soil nutrients, microbial communities, and the bacteria-to-fungi ratio (Zhang et al., 2020; Stanek and Stefanowicz, 2019). Nevertheless, our mixed-effects model analysis revealed a statistically significant difference in R_s between *Quercus robur* and *Q. rubra*. However, recent studies in forest ecosystems contradict the observed difference in R_s between *Quercus robur* and *Q. rubra*. Woś et al. (2023) analyzed the effects of *Quercus robur* and *Q. rubra* on soil physicochemical properties, microbial properties, and R_s in post-fire ecosystems in Poland. The study found no differences in R_s between the two species. However, *Quercus rubra* exhibited a higher C/N ratio compared to *Q. robur*. Similarly, Stanek and Stefanowicz (2019) reported higher C/N and C/P ratios in *Quercus rubra* litter, along with lower total soil microbial biomass and bacterial biomass under *Q. rubra* than under *Q. robur*. In contrast, fungal biomass showed no statistically significant difference between the two species. The overall structure of the microbial community in the organic soil horizon differed between *Quercus robur* and *Q. rubra*. This suggests that differences in litter quality and nutrient composition influence the composition of the soil microbial community, soil microbial biomass, and R_s rates (Eiland et al., 2001; Stanek and Stefanowicz, 2019).

We found no statistically significant differences in soil C/N ratios between *Quercus robur* and *Q. rubra* that would suggest variations in the soil microbial community structure due to differences in nutrient availability. However, further investigation is needed into precise C/N and P/N ratios, microbial community structure, and litter inputs to better understand how *Quercus robur* and *Q. rubra* influence R_s in urban environments. Trees can also influence R_s through root respiration, which contributes about 40 % to total R_s in urban deciduous forest ecosystems in Europe (Goncharova et al., 2020). Photosynthesis and assimilate translocation to roots are key drivers (Han et al., 2014, Högberg et al. (2001); Chen et al., 2014). While *Quercus robur* has a higher maximum assimilation rate than *Q. rubra* ($\sim$13 mol/m^{2}s *vs.* 11.5 mol/m^{2}s) (Morecroft and Roberts, 1999; Dyderski et al., 2020), *Q. rubra* has a larger leaf area index (3.1–4.5 m^2/ m^2 *vs.* $\sim$2.5 m^2/ m^2) (Dyderski et al., 2020; Bequet et al., 2011), potentially compensating for the lower assimilation rate. This suggests potentially higher gross primary production of *Q. rubra* leading to a greater availability of assimilates. The higher R_s values we measured could therefore be attributed to this increased availability of assimilates. Measuring gross primary production and root assimilate allocation would be necessary for confirmation.

4.2. Influence of environmental and neighbouring variables on soil respiration

4.2.1. Soil pH

R_s decreased with decreasing soil pH, aligning with previous findings (e.g. Li et al., 2018; Liang et al., 2013). Li et al. (2018) attributed this to lower root biomass, microbial activity, and respiration at lower soil pH.

In our data, soil pH differed statistically significant between park and street locations (park > street, $p < 0.01$) and development stage (development stage I > II = III, $p < 0.0001$). Saiz et al. (2006) reported decreasing soil pH with stand age in spruce forests, but not urban trees. Karvinen et al. (2024) found the opposite for park and street locations compared to this study. Wei et al. (2014) observed higher soil pH under and near sealed surfaces, also contrasting our findings. Viswanathan et al. (2011) hypothesized that higher CO_2 under impervious surfaces could lead to potential acidification but did not detect soil pH changes. Chen et al. (2016) reported declining soil pH with nitrogen input, but we did not find a relation between nitrogen contents and soil pH.

The lower soil pH at street locations could explain the trend of lower R_s. Additionally, the statistically significant difference in R_s between park and street for development stage III ($p < 0.05$) aligns with the lower soil pH in development stage II and development stage III ($p < 0.0001$).

In addition to R_s, soil pH is also an indicator of soil quality (Kim and Yoo, 2021). Similar to R_s, soil pH values tend to be lower at street locations compared to park locations. This further supports the hypothesis that soil quality might be poorer at street locations than at park locations.

4.2.2. Soil temperature and soil water content

R_s and T_s showed a weak negative correlation ($\rho = -0.23$, $p < 0.05$) and T_s was not a statistically significant predictor for R_s. T_s also lacked statistically significant differences between habitats, the two *Quercus* species, and development stages, despite studies like Karvinen et al. (2024) finding higher T_s in street locations due to increasingly sealed surfaces.

Karvinen et al. (2024) also found no T_s effect on R_s in urban ecosystems, aligning with Borges Pinto et al. (2018) in tropical forests. This contrasts with studies by Chen et al. (2014), Jia et al. (2018), and others who reported strong T_s–R_s relationships. However, T_s becomes a statistically significant predictor for R_s when considering its interaction with SWC, as observed by Miao et al. (2004) and others. In literature, the correlation between R_s and T_s also differed depending on SWC. Positive correlations occurred at high SWC, while no relationship existed at low SWC (Chang et al., 2016; Saiz et al., 2006). Similar patterns emerged in our data. For R_s measurements with SWC below the average minus one standard deviation, no correlation with T_s was found. Conversely, measurements with SWC above the average plus one standard deviation showed a moderate positive correlation ($\rho = 0.55$, $p = 0.171$). These observations suggest that water availability, as noted by Goncharova et al. (2020) and Tao et al. (2016), might be a key determinant of R_s for urban trees during hot summers.

Supporting the hypothesis, SWC explained a statistically significant portion of R_s variance (unlike T_s) in our study. R_s and SWC correlated weakly ($\rho = 0.25$, $p < 0.0001$), aligning with prior studies (e.g. Jassal et al., 2005; Zhao et al., 2016) with few exceptions (Karvinen et al., 2024; Borges Pinto et al., 2018). Karvinen et al. (2024) attributed the missing SWC effect on R_s to uniform moisture conditions. In contrast, we found higher SWC values in parks than in streets ($p < 0.001$) and lower SWC with the progressing development stage. Lower SWC in streets is likely due to higher radiation exposure leading to higher evapotranspiration (more exposed leaves, higher photosynthetic rates) (He et al., 2014); direct soil water evaporation due to solar radiation exposure (He et al., 2014); higher proportion of impervious surfaces under the canopy, restricting water infiltration (Savi et al., 2015). We found higher crown light exposure ($p < 0.01$) and imperviousness for street locations, aligning with results from the literature.

Lower SWC in development stage III could also be due to higher evapotranspiration and interception from larger crown projection areas (CPA) (Pflug et al., 2021; Goodwin et al., 2006). However, SWC did not differ between development stage I and II despite a larger crown projection area in development stage II (ΔCPA $= 165 \pm 17$ m^2).

Assuming water availability dictates R_s, street trees should have lower R_s than park trees. While no overall park-street difference was observed, a trend of lower R_s in streets emerged with increasing development stage, eventually becoming statistically significant in development stage III.

4.2.3. *Impervious surface beneath canopy*

ANOVA identified imperviousness beneath the canopy as the variable explaining the most variance in R_s. Regression analysis confirmed that R_s decreased with increasing soil sealing. Literature on the relationship between impervious surface beneath a canopy and R_s is mixed. While Wei et al. (2014) and Wu et al. (2016) observed decreasing R_s with increasing sealing, Fini et al. (2016) found the opposite. Interestingly, Fini et al. (2016) reported higher R_s for trees with higher imperviousness beneath the canopy but also observed stress indicators in trees with high imperviousness beneath the canopy. We, therefore, conclude that a high degree of impervious surfaces beneath the canopy, in addition to the previously discussed influence on soil water content and thus R_s, may also lead to reduced assimilation rates due to stress induced by the impervious surfaces. Assimilation rates are a key driver of R_s (Han et al., 2014; Högberg et al., 2001), and thus lower assimilation rates could result in lower R_s. This could be especially true for trees in advanced development stages, as we observed a higher proportion of impervious surfaces beneath their canopies as well as generally lower R_s values.

4.2.4. *Use of planting substrate and potential impact on soil respiration*

Human management practices influence urban tree health. In Karlsruhe, Germany, the City Department of Gardens and Green Spaces manages existing trees and plantings. They use technical substrates for street trees, meeting FLL guidelines (FLL, 2010). Open spaces like parks typically rely on existing soil conditions with amendments like topsoil, sand, compost, or lava, depending on project goals. For example, some managers prioritize topsoil for grass establishment and potential drought tolerance. Green areas might use a 20 % lava and 80 % green waste compost mix. This variable topsoil composition across the city could influence factors like soil pH, water holding capacity, and nutrient availability, impacting plant, fungal, and microbial communities (Tong et al., 2018; Skirde, 1974).

While no general statistically significant differences in R_s were found between park and street locations, observed variations in soil pH and SWC could be explained by these management practices.

4.3. *Limitations of this study*

We employed a cross-sectional study design, conducting measurements at each tree at a single point in time. Although cross-sectional studies are efficient and can identify associations, they make it difficult to establish causal relationships (Mann, 2003). However, the observed relationships can serve as a foundation for future studies that can delve deeper into these findings.

This study encountered limitations inherent to field measurements in urban environments. Beyond soil disturbances prior to soil respiration measurements (e.g., due to PVC collar removal or curious passersby), equipment theft led to delays in the experimental procedure and schedule. Researchers should account for these factors in both experimental design and scheduling.

Furthermore, the classification into development stages was found to be limited. Trees in development stage I were relatively similar in size (min. DBH = 6 cm, max. DBH = 28 cm), while the range of DBH values was much greater in development stage II (min. DBH = 23 cm, max. DBH = 145 cm) and III (min. DBH = 91 cm, max. DBH = 156 cm), making comparisons more difficult. While DBH as a continuous proxy for age in a mixed effects model had a statistically significant effect on R_s, it led to a poorer model fit.

The limited availability of accessible information regarding the age of the tree at planting, and prior treatments of the site before tree planting restricts our ability to interpret the results. More precise information would have facilitated drawing conclusions about why differences exist between the sites and development stages, as well as providing recommendations for future plantings and soil treatments. If responsible authorities could collect and store such information in an accessible manner, they should consider doing so for future purposes.

5. Conclusion and outlook

We observed statistically significant differences in R_s both between *Quercus robur* and *Q. rubra* and between tree development stages, but not between habitats (park vs. street). Soil pH, the impervious surface beneath the canopy, SWC, and the interaction between SWC and T_s all explained statistically significant share of variance in R_s. Park sites had higher SWC, soil pH, and lower impervious surface beneath the canopy, all factors promoting R_s and potentially explaining the generally higher R_s observed in parks. SWC and soil pH decreased while the impervious surface beneath the canopy increased with the progressing development stage. This trend, known to impede R_s, aligns with the lower R_s observed for street trees in development stage III, where the negative effects of street location and mature development stage overlap. The combination of generally lower R_s and soil pH values at street locations, along with the varying application of soil amendments between the sites, suggests that soil quality may be poorer at street locations compared to park locations. Despite the statistically significant difference in R_s between the two *Quercus* species, no differences in soil pH, SWC, or impervious surface beneath the canopy were found, which could possibly be attributed to greater availability of assimilates by *Q. rubra* or differences in soil microbial and fungal community. Detailed investigations are required to verify these results and assumptions. These should include studies on nutrient inputs, the effect of leaf litter on urban microbial communities, differences in gross production and carbon allocation between *Quercus robur* and *Q. rubra*, and time-series analyses to elucidate cause-and-effect relationships.

The question of whether one of the two *Quercus* species is superior to the other in urban areas is not straightforward. Based solely on our results (specifically, no difference in soil pH between species and higher soil respiration under non-native *Q. rubra*), it could be inferred that soil quality and conditions for R_s, and active soil life might be better under *Q. rubra* or that assimilation rates might be higher. Both aspects would favor *Quercus rubra*. However, other studies have shown that native *Quercus robur* provides a greater diversity of microhabitats than *Q. rubra* (Laux et al., 2022). Therefore, the choice of species will depend mainly on the specific goals of the urban greening project and the needs of the city's inhabitants.

CRediT authorship contribution statement

Blumenstein Kathrin: Writing – review & editing, Supervision, Conceptualization. **Totzki Jan:** Writing – review & editing, Writing – original draft, Visualization, Validation, Software, Methodology, Investigation, Formal analysis, Data curation. **Saha Somidh:** Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Software, Resources, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization.

Declaration of Competing Interest

The authors declare that they have no conflict of interest. Permissions for fieldwork were obtained from the Karlsruhe City Horticulture Office. Data can be made available after the research article is published.

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