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# The urban heat island accelerates litter decomposition through microclimatic warming in temperate urban forests

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1 The urban heat island accelerates litter decomposition through microclimatic  
2 warming in temperate urban forests

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39

## 40 Abstract

41 Forests worldwide are experiencing high levels of fragmentation, with especially important consequences for  
42 ecosystems bordering urbanized areas. Urban forests experience local warming due to the urban heat island  
43 associated with built-up area which affects their biodiversity and ecosystem functioning. One of the key ecosystem  
44 functions affecting carbon and nutrient cycling in forests is litter decomposition, a process driven by the local  
45 microclimate (temperature and soil moisture). Our aim is to clarify the impact of the urban heat island on litter  
46 decomposition in urban forests.

47 We studied soil microclimate and litter decomposition in six urban forests across Europe and along local gradients  
48 from the urban forest edge to the forest interior. We expected to find edge effects altering the soil microclimate.  
49 To unequivocally quantify decomposition independent from local forest composition and litter quality, we used  
50 standardized green tea and rooibos tea litterbags. We then assessed the effects of built-up area and forest area in  
51 the surroundings on the soil microclimate for different spatial scales. Secondly, we determined the role of the soil  
52 microclimate and other environmental drivers for litter decomposition.

53 Soil characteristics were only driving green tea and not rooibos tea decomposition. On the contrary, soil  
54 temperature was an important driver of decomposition for both tea types. Higher soil temperatures resulted in  
55 faster rates of litter decomposition and were related to the proximity to the forest edge, but even more strongly to  
56 the proportion of built-up area in the landscape.

57 Via structural equation modelling we detected cascading effects of the urban heat island on litter decomposition.  
58 Such changes in litter decomposition due to urban heat have the potential to alter the soil food web, nutrient  
59 cycling and carbon drawdown in urban forests, and could result in significant interactions between urbanisation  
60 and ongoing climate change.

## 61 Keywords

62 Forest edge, litter decomposition, microclimate, soil temperature, urban forest, urban heat island

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## 70 **Introduction**

71 Worldwide, the area covered by natural and rural areas is decreasing at the expense of urban settlements. This  
72 process of land-use change is driven by urbanisation, the increasing proportion of humans living in cities, towns  
73 and agglomerations (UN 2019). Currently already 55% of humans live in cities and this is expected to further  
74 increase to 68% by 2050 (UN 2019). Urban land use shapes the environment and impacts organisms in many  
75 different ways, for instance, through air pollution, noise, light pollution and high amounts of impervious surfaces  
76 (Baklanov et al. 2016, Miller and Hutchins 2017, van den Bosch and Ode Sang 2017, Aulsebrook et al. 2018, Jerem  
77 and Mathews 2021). Furthermore, high proportions of imperviousness and a lack of vegetation contribute to the  
78 Urban Heat Island (UHI), which is defined as higher air temperatures in a city than in its surrounding countryside  
79 (Oke 1982). Temperatures increase within urban areas due to several mechanisms, such as heat storage in buildings  
80 and pavements and reduced transpiration due to lack of vegetation (Oke 2002, Kleerekoper et al. 2012).

81 Urban forests can lower urban air and soil temperatures through shading and evapotranspiration (Rakoto et al.  
82 2021, Wang et al. 2021b). Urban forests generally refer to all forest and tree resources in (and close to) urban areas  
83 (Konijnendijk 2003). They are increasingly brought forward as a nature-based solution for urban heat (van den  
84 Bosch and Ode Sang 2017, Ziter et al. 2019, Wang et al. 2021b), which can be of vital importance for citizens given  
85 the higher mortality risks associated with heat stress (Smoyer et al. 2000, Wolff et al. 2021). Urban forests provide  
86 many more ecosystem services, such as reducing the effect of air pollution which lowers the prevalence of  
87 respiratory diseases (Nowak et al. 2006, Nowak et al. 2018), or the attenuation of traffic noise (Dzhambov and  
88 Dimitrova 2014, Ow and Ghosh 2017). Urban forests also provide recreational space for citizens which stimulates  
89 physical exercise (Tesler et al. 2018, Remme et al. 2021), lowers stress levels and contributes to people's well-being  
90 (Tyrväinen et al. 2014, Carrus et al. 2015, Aerts et al. 2022). Urban forests can have a high intrinsic biodiversity  
91 value and provide valuable habitat for fauna and flora (Enedino et al. 2018, Rega-Brodsky et al. 2022).

92 However, the urban environment puts pressure on urban forests and the many ecosystem services they provide  
93 (Referowska-Chodak 2019). For example, the UHI exacerbates drought stress and pest susceptibility of urban trees  
94 (Meineke et al. 2013, Dale and Frank 2017, Zipper et al. 2017). The UHI effect may also foster spreading of alien  
95 plant species through garden escapes and by serving as a stepping stone, increasing the risk posed by invasive alien  
96 species (Kowarik 2011, Aronson et al. 2016, Géron et al. 2021). Furthermore, temperature has a strong effect on  
97 many natural processes in nutrient and carbon cycling, such as primary productivity and soil organic matter  
98 decomposition or accumulation (Conant et al. 2011, Wu et al. 2011, Zhang et al. 2020). Therefore, it could be

99 expected that urban heat affects urban forest functioning. Litter decomposition, for example, is an essential  
100 component of an ecosystem's carbon and nutrient cycles (Coûteaux et al. 1995). Dead organic matter is broken  
101 down by decomposers, using the energy and matter to grow and making nutrients available again for uptake by  
102 plants (Swift et al. 1979, Handa et al. 2014). Temperature and moisture are key determinants of litter decomposition  
103 rates, in addition to the quality of litter and the decomposer community (Coûteaux et al. 1995, Aerts 1997, Tresch  
104 et al. 2019). Generally, decomposition rates increase with rising temperatures and with higher moisture availability  
105 given their positive effects on microbial activity (Kirschbaum 1995, Zhang et al. 2008), which has already been  
106 included in (global) carbon cycling models (Coleman and Jenkinson 1996, Bonan et al. 2013). However, the  
107 response of the microbial community to variation in climate depends strongly on the factors limiting microbial  
108 activity (Kirschbaum 1995, Berg and McClaugherty 2008). For example, if soil moisture falls below a certain  
109 threshold, it limits microbial activity and increases in soil moisture will then increase decomposition rates, whereas  
110 in wet environments higher soil moisture will not necessarily result in accelerated litter decomposition (Prescott  
111 2010).

112 Previous studies have reported higher soil or forest floor temperatures in urban vs. rural forests due to the UHI  
113 effect (Pouyat et al. 1997, Enloe et al. 2015). Given the importance of temperature for litter decomposition, it  
114 could be expected that the UHI effect accelerates decomposition in urban forests. Several studies found higher  
115 decomposition rates in urban vs rural green areas and attributed this to the UHI (Pouyat et al. 1997, Pouyat and  
116 Carreiro 2003, Tresch et al. 2019), its positive association with microbial activity (Tresch et al. 2019), and  
117 breakdown by (exotic) earthworms (Pouyat et al. 1997, Pouyat and Carreiro 2003). However, most of these studies  
118 had not quantified soil temperatures in the field (Pouyat et al. 1997, Pouyat and Carreiro 2003, Tresch et al. 2019)  
119 or found no significant difference in soil temperature (Melliger et al. 2017). Other studies found no conclusive or  
120 even contrasting results when studying litter decomposition along rural-to-urban gradients and attributed these to  
121 drought effects, changes in microbial communities or heavy metal pollution (Pavao-Zuckerman and Coleman  
122 2005, Enloe et al. 2015, Meyer et al. 2020, Fung et al. 2022). Additionally, one study in Singapore found faster litter  
123 decomposition rates of teabags in soils of urban forests vs. beneath urban trees and related this to lower  
124 invertebrate (termite) activity and a tendency for lower infiltration capacities in the soil beneath urban trees vs. in  
125 the urban forest (Fung et al. 2022) Up till now, there is no conclusive answer on the role of the UHI in driving  
126 litter decomposition rates in urban forests.

127 Edge effects can be assumed to be substantial in urban forests given the exceptionally strong abiotic and biotic  
128 contrasts between forest and matrix (Hamberg et al. 2009). In general, temperate forest edges are characterised by  
129 higher temperatures and temperature variability (Chen et al. 1999, Meeussen et al. 2021a) and lower soil and litter  
130 moisture content (Matlack 1993, Gehlhausen et al. 2000). Additionally, soil pH differs between forest edges and  
131 interiors (Honday et al. 2002, Wuyts et al. 2013) and temperate forest edges have higher nutrient and carbon stocks  
132 compared to forest interiors (Remy et al. 2016, Meeussen et al. 2021b). Furthermore, microbial communities can  
133 differ between forest edges and interiors (Kageyama et al. 2008, Malmivaara-Lämsä et al. 2008, Remy et al. 2018b),  
134 as well as the abundance and activity of soil macro- and mesofauna (Riutta et al. 2016, De Smedt et al. 2019).  
135 Generally, edge effects reach distances up to 50 m for abiotic and vegetation effects, and up to 100 m for  
136 invertebrates (Ries et al. 2004). Many of these changes affect litter decomposition in different ways, resulting in  
137 either faster or slower decomposition rates at forest edges depending on their interaction or which factor is most  
138 important in the local environmental context (Malmivaara-Lämsä et al. 2008, Riutta et al. 2012, Remy et al. 2018a).

139 Here we assess how urbanization affects the soil microclimate and litter decomposition in six large urban forests  
140 across Europe and along local gradients from the urban edge to the forest interior. In doing so, we specifically aim  
141 to quantify the effect of urbanisation on the soil microclimate in the forest understory and how this potentially  
142 cascades to affect litter decomposition rates of reference litter types. We hypothesize that: (1) the understory soil  
143 microclimate is affected by the UHI, increasing mean soil temperatures values towards the urban forest edge and  
144 (2) that these increased soil temperatures accelerate litter decomposition.

145

146

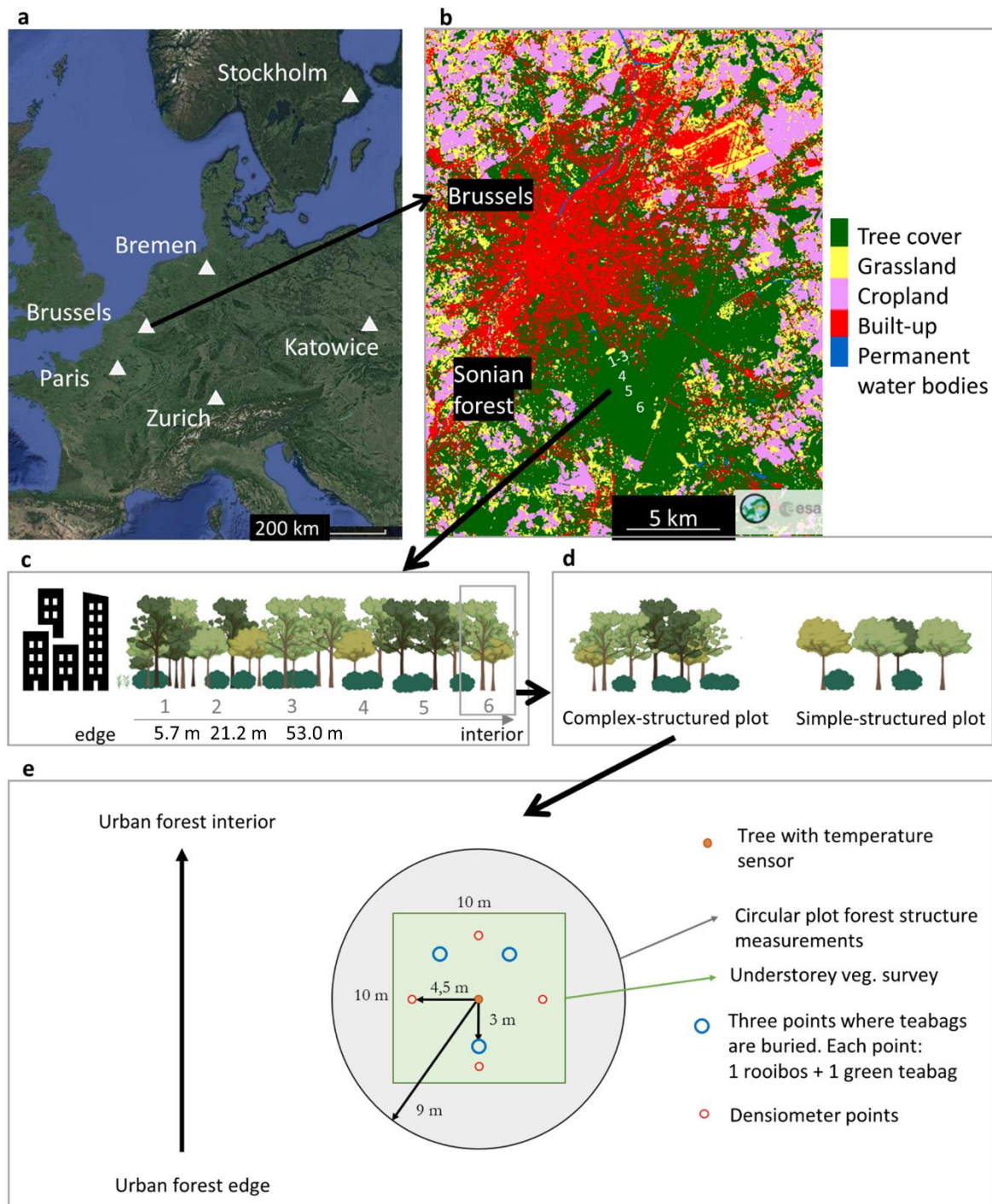
## 147 Materials & methods

### 148 Study regions and design

149 We studied six urban forests close to large European cities, along a 1,400-km long macroclimatic gradient from  
150 Stockholm (Sweden) in the north, to Zurich (Switzerland) in the south (Figure 1). These urban forests differ  
151 strongly in temperature (monthly average minima of coldest month between  $-4.9$  and  $1^{\circ}\text{C}$ , monthly average  
152 maxima of warmest month between  $22.4$  and  $25^{\circ}\text{C}$ ), but also in mean annual precipitation (from  $561$  mm in  
153 Stockholm up to  $1,107$  mm in Zurich; precipitation and temperatures are 30-year averages from Terraclimate 1981–  
154 2010, resolution of  $\sim 4$  km (Abatzoglou et al. 2018); Table 1). The UHI effect of these six studied European cities is  
155 particularly pronounced, with urban-to-rural land surface temperatures differences of summer daytime maxima  
156 from  $+1.7$  to  $+2.7^{\circ}\text{C}$  (CIESIN 2016). We searched for large, deciduous forests close to these cities, dominated by  
157 pedunculate oak (*Quercus robur*) to ensure a minimum of comparability between our selected urban forests.  
158 Pedunculate oak is one of the most dominant tree species of temperate deciduous forests in Europe (Barbati et al.  
159 2014), with high ecological and economic importance. Indeed, pedunculate oaks support a high number of  
160 associated species, resulting in rich woodland biodiversity (Eaton et al. 2016, Mölder et al. 2019), and provide high  
161 economical value (Mölder et al. 2019, Şöhretoğlu and Renda 2020). More environmental variables of the urban  
162 forests are given in Table 1.

163





164

165 **Fig. 1 Design of the experimental setup, with an example map from Brussels.** (a) Six urban forests were studied  
 166 across Europe (background map from GoogleMaps 2022). (b) Each urban forest had twelve plots situated in six pairs  
 167 across a transect from the forest edge closest to the city (1) towards the forest core (6). The map shows the plot locations  
 168 for Sonian forest, an urban forest in Brussels, Belgium (background land cover map from European Space Agency  
 169 WorldCover (Zanaga et al. 2021)). (c) The three first plot pairs were located at approximately fixed distances from the  
 170 forest edge: on average 5.7, 21.2, 53 m and the next three were placed at set intervals. (d) Within each plot pair, we  
 171 contrasted one plot with a more complex forest structure vs. one plot with a rather simple forest structure. (e) Overview  
 172 of plot and sampling points. Figures c and d created with Biorender.com.

173 **Table 1. Environmental variables for each forest and number of useable teabags given per region.** Minimum  
174 and maximum temperature of coldest and warmest month, respectively, mean annual precipitation (Terraclimate 1981–  
175 2010, resolution of ~4 km (Abatzoglou et al. 2018)), and the precipitation recorded during the decomposition periods  
176 for each of the sites based on daily data from the Global Historical Climatology Network (NCEI 2021) (closest weather  
177 stations available in the dataset were Stockholm, Bremen, Uccle, Balice, le Bourget and Zurich). Additionally, latitude,  
178 longitude, dominant tree species in study plots, soil texture (SoilGrids, resolution 250 m (Hengl et al. 2017) and texture  
179 calculated with USDA texture triangle), size of the urban forest (derived from the European Local Climate Zone map  
180 produced by Demuzere *et al.* (2019)), distance between urban forest edge and interior plots (for simple and complex  
181 structured plots respectively) and size of the city population are listed (citypopulation.de , Eurostat 2021). Also an  
182 estimation of the land surface urban heat island is given per city based on the global UHI dataset from the Center for  
183 International Earth Science Information Network (difference in average summer daytime maximum land surface  
184 temperature between the urban area and buffer area) (CIESIN 2016). The number of usable teabags is given per region  
185 and as the percentage of the incubated amount. Several teabags were not usable due to loss in the field or severely  
186 ruptured mesh.

	<i>Lill-Jansskogen</i> (Stockholm, Sweden)	<i>Bürgerpark</i> (Bremen, Germany)	<i>Sonian Forest</i> (Brussels, Belgium)	<i>Lasy</i> <i>Murkowskie</i> (Katowice, Poland)	<i>Forêt</i> <i>domaniale de</i> <i>Montmorency</i> (Paris, France)	<i>Hönggerberg</i> (Zurich, Switzerland)	<i>Total</i>
<i>Latitude (°)</i>	59.3	53.1	50.8	50.2	49.0	47.4	
<i>Longitude (°)</i>	18.1	8.8	4.4	19.0	2.3	8.5	
<i>Elevation (m above sea level)</i>	37	10	115	310	177	520	
<i>Mean daily min temperature coldest month (°C)</i>	-4.5	-0.8	0.4	-4.9	1.0	-1.7	
<i>Mean daily max temperature warmest month (°C)</i>	22.4	23.2	23.5	24.2	24.4	25	
<i>Mean annual precipitation (mm)</i>	561	698.5	850	726	662	1107	
<i>Precipitation during teabag decomposition period (mm)</i>	138.4	152.7	227.1	190.2	133.6	189.5	
<i>Dominant tree species</i>	<i>Quercus robur,</i> <i>Fraxinus excelsior,</i> <i>Acer platanoides</i>	<i>Quercus robur,</i> <i>Fagus sylvatica</i>	<i>Quercus robur,</i> <i>Fagus sylvatica</i>	<i>Quercus robur,</i> <i>Quercus rubra,</i> <i>Fagus sylvatica</i>	<i>Quercus robur,</i> <i>Castanea sativa</i>	<i>Quercus robur</i>	
<i>Soil texture sand-silt-clay (g/kg)</i>	430-320-250 (loam)	530-310-160 (sandy loam)	120-690-190 (silt loam)	250-530-220 (silt loam)	170-610-220 (silt loam)	380-360- 260 (loam)	
<i>Size urban forest (ha)</i>	132	170	6139	8093	2686	360	
<i>Distance edge – interior plots (m)</i>	650/620	2450/2450	3150/3260	290/690	2350/2350	440/515	
<i>City population size (nr inhabitants)</i>	950,000	570,000	1,200,000	209,000	2,100,000	410,000	

<i>Urban heat island (°C)</i>	2.5	1.7	2.1	2.1	2.6	2.4	
<i>Nr of usable teabags</i>	48	48	59	60	32	57	304
<i>Green tea</i>	25	24	32	29	14	28	152
<i>Rooibos tea</i>	23	24	27	31	18	29	152
<i>Usable teabags (%)</i>	67	67	82	83	44	79	70

187

188 In each of the six urban forests, we established twelve circular plots (9-m radius around a central tree) along a  
189 transect from the urban forest edge (defined as the hypothetical line of tree stems at the edge closest to the urban  
190 area) into the core of the urban forest. These transects were 290 m to 3 km long and covered the generally  
191 acknowledged distance of edge effects (up to 100 m) and extended even further given the larger spatial scale of  
192 UHI effects, e.g. >50 m up to several km (Luo and Li (2014), Estoque et al. (2017), Ziter et al. (2019)). In each  
193 transect, plots were established in six pairs. The first, second and third plot pair were located at approximately  
194 fixed distances of on average 5.7, 21.2 and 53 m from the forest edge. The sixth pair was located in the forest core,  
195 while the fourth and fifth plot pair at intermediate distances (Figure 1). Within each plot pair, one plot had a  
196 complex forest structure (i.e. high canopy cover; median tree and shrub cover being 88.5% and 37.5%, respectively;  
197 with multiple tree layers) and the other had a simpler forest structure (i.e. more open canopy; median tree and  
198 shrub cover being 70% and 15%, respectively; with one tree layer). These differences in forest structure between  
199 paired plots originated from management actions in the past, natural disturbances or different land-use histories,  
200 and were not created by us but carefully looked for during preparatory field visits. The contrast in stand structure  
201 allowed us to study differences in forest microclimate (Zellweger et al. 2019, Meeussen et al. 2021a) within each of  
202 the six pairs. The distance between paired plots depended strongly on the region, but was on average 441 m (range  
203 21–1772 m).

204 To assess the impact of land cover on the soil microclimate and litter decomposition rates, we used the ESA  
205 WorldCover map (10 m resolution, eleven land-use classes) (Zanaga et al. 2021). We calculated the fraction of land  
206 area classified as built-up area or forest area around each plot in buffer circles with increasing radii (50, 100, 200,  
207 400, 800, 1600 and 3200 m) as different types of land use can indeed affect microclimates at different spatial scales  
208 (Ziter et al. 2019). Built-up area is defined as land covered by buildings (both residential and industrial), roads,  
209 other man-made structures impermeable to water; forest cover is defined as any area dominated by trees with a  
210 cover of 10% or more (Product User Manual ESA WorldCover map).

211 The differences in forest structure in the study setup were expected to strongly affect the forest and soil  
212 microclimate (Zellweger et al. 2019, Meeussen et al. 2021a). To take this into account, we determined the  
213 percentage canopy cover with a convex spherical densiometer (Baudry et al. 2014). Canopy cover was calculated  
214 as the average of four densiometer readings in every plot, one in each cardinal direction at a distance of 4.5 m from  
215 the central tree (Figure 1).

## 216 Teabags as proxy for litter decomposition

217 To estimate litter decomposition rates with a substrate of standardized litter quality, we used standard Lipton  
218 teabags with green and rooibos tea. Green tea is easily decomposable (comparable with high quality litter from, for  
219 example *Sambucus nigra*), whereas rooibos tea contains more lignin and has a higher C:N ratio, which makes its  
220 decomposition slower (comparable with low quality litter, such as from *Quercus robur*; (Keuskamp et al. 2013). The  
221 bags have a mesh size of 0.25 mm allowing both microorganisms (bacteria and fungi) and mesofauna (e.g.  
222 springtails) to access the tea, but not macrofauna (e.g. earthworms; (Keuskamp et al. 2013)).

223 Within each plot, we buried three pairs of green and rooibos teabags in the topsoil (approximately 8 cm depth) at  
224 3 m from the central tree during August or September 2020 (Figure 1). The teabags were in the soil for  
225 approximately 90 days (Table S1), after which they were dug up and dried at 70 °C for 48 h. Ingrown roots were  
226 carefully removed before weighing the remaining tea material (to 0.1 mg,  $Tea\ mass_{end}$ ), combusting at 550 °C for  
227 4 - 6 h and re-weighing the left-over mineral ash content ( $Ash\ mass_{end}$ ). We also determined the weight of empty  
228 bags and the ash content of five non-incubated teabags for both tea types ( $Ash\ mass_{start\ ref}$ ,  $Bag\ mass_{start\ ref}$ ) and  
229 weighted all teabags ( $Teabag\ mass_{start}$ ) before incubation to calculate initial ash-free dry mass. Finally, the litter  
230 decomposition rate was estimated as percentage dry mass of ash-free content lost during incubation (equation 1).  
231 We initially buried 432 teabags (216 green and 216 rooibos; 12 plots \* 6 cities \* 3 teabags \* 2 tea types), from which  
232 304 were retrieved and usable. Others were lost in the field (e.g. due to wild boar), ripped open by ingrowing roots  
233 or mice or fully overgrown by fungi (Table 1). We used the ash-free mass loss (equation 1) as a proxy for litter  
234 decomposition rates.

$$235 \text{ Ash-free mass loss \%} = 1 - \frac{Tea\ mass_{end} - Ash\ mass_{end}}{Teabag\ mass_{start} - Ash\ mass_{start\ ref} - Bag\ mass_{start\ ref}} \quad \text{equation 1}$$

236

## 237 Environmental drivers

238 To explain variation in litter decomposition rates, we assessed three important sets of drivers: (1) the soil  
239 microclimate (soil temperature); (2) the chemical soil properties (pH, C content, C:N ratio and soil nutrient regime);  
240 and (3) the litter quality of the tree and shrub layer in all forests. Additionally, given the known **edge effect** on soil  
241 characteristics and microclimate (Remy et al. 2018b, Meeussen et al. 2021a), we derived the shortest distance to a  
242 forest edge (not necessarily the urban forest edge) for each plot from Google Maps (Google n.d.).

243 **Soil temperature** was measured with microclimate soil loggers (Lascar EL-USB-1, range of -30 to +80 °C,  
244 resolution of 0.5 °C) buried at a soil depth of c. 5 cm (after litter removal) near the north side of the central tree  
245 and in a plastic tube for protection against moisture. We calculated the number of **growing degree hours** for each  
246 plot with a threshold value of 0 °C (which is a general threshold for decomposition (Portillo-Estrada et al. 2016))  
247 to obtain a cumulative measure of the temperature during incubation of the teabags, excluding freezing periods.  
248 The equation of Lindsey and Newman (1956) was used based on daily maximum and minimum temperatures. In  
249 17 plots (out of 72), soil loggers were dug up by wild boar or disturbed. For these plots, growing degree hours of  
250 soil temperature were estimated based on a linear regression with the growing degree hours of air temperature  
251 measured by a second logger attached to the tree trunk at a height of three metres (soil temp (gdh) = 7407.1743 +  
252 0.7384 \* air temp (gdh) with R<sup>2</sup>= 0.94, Figure S1, more details on methodology in appendix).

253 **Soil moisture** was measured gravimetrically at the start and end of the incubation period by taking a soil sample  
254 of the topsoil (0–10 cm depth) and drying it to constant weight at 40 °C. We preferred gravimetric soil moisture  
255 measurements as those are consistent across a range of soil types and textures, in comparison to electromagnetic  
256 sensors, which needs multiple soil-specific calibrations (Vaz et al. 2013, Sui et al. 2019). Nevertheless, we deemed  
257 two soil moisture measurements per site too few to realistically represent soil moisture conditions during the teabag  
258 incubation period of three months. Soil moisture was therefore excluded from the analysis.

259 For **chemical soil** analyses, five subsamples were taken at a depth of 0–10 cm per plot and pooled together. The  
260 samples were dried to constant weight at 40 °C and for at least 48 h, ground and sieved over a 2 mm mesh. Then,  
261 they were analysed for pH-H<sub>2</sub>O by shaking a 1:5 ratio soil/H<sub>2</sub>O mixture for 5 min at 300 r.p.m. and measuring  
262 with an Orion 920A pH meter with a Ross sure-flow 8172 BNWP pH electrode model (Thermo Scientific Orion,  
263 USA). Carbon and nitrogen content were determined by sample combustion at 1150 °C and gas measurements by  
264 a thermal conductivity detector in a CNS elemental analyzer (vario Macro Cube, Elementar, Germany).

265 To account for potential variation within urban forests in **humus type** (mor, moder, mull), we estimated the **soil**

266 **nutrient regime** for each plot, which can be defined as a composite gradient of several chemical soil characteristics  
267 broadly related to the humus type (Wilson et al. 2001). Rogister (1978) developed a method to assess soil nutrient  
268 regimes based on vegetation indicator values. For this reason, vegetation surveys of the understorey were made in  
269 early to mid-summer 2021 in a 10 m by 10 m square centred around the central tree in the plot. For each species,  
270 the percentage ground cover was visually estimated. We extracted Ellenberg indicator values for the species with  
271 the ‘TR8’ package (Bocci 2015) from the Ecoflora database (Fitter and Peat 1994). When multiple values were  
272 found, the most frequent value was used. For species which were missing indicator values, we first used Ellenberg  
273 and Leuschner (2010) and secondly Hill et al. (1999) to fill gaps. We could not assign indicator values to all species  
274 in our dataset; 13 species (on a total of 211) were only identified up to genus level (those species had an average  
275 cover in plots of 5%). The community-mean abundance-weighted indicator value for acidity (mR, soil reaction)  
276 was multiplied with the abundance-weighted indicator value for soil fertility (mN, soil nitrogen). This product, mR  
277  $\times$  mN, proved a reliable estimate for humus quality (Rogister 1978) and has been used before as a proxy for general  
278 nutrient availability (Godefroid et al. 2005, Bernhardt-Römermann et al. 2015).

279 **Litter quality** of the present litter layer was determined as the abundance-weighted mean of species-specific tree  
280 and shrub litter quality scores (1–5; 1, low decomposability; 5, high decomposability; Table S2) weighted by the  
281 percentage canopy cover of each species in the tree and shrub layer of the plot *sensu* Verheyen et al. (2012), Maes  
282 et al. (2019), Depauw et al. (2020).

## 283 Statistical analyses

284 All analyses were performed in R v.4.1.1 (R Development Core Team version 4.1.1 2020) and data made available  
285 online (De Pauw 2022). First, we assessed the effect of gradients in built-up and forest land cover on the soil  
286 microclimate (hypothesis 1; H1). Second, we analysed which environmental variables were driving litter  
287 decomposition rates. Third, we studied cascading effects of the UHI effect on litter decomposition by using  
288 structural equation models (SEM) (hypothesis 2; H2).

### 289 1. Urban heat island effects on the soil microclimate

290 We used linear mixed-effects models (LMMs) to infer the effect of the UHI on the soil microclimate. The soil  
291 temperature was modelled as response variable with either the percentage of built-up area or forest cover  
292 surrounding the plot as the predictor variable of interest (i.e. fixed effects). We added the local percentage canopy  
293 cover as a covariate in the fixed effects (no interactions included) and the site ID (i.e. a factor variable with six

294 levels, one per urban forest) as a random intercept term (grouping variable). This model was repeated for the  
295 different scales at which land use was assessed to study the scale-dependency of land-use effects (equation 2).

$$\begin{aligned} 296 \quad & \text{soil temperature} \sim \text{land - use surrounding plot (\% built - up area or \% forest area)} \\ 297 \quad & + \text{local canopy cover} + (1|\text{city}) \qquad \qquad \qquad \text{equation 2} \end{aligned}$$

## 298 2. Environmental drivers of litter decomposition

299 To assess the environmental drivers of litter decomposition, we modelled the percentage mass loss of green tea  
300 (N = 152) and rooibos tea (N = 152) as a function of soil temperature, soil chemical characteristics (pH, C content,  
301 C:N ratio and humus quality) and litter quality. To reduce the dimensions of the soil chemical data, a principal  
302 component (PC) analysis was performed. The first two axes comprised 59.3% and 23.7% of the variation in soil  
303 characteristics (Figure S2). The first axis was strongly positively correlated with C:N ratio and C content and  
304 negatively with pH and humus quality (Figure S2, Table S3). The second axis was positively related with all four  
305 soil characteristics, but most strongly with C content and pH (Figure S2, Table S3). Given the high percentage of  
306 variation explained by the first axis, we used only the first axis as representative predictor for soil chemical  
307 characteristics in the statistical models ('first PC axis soil').

308 We ran LMMs with the plot ID nested in site ID as a random intercept term (i.e. a factor variable with 72 levels),  
309 to account for the nested sampling design (i.e. three teabags or pseudo-replicates per plot) (equation 3). To avoid  
310 multicollinearity issues among predictor variables, we calculated variance inflation factors (VIFs; *vif* function,  
311 package 'cars'). For all models, VIFs were smaller than 1.5 indicating no multicollinearity issues (Neter et al. 1990).  
312 LMMs were fitted with the function *lmer* from the 'lme4' package (Bates et al. 2015), conditional and marginal R<sup>2</sup>  
313 were determined with the 'MuMIn' package (Barton 2019). The marginal and conditional R<sup>2</sup> give the proportion  
314 of mass loss variance explained by the fixed and fixed plus random effects, respectively (Nakagawa and Schielzeth  
315 2013). All predictors were standardised to unit variance and mean zero.

$$316 \quad \text{Teabag mass loss} \sim \text{soil temperature} + \text{litter quality} + \text{1st PC axis soil} + (1|\text{city/plot}) \qquad \text{equation 3}$$

## 317 3. Urban heat island effects on litter decomposition

318 Finally, to assess cascading effects of the urban heat island on litter decomposition, we combined the models of  
319 steps (1) (*Urban heat island effects on the soil microclimate*) and (2) (*Environmental drivers of litter decomposition*) into a  
320 structural equation modelling (SEM) framework. As environmental variables were determined at plot level, we

321 calculated the mean of replicate tea bag losses per plot to determine mass loss at the plot level (N = 66 green tea,  
322 N = 68 rooibos tea). For both green and rooibos tea, this makes the ratio of samples over variables in the model  
323 larger than 5 (9.4 and 13.6 for green and rooibos tea, respectively) which indicates an acceptable sample size (Grace  
324 et al. 2015, Lefcheck 2016). Mass loss of teabags was modelled with predictors proven to be significant in step (2).  
325 Therefore, the hypothesized SEM network for rooibos tea decomposition had less terms than the one for green  
326 tea (Figure 5, hypotheses in Figure S3). Additionally, soil temperature was modelled as the response variable to the  
327 percentage of built-up area to test for cascading urban heat island effects. We chose to use the largest spatial scale  
328 (3,200 m) as the effect of built-up area on microclimate was found to be highest for the largest scale in step (1).  
329 The local canopy cover was used as covariate for soil temperature, as done in step (1), given the known buffering  
330 effect of the canopy for soil temperatures.

331 To consider edge effects expected for soil and litter characteristics (Remy et al. 2016), we added two more models  
332 to the SEM network which model the soil and litter variables as response terms in function of the shortest distance  
333 to a forest edge as single predictor. Also, for the soil microclimate we expected an edge effect (Meeussen et al.  
334 2021a). The edge distance was natural log-transformed before modelling, given the exponentially increasing  
335 distance to the urban forest edge in the study design and the exponential nature of edge effects. However, we  
336 found that the distance to the edge was correlated with the percentage cover of built-up area (Pearson correlation  
337 = -0.39,  $p < 0.001$ , Figure S4). When the edge distance was added as predictor to the soil temperature model,  
338 model coefficients for built-up area and litter quality changed significantly. Moreover, when built-up area and  
339 distance to the edge were tested as single predictor, they both proved significant predictors for the soil temperature;  
340 but became insignificant when tested together indicating a confounding effect (Graham 2003). Therefore, we ran  
341 separate models for both predictors (Figure 5).

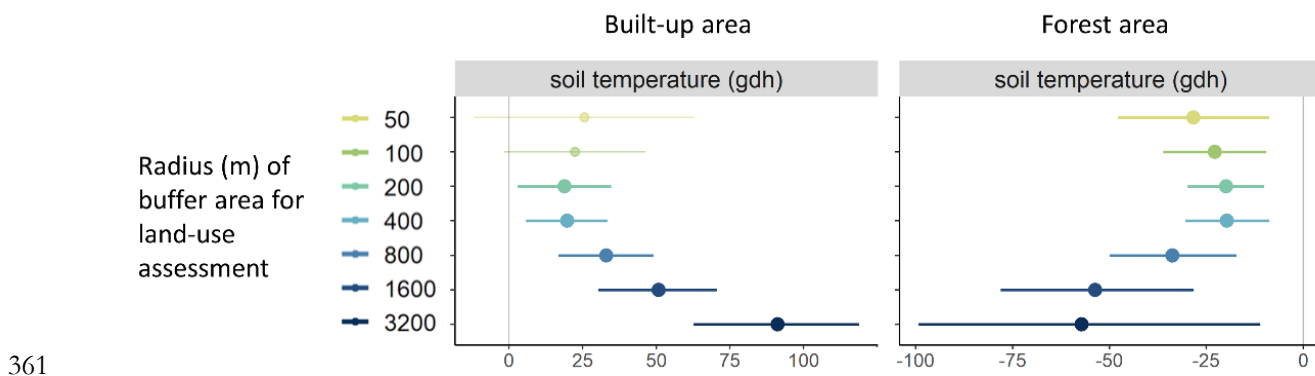
342 We used the 'piecewiseSEM' package to allow for more flexible path modelling including the modelling of random  
343 effects in separate models (Lefcheck 2016). Within this package, the LMMs were fitted with the function *lme* from  
344 the 'nlme' package (Pinheiro et al. 2021) given its compatibility with the 'piecewiseSEM' package. Shipley's test of  
345 directed separation was used to assess model fit. A  $p$ -value above 0.05 indicates relations consistent with the data,  
346 and that there are no missing relationships between unconnected variables (Shipley 2009).



## 347 Results

### 348 1. Urban heat island effects on the soil microclimate

349 We found that the proportion of land-use categories (urban vs. forest) surrounding the plots significantly affected  
350 the soil microclimate; soil temperature in the urban forests increased when built-up area around the plot increased,  
351 but only at larger spatial scales (radius  $\geq 200$  m) (Figure 2, Table S4 for all model estimates, confidence intervals  
352 and  $R^2_m$  and  $R^2_c$  values, Figure S5 for regression lines based on model predictions). Given the constant range of  
353 percentages built-up area for all scales (0–50%, Figure S7), the larger effect at larger scale was not caused by an  
354 increasing fraction of built-up area. When relating the soil temperature to the amount of forest area around the  
355 plots, we found that more forest cover lowered the soil temperature at small and large spatial scales. The local  
356 percentage canopy cover of the overstorey did decrease the soil temperature for all radii (Figure S6). Model  
357 estimates of land-use predictors and the local canopy cover are given in Table S4, accompanied with standard  
358 deviations and confidence intervals. Model fit was assessed with marginal ( $R^2_m$ ) and conditional  $R^2$  ( $R^2_c$ ), which give  
359 the proportion of mass loss variance explained by the fixed and fixed + random effects, respectively (Nakagawa and  
360 Schielzeth 2013). On average, the soil temperature models had an  $R^2_m$  of 0.05 and an  $R^2_c$  of 0.96.

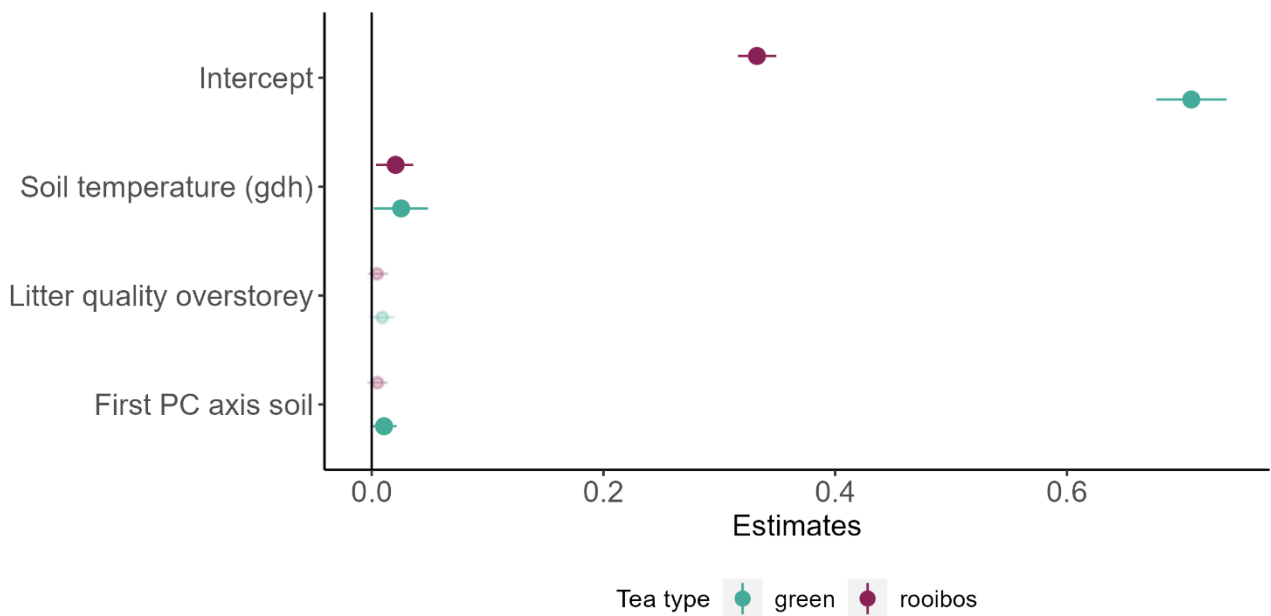


361

362 **Fig. 2 Changes in soil temperature for 1% increase in built-up or forest area.** Estimates and confidence intervals  
363 are shown for the effect sizes of land-use predictors for soil temperature based on linear mixed-effects models (LMMs).  
364 Each of the 14 (7 buffer radii  $\times$  2 land-use variables) LMMs accounts for canopy cover as a covariate in the fixed effects  
365 (see Figure S6 for canopy cover effect sizes) while site ID (six levels) was used as a random intercept term. The  
366 proportion (%) of built-up or forest area surrounding the plot was calculated in buffers with increasing radii (50 to 3,200  
367 m) from the ESA WorldCover map (Zanaga et al. 2021). Significant estimates have confidence intervals not overlapping  
368 0 and are shown larger and less transparent. Soil temperature is expressed in growing degree hours (gdh) during the litter  
369 decomposition experiment, the proportion land-use area as percentage 0–100%. Estimates thus represent changes in  
370 the response variable for 1% increase in built-up or forest area. See Figure S5 for regression lines based on model  
371 predictions.

372 2. Environmental drivers of litter decomposition

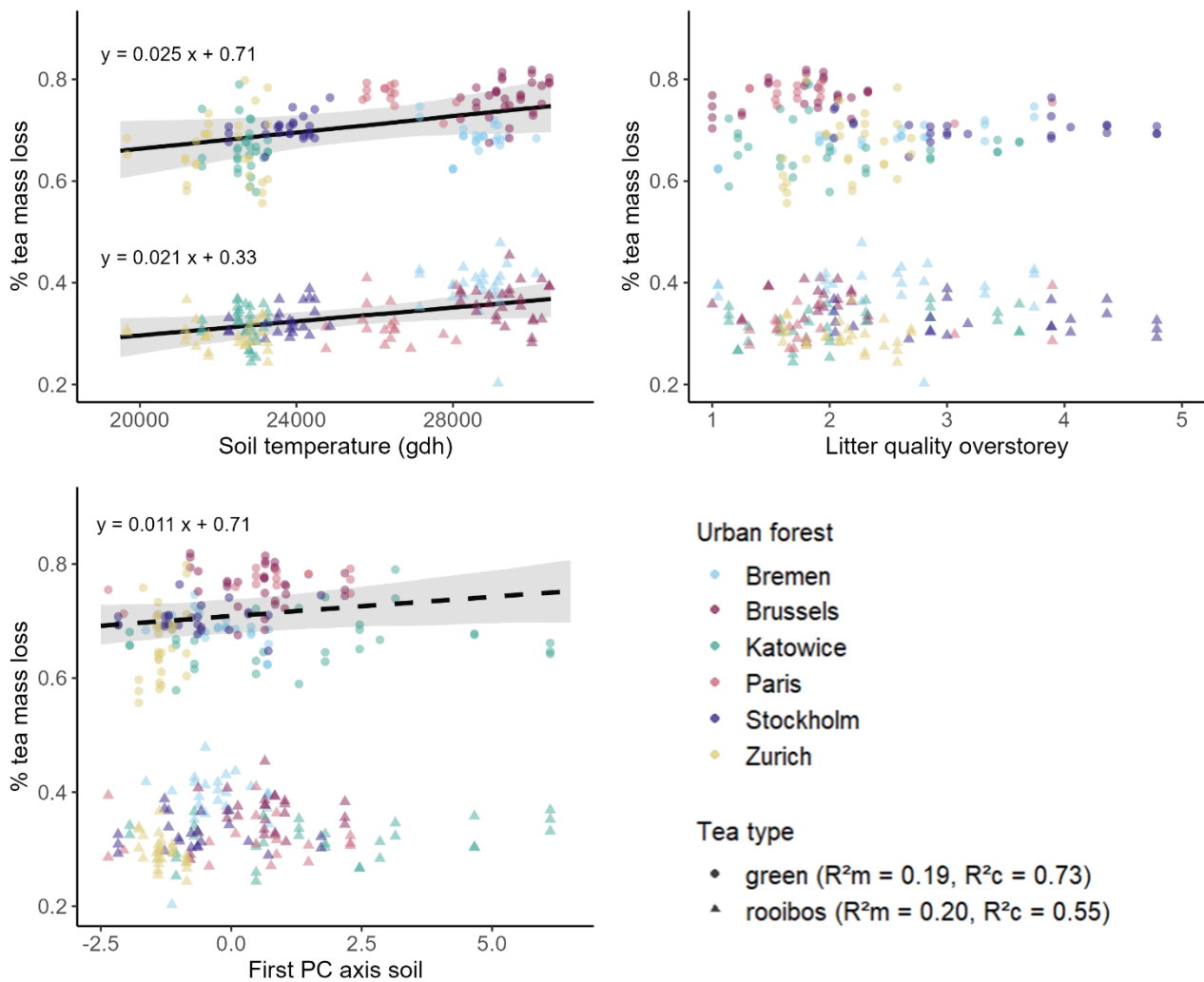
373 We found that the average green tea mass loss of 70.7% (SD 1.5%), was approximately twice as much as the 33.2%  
374 average rooibos tea mass loss (SD 0.8%) (Figure 3). These results confirm the difference in litter quality between  
375 the two tea types. In terms of soil microclimate, the mass loss of both green and rooibos tea was higher when soil  
376 temperature increased (Figure 3, Figure 4). The variation in green tea mass loss was additionally explained by the  
377 first principal component axis of the soil variables (strongly positively correlated with C:N ratio and C content,  
378 negatively with pH and humus quality, Figure S2, Table S3), which positively affected green tea decomposition,  
379 but had no significant effect on rooibos tea decomposition (Figure 4). Most urban forests showed a large variation  
380 in pH and humus quality along this first PC axis. Also differences among cities were present. For instance, soils  
381 from the urban forests in Brussels and Katowice had lower pH and humus quality and showed a smaller gradient  
382 compared to the other cities (Figure S2). Models for green and rooibos tea had a  $R^2_m$  of 0.19 and 0.20, and a  $R^2_c$  of  
383 0.73 and 0.55, respectively.



384

385 **Fig. 3 Estimates and confidence intervals of the effect sizes of environmental variables for the prediction of**  
386 **green and rooibos tea decomposition based on linear mixed-effects models (LMMs).** For both LMMs site ID  
387 (six levels) was used as a random intercept term. Significant estimates are shown larger and less transparent (\*\*\*)  $p <$   
388 0.001; \*  $p <$  0.05; (\*)  $p <$  0.1). Soil temperature is expressed in growing degree hours (gdh) during the litter  
389 decomposition experiment. See Figure 4 for regression lines based on model predictions.

390



391

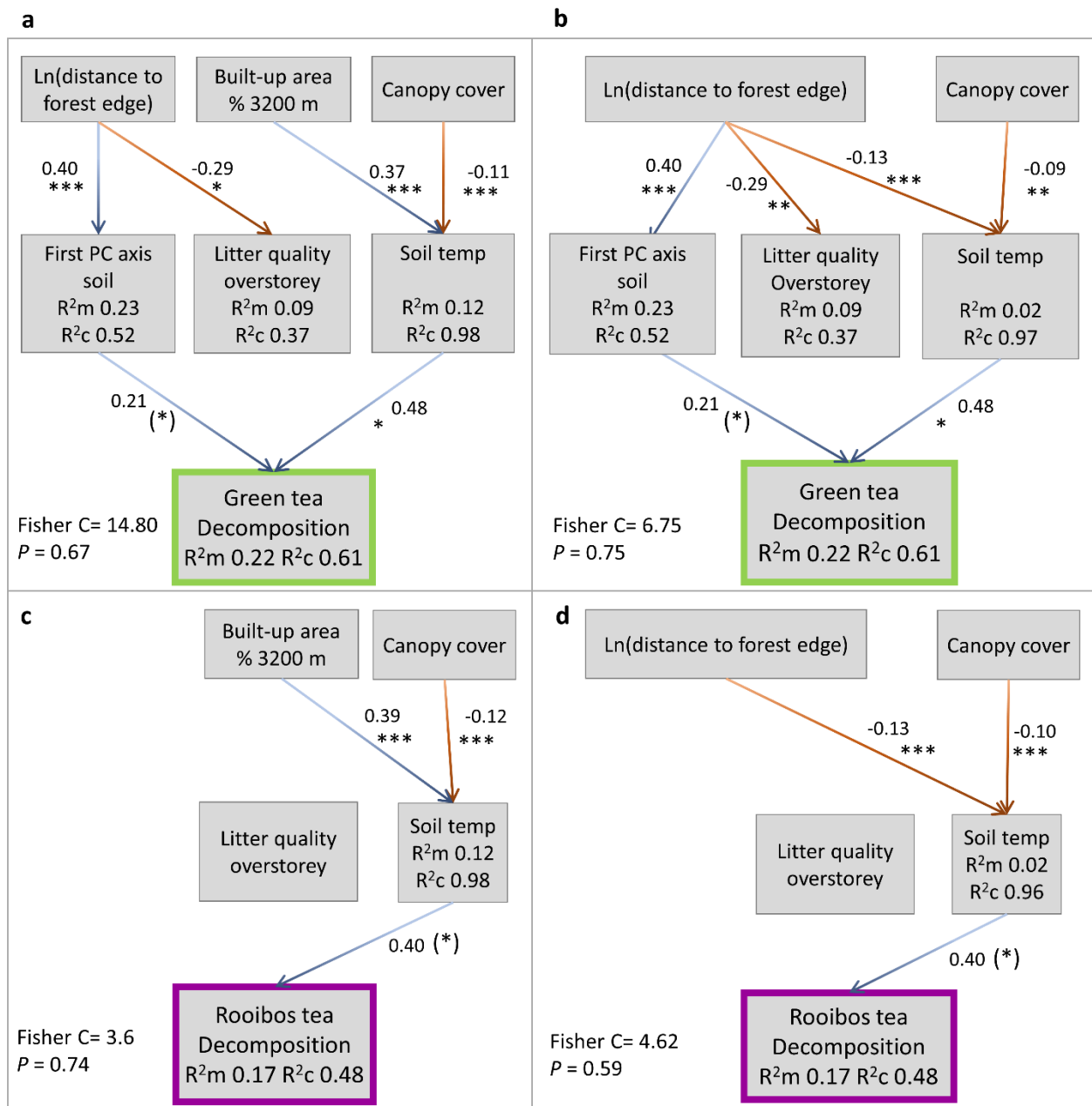
392 **Fig. 4 The relationship of tea mass loss (fraction) with environmental drivers for green and rooibos tea.** Tea  
 393 mass loss as a function of soil temperature (growing degree hours), litter quality of the overstorey (community weighted  
 394 mean value of species-specific litter quality scores) of tree and shrub species and the first PC axis of soil chemical  
 395 variables (strongly positively correlated with C:N ratio and C content, negatively with pH and soil nutrient regime, Figure  
 396 S2, Table S3). The lines show model predictions for significant parameter estimates (full line  $p < 0.05$ ; dashed line  $p <$   
 397  $0.10$ ) based on linear mixed-effects models (see methods for model specifications) and shading corresponds to 95%  
 398 confidence intervals. For each model prediction, other predictor variables were set to their mean values and not shown  
 399 in the equations for brevity.

### 400 3. Urban heat island effects on litter decomposition

401 We detected a cascading effect of the percentage of built-up area surrounding the plots on litter decomposition  
 402 through a positive effect on soil temperature in the SEMs (Figure 5a, 5c). The microclimate gradient also  
 403 encompassed an edge effect in our study design. In a second SEM candidate, we substituted the large-scale buffer  
 404 (3,200 m) of built-up area with the distance to the nearest forest edge. We found an edge effect, with higher soil  
 405 temperatures closer to the urban forest edge (Figure 5b, 5d). However, the effect size of built-up area, i.e. 0.37 for

406 green tea, and 0.39 for rooibos tea SEM model, and the marginal  $R^2$  value of the soil temperature models including  
407 built-up area as a predictor variable ( $R^2_m$  of 0.12 for both green and rooibos tea SEM models) were significantly  
408 larger compared to the model with distance to edge as a predictor variable (standardized effect size of distance to  
409 edge: -0.13 for green and rooibos tea SEM models,  $R^2_m$  0.02), indicating more variation in soil temperature  
410 explained when including built-up area instead of distance to the forest edge as predictor (Figure 5). We found that  
411 the local canopy cover negatively affected soil temperatures as well, and in turn litter decomposition rates. We  
412 hypothesized that the litter quality of the overstorey could affect soil temperatures through the thickness of the  
413 litter layer (with thicker litter layers leading to warmer soil temperatures in autumn), while this pathway was not  
414 confirmed (Figure 5, hypotheses shown in Figure S3).

415 For green tea, the hypothesized model contained more predictors for litter decomposition (soil characteristics and  
416 litter quality of the overstorey) as those proved significant for green tea, but not for rooibos tea in step 2 (Figure  
417 3). We found that more edge effects were affecting litter decomposition rates through these additional  
418 environmental drivers. An increase in distance to the forest edge positively influenced the first PC axis of soil  
419 chemical characteristics (positively correlated with C:N ratio and C content and negatively with pH and humus  
420 quality, Figure S2, Table S3), while it negatively influenced the overstorey litter quality scores. Contrary to step 2,  
421 in the SEM model we found that only the chemical soil characteristics significantly affected green tea  
422 decomposition, and not overstorey litter quality (Figure 3, Figure 5). All marginal and conditional  $R^2$  values of the  
423 models included in the SEM model are given in Figure 5.



424

425 **Fig. 5 SEM model results exploring cascading effects of built-up area and edge distance for green tea and**  
 426 **rooibos tea decomposition.** In the left panels (a, c), the results of the first SEM candidate model are shown, with built-  
 427 up area as predictor for soil temperature. In the right panels (b, d), the results of second SEM candidate model are  
 428 shown with distance to forest edge as soil temperature predictor. Standardized estimates are given and *p*-values (\*\*\*) *p* <  
 429 0.001; \*\* *p* < 0.01; \* *p* < 0.05; (\*) *p* < 0.1). Positive effects are given in blue, negative effects are given in red. Non-  
 430 significant relationships are not shown in the result graphs. Also, Fisher's C statistic and *p*-value are provided of the  
 431 global goodness-of-fit. A *p*-value above 0.05 indicates that there are no missing relationships between unconnected  
 432 variables detected in the model and the hypothesized relations are consistent with the data (Shipley 2009).

433

## 434 Discussion

### 435 1. Cities increase urban forest soil temperatures (H1)

436 We found that the percentage of built-up area in the surrounding landscape of urban forests in six large cities in  
437 Europe increased soil temperature due to the Urban Heat Island (UHI) effect. A greater percentage of forest cover  
438 in the surrounding landscape, on the other hand, lowered soil temperatures inside urban forests, most likely  
439 through a mass effect of shading and transpiration by trees. There was, however, an interesting difference in scale.  
440 Forest cover in the surroundings affected soil temperature from small to larger spatial scales (in buffer zones with  
441 50–3,200 m radius). The positive effect of built-up area on soil temperature became larger and more significant at  
442 larger spatial scales (200–3,200 m radius). This means that urban areas warm the soil microclimate at a larger spatial  
443 scale (i.e. more distal effects), whereas forest cover predominantly cools the soil locally (i.e. more proximal effects).  
444 Also Estoque et al. (2017) found that the correlation of impervious surface and green area with mean land surface  
445 temperature increased at larger spatial scale for impervious surface but decreased for green space. However, Ziter  
446 et al. (2019) tested buffer zones with 10–90 m radius, and found that both the warming and cooling effects of  
447 impervious surface and forest cover were largest for the 60 and 90 m buffer zones.

### 448 2. Drivers of litter decomposition in urban forests (H2)

449 Higher soil temperatures increased decomposition rates, a well-known pattern mainly attributed to increasing  
450 activity of the microbial community (Berg and McClaugherty 2008). We found that the UHI increases the urban  
451 forests' soil temperature, and as a consequence accelerates litter decomposition, affecting the rate of nutrient and  
452 carbon cycling. Additionally, increasing canopy cover reduced soil temperatures and slowed litter decomposition.  
453 Previous studies mostly compared rural with urban forests and some indeed found higher decomposition rates in  
454 urban forests (Pouyat and Carreiro 2003, Nikula et al. 2010, Tresch et al. 2019). However, this increase was not  
455 always solely attributed to UHI effects, but also to the presence of exotic invasive earthworms, higher N availability  
456 in the soil, changes in litter quality or soil fauna abundance (Pouyat and Carreiro 2003, Nikula et al. 2010, Tresch  
457 et al. 2019). In our study, we minimized variation in litter quality through the use of standardized litter, we excluded  
458 macrofauna given the small mesh sizes of teabags and took soil chemical variation into account. Still, rural-to-  
459 urban gradients differ in many aspects, and it is possible that some confounding factors covary with soil  
460 temperatures. For example, the microbial soil community composition can differ between urban and rural sites  
461 (Epp Schmidt et al. 2017, Hui et al. 2017), and might change along the urban forest edge-to-interior gradient in  
462 our study.

463 Furthermore, also soil moisture can have large effects on litter decomposition. In other studies, soil moisture has  
464 been shown to stimulate decomposition rates (Seidelmann et al. 2016, Wallace et al. 2018). However, an increase  
465 in microbial activity due to increases in soil moisture occurs especially when moisture levels are limiting or below  
466 a certain threshold (Berg and McLaugherty 2008, Prescott 2010). Our study was conducted during autumn, a rainy  
467 season in NW-Europe (Table S5), therefore, we could assume that soil moisture was probably not limiting  
468 microbial activity. Additionally, our European set-up made it difficult to monitor soil moisture continuously during  
469 the decomposition experiment. However, we acknowledge the importance of this driver and advocate for more  
470 studies on the role of soil moisture and the interaction with soil temperature. Especially in summer, UHI effects  
471 are often at their maximum (Arnfield 2003, van Hove et al. 2015) and litter decomposition can be at its highest  
472 rate (Sun and Zhao 2016, Vaidelys et al. 2020), yet summer drought has been shown to limit decomposition at  
473 urban forest edges (Garvey et al. 2022). The seasonal variation in litter decomposition and its drivers has received  
474 only limited attention in urban studies and is an important avenue for further research.

475 Gradients in soil chemical variables were (partly) driven by an edge effect. We noticed quite large gradients in  
476 humus quality within the urban forests, often coinciding with edge-to-interior gradients. The soils closer to the  
477 edge were characterised by a higher humus quality (Rogister 1978, Godefroid et al. 2005), which typically have  
478 higher bacteria-to-fungi ratios and relatively faster decomposition rates (Fierer et al. 2009, de Vries et al. 2012).  
479 Therefore, the edaphic environment at the forest edge was expected to facilitate litter decomposition, also due to  
480 a higher pH and lower C:N ratios. Surprisingly, we found the opposite. Green tea litter decomposition was slower  
481 in the soils with a higher pH, lower C content and C:N ratios, situated closer to the edge.

482 One potential mechanism causing the green tea decomposition to be slower in these soils, could be the high  
483 amounts of nitrogen typical of urban forest soils (Cusack, 2013; Mgelwa et al., 2020). Kahan et al. (2014) found,  
484 for example, greater nitrogen availability in soils of forest edges surrounded by higher housing densities. In addition  
485 to many early studies observing faster decomposition with higher N availability (e.g. Richards and Norman 1931),  
486 it has also been shown that nitrogen addition can suppress decomposition due to lower microbial biomass  
487 (Treseder, 2008) and shifts in metabolic capabilities (Ramirez et al., 2012). Another explanation might lay in the  
488 biotic aspect of decomposition. We have no data on the microbial community to confirm patterns in microbial  
489 biomass or bacteria-to-fungi ratios, but it is known that the composition of the microbial community can differ  
490 between urban and rural sites (Epp Schmidt et al., 2017; Hui et al., 2017), and might change along the urban forest  
491 edge-to-interior gradient. Additionally, a variation in the presence and abundance of herbs can affect litter

492 decomposition rates as well (Chomel et al. 2016, Wang et al. 2021a). Finally, also typical urban effects can affect  
493 decomposition rates, such as more disturbed soils with a high amount of stones near the urban forest edge, more  
494 soil compaction, land-use changes and time since former land use (Kahan et al. 2014), or the presence of high dog  
495 densities leading to nutrient fertilization (De Frenne et al. 2022).

### 496 3. Carbon and nutrient cycling in urban forests

497 We found that the percentage of build-up area surrounding urban forests accelerates litter decomposition through  
498 soil microclimate warming. This means that urban heat can accelerate carbon and nutrient cycling in urban forests,  
499 with potentially important consequences for ecosystem functioning. Faster litter decomposition increases  
500 respiration and lowers litter accumulation (Mack et al. 2004). Urban forests subject to heat islands could potentially  
501 store less belowground carbon than expected, when decomposition is not limited by drought, see e.g. Enloe et al.  
502 (2015), Garvey et al. (2022). Faster litter decomposition, however, also results in higher nutrient availability  
503 potentially increasing plant growth (Standish et al. 2004, Liao et al. 2008, Sokolov et al. 2008) as well as biotic  
504 homogenization in the forest understorey (Staude et al. 2020). The presence of exotic and invasive species, often  
505 higher in urban forests (Heckmann et al. 2008, Blood et al. 2016), could further increase the risk of biotic  
506 homogenization. It seems thus that urban heat can accelerate forest ecosystem dynamics, with several potential  
507 consequences for nutrient and carbon cycling, and vegetation dynamics.

508 Extreme summer droughts in urban forests can result in opposite patterns, hampering litter decomposition  
509 (Santonja et al. 2015, Xie et al. 2020) and decreasing tree growth due to heat and drought stress (Vaz Monteiro et  
510 al. 2017, Moser-Reischl et al. 2019). Seasonal changes in both UHI strength (Arnfield 2003, van Hove et al. 2015)  
511 and litter decomposition rates and drivers (Vaidelys et al. 2020) might result in a dynamic effect of urban  
512 environments on litter decomposition in urban forests. As a consequence, making up the final resulting balance  
513 leading to carbon losses or gains stays extremely challenging and requires more information on long-term patterns.  
514 For example, many studies on litter decomposition are limited to the early stages (the first three months) of  
515 decomposition, whereas this is only the very start of the transforming process from litter to humus (Prescott 2005).  
516 Furthermore, given the fact that litter decomposition is only one phase of a complete cycle of nutrient and carbon  
517 flows, feedback mechanisms between soil processes, vegetation and climate are to be expected, as well as  
518 interactions between the carbon and nitrogen cycle, but up till now these are only limitedly studied (Melillo et al.  
519 2002, Mack et al. 2004, Sokolov et al. 2008, Hobbie 2015, Melillo et al. 2017). This information is also more and



520 more valuable for global atmospheric carbon models given the ever-increasing rate of global urbanization and  
521 share of natural areas that become embedded in urbanised landscapes (Seto et al. 2011, Li et al. 2021).

## 522 Conclusion and outlook

523 Our results underpin the important role of soil temperature for litter decomposition in urban forests during autumn  
524 and confirm the hypothesized acceleration due to urban heat islands. As a consequence, urban forest carbon and  
525 nutrient cycles might accelerate which could significantly affect carbon sequestration in urban forests. Additionally,  
526 soil microclimate was strongly affected by canopy cover, supporting the important local impact of tree cover and  
527 stand characteristics on microclimates in urban forests. Litter decomposition rates were not only affected by soil  
528 temperature, but also by other factors which control microbial and mesofauna activity, such as soil chemical  
529 properties. Most studies on litter decomposition have performed trials in a short-term setup. To improve our  
530 understanding on how urban heat affects ecosystems' nutrient and carbon cycling, we suggest to design studies  
531 spanning longer time periods and multiple seasons, and measure or manipulate soil moisture and decomposition  
532 in a more continuous way through experiments. Furthermore, the role of biota in these processes should not be  
533 overlooked, it would be highly valuable to further explore urban warming effects on macro-, mesofauna and  
534 microbial communities. With more than half of the world's population currently living in cities (UN 2019), urban  
535 forests provide invaluable ecosystem services such as buffering against urban heat and climate change. However,  
536 we show that these urban forests are themselves not unaffected by warming, which impacts ecosystem processes  
537 crucial for their functioning.

## 538 Statements and declarations

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### 545 Competing interests

546 The authors have no relevant financial or non-financial interests to disclose.

547

548 Author contributions

549 KDP, PDF, PV, LD and KV conceived the ideas and designed methodology; all authors collected data; KDP led  
550 the data analyses and the writing of the manuscript in close collaboration with PDF, LD, PV and KV. All authors  
551 contributed critically to the drafts and gave final approval for publication.

552 Data availability statement

553 Data will be made available online (through Figshare) and is now already accessible through a private link:

554 <https://figshare.com/s/192477e16d62fa45f2a0>.

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