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Leaf litter quality is a stronger driver of decomposition in lawns than the microclimatic effect of trees

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Lawns and wooded areas dominate urban green spaces (UGSs) and are known to store substantial amounts of carbon in soils, yet the role of lawn trees in litter decomposition dynamics remains poorly understood. This study assessed how trees influence litter decomposition through (i) microclimatic (temperature and moisture) and (ii) soil properties effects. Decomposition rates of 8 tree and shrub litters (*Quercus robur, Quercus rubra, Fagus sylvatica, Platanus* × *acerifolia, Pinus radiata, Tilia* × *europaea, Cornus alba, Spiraea vanhouttei*) frequently encountered in urban areas were measured using litterbags(5 mm and 2 mm mesh) in adjacent open and wooded lawns, allowing assessment of macrofauna contribution. Microclimatic conditions and soil physicochemical properties were monitored. Decomposition rates varied widely among litter types, with 5 tree litters decomposing slowly ($0.29 < k < 0.85\ y^{-1}$) and 3 litters—mostly shrubs—decomposing faster ($3.06 < k < 6.23\ y^{-1}$). Despite a higher average temperature in open lawns (+1.6 °C), the decomposition rates were similar between lawn types except for the two *Quercus rubra* and *Q. robur*, which decomposed 87% and 54% faster in wooded lawns, respectively.Macrofauna exclusion revealed a stronger contribution to the decomposition of labile litter in both lawn types. Overall, decomposition rates were primarily driven by litter quality, especially lignin and soluble compound content (R^2 = 0.59–0.92), while microclimate and soil properties had limited influence. These results indicate that litter quality outweighs microclimatic effects of trees on decomposition in UGSs and suggest that maintaining litter in wooded lawns could support soil carbon processes without impairing decomposition. Together, these findings offer UGS managers an initial framework to assess litter decomposition from various litter types when recycled on open and wooded lawn soils.

KEYWORDS
lignocellulose index, litter mass loss, litter recycling, macrofauna, shading, urban soil

1 Introduction

Vegetation and soil in urban green spaces (UGSs) are the most important providers of urban ecosystem services, such as C storage and climate regulation. Therefore, they significantly contribute to urban ecosystem sustainability (1, 2). In these systems, litter decomposition is a central process governing soil carbon dynamics. Lawns cover 50 to 70%

of UGS areas and are the most representative urban ecosystem under management (3, 4), that can mitigate GHG emissions from cities by storing C in soil and vegetation (2, 5). However, their functioning is influenced by vegetation structure, particularly the presence of trees forming "wooded lawns", which modify both organic matter inputs and microclimatic conditions (6, 7). Although open lawns are not entirely free of tree shading, they are exposed to more sunlight overall, with inputs of labile organic matter primarily derived from mowed grass and root exudate turnover (8, 9). Wooded lawns are formed of usually deciduous trees with varying spatial arrangements, ranging from a single tree to groups of treesunder which freshly fallen leaf litter is regularly raked to maintain the lawn underneath. Thus, leaf litter does not decompose under the trees, or it is concentrated under a few trees for commodity (10, 11), and lawn biomass production is strongly limited by exposure to sunlight (12).

In natural ecosystems, litter decomposition is primarily driven by two key factors: litter quality and environmental conditions (13, 14). First, litter biochemical quality plays a particularly important role in the initial stages of decomposition (15), with recalcitrant litter (e.g., lignin-rich *Fagus sylvatica* and *Pinus sylvestris* litter) generally decomposes slower than labile litter [e.g., N-, soluble-compound-litter from *Lolium perenne* or *Tilia vulgaris* (16);]. As in natural ecosystems, litter biochemical quality influences decomposition in different UGS types. The labile litter of cool-season lawns (i.e., mowing residues) decomposes rapidly (≈ 3 weeks) due to its low C:N *ratio* (≈ 10) and lignin content (17) that make it easily accessible to microbes and macrofauna (18–20). In comparison, litter from trees and shrubs generally decomposes more slowly (from months to years) because of their more recalcitrant biochemical composition (21), but significant differences between species persist (22).

Second, trees can indirectly influence decomposition by modifying local microclimatic conditions such as temperature, radiation, and soil moisture through shading and rainfall interception (23), and in turn microbial activity (24). In natural environments, temperature can have a positive effect on litter decomposition when moisture conditions are non-limiting (25, 26). Rainfall interception by trees in temperate ecosystems can limit the decomposition process by limiting the moisture content of litter (27, 28). In addition, the various litter inputs from trees can locally modify the soil properties (29–31) and have other indirect effects on decomposition. For example, pH and organic matter inputs in various UGSs (i.e., lawns, shrubs, trees) strongly drive soil fauna and microbial enzymatic activity (32–34).

In urban lawns, litter quality and environmental conditions may interact in complex ways. While litter inputs from trees are often removed through regular raking (i.e., every 15 days), limiting their influence on soil properties (6, 10, 11, 35), the microclimatic effects of tree canopies may remain a dominant control on decomposition. However, the relative importance of litter quality versus tree-induced microclimate in regulating decomposition in urban lawns remains poorly understood.

The decomposition process has been studied in various terrestrial ecosystems, in various litter types and involving various soil fauna [especially macro and mesofauna (36–38);], but the influence of trees on this process has only been characterized in dry environments (39) and forests (40), and very little so in urban environments (24, 30). Studies on decomposition processes in urban areas have focused on the impact of urbanization (heat islands, atmospheric deposition) on the decomposition of tree litter in forests [e.g (41)] and on the impact of cultivation practices on soil fauna diversity in allotment gardens [e.g (42)]. Moreover, studies in temperate urban lawns remain scarce and limited to boreal, tropical or oceanic climates (17, 30, 43). As a result, it remains unclear whether decomposition in these systems is primarily controlled by litter characteristics or by microclimatic conditions created by trees. In addition, UGS managers are increasingly questioning the benefits of tree litter raking, which has a significant cost in terms of labor time and fossil resources, and contributes negatively to the C footprint of UGSs (44). It is likely that maintaining litterfall under trees could enhance ecosystem servicesthrough soil C sequestration and soil quality (11, 29, 43). However, the ability of urban lawns to decompose different tree and shrub litter types remains poorly understood (45). Providing UGS managers with an initial framework for assessing litter decomposition when recycled on lawn soils is therefore essential.

The aim of this study was to assess the relative influence of litter quality, as well as tree-induced microclimatic and soil property effects on litter decomposition in urban lawns. Specifically, we compared decomposition rates of multiple tree and shrub litter types, frequently encountered in temperate-climate UGSs, in adjacent open and wooded lawns under similar soil types (Table 1). We hypothesized that i) tree canopies would reduce decomposition rates by lowering temperature through shading, ii) tree canopies would slow down decomposition by reducing precipitation beneath the trees, iii) litter quality would strongly control decomposition rates, with recalcitrant litters (e.g., *Platanus*, *Fagus*, *Pinus* litters) decomposing more slowly than labile ones (e.g., *Tilia*, *Cornus* litters), and iv) macrofauna (i.e., earthworms) would contribute more to the decomposition of labile litter.

TABLE 1 Soil properties according to lawn types.

Lawn type	Total organic C (%)	Total N (%)	C:N	Total P (%)	pH	Bulk density (g cm^{-3})	Microbial biomass (mg C kg^{-1})
Wooded lawn	3.16 (0.88)	0.25 (0.06)	**12.8 (0.59)**	0.04 (0.01)	**5.57 (0.14)**	1.04 (0.04)	**680 (23.4)**
Open lawn	2.00 (0.12)	0.19 (0.01)	**10.6 (0.24)**	0.04 (0.01)	**5.80 (0.03)**	0.99 (0.04)	**502 (56.7)**

Organic C, Total N (ISO 10694); Total P (ISO 11263); Microbial biomass (ISO 14240-2); pH (ISO 10390); Bulk density (ISO 11272.)
Values are means (n=3) and standard deviations are in parentheses. Bold text indicates significant differences between lawn types (Student t-test, P<0.05).

2 Materials and methods

2.1 Study area and sites

Our study was conducted in a 5-ha urban green space (UGS) located in the city of Angers, France (47°28′59″N, 0°30′5″W), which provided a uniform pedoclimatic context across the study site. Climate in the region is warm and temperate (Cfb type, warm temperate - fully humid - warm summer) according to the Köppen-Geiger classification (46). Angers had an average annual temperature of 12.6 °C over the 1991–2020 period, and annual rainfall averaged 709 mm and was evenly distributed throughout the year.

We selected three pairs of open and wooded lawn areas, within the UGS, all with comparable orientation relative to the sun's trajectory. All lawn areas had been established for at least 18 years and were mainly composed of *Lolium perenne*, along with some spontaneous species such as *Trifolium repens*, *Agrostis capillaris*, *Potentilla reptans*, and *Bellis perennis*. The wooded lawns were shaded by 45-year-old red oaks (*Quercus rubra*), whose canopy diameter was approximately 15 m. These species describe the background vegetation of the study sites and were not used as experimental litter. In each lawn pair, we established one experimental block for litter decomposition in both the open and wooded areas, resulting in three replicates per lawn type. In wooded lawns, blocks were placed directly under the tree canopy, at least 2 meters from the trunk. In open lawns, blocks were positioned at least 5 meters away from the nearest tree canopy. Microclimatic, biological, and soil bio-physicochemical data were collected within each experimental block (Table 1).

The wooded and open lawns were mowed twice a month (cutting height: 10 cm) from March to October 2022 with grass clipping retention, without irrigation and fertilization. Litterfall was removed from the wooded lawns twice a month using a blower, from October 2021 to January 2022.

2.2 Soil properties

According to the world reference base (47), the soils were Brunisols with a sandy loamy texture. We analyzed the bio-physicochemical properties of each block (total organic C, total N, total P, microbial biomass, pH, bulk density, soil texture, CEC). The soils were sampled between 0 and 10 cm depth with an auger (A horizon) in February 2022. Soil cores were also collected between 0 and 10 cm depth using a steel cylinder (5.5 cm diameter; 6 cm height), oven-dried at 105 °C, and weighed for bulk density calculations. Bulk density (g cm^{-3}) was determined from the oven-dried mass of the soil cores and core volumes. Potential historical soil pollution was checked on a composite sample from the 6 blocks for metallic trace elements (Pb, Hg, As, Cu, Zn, Mn) and polycyclic aromatic hydrocarbons (fluoranthene, benzo(b) fluoranthene, benzo(a)pyrene). These analyses were conducted to ensure that soil contamination did not confound litter decomposition processes, as such pollutants can affect microbial and faunal activity (48). All measured concentrations were below the nation-wide fixed threshold values (Supplementary Table 1).

2.3 Lawn biomass and earthworm abundance

Grass biomass production (i.e., clipping retention) of the open and wooded lawns was measured twice a month during the plant growth period (June to December 2022). Fresh biomass was collected in each experimental block using a mower (cutting height 10 cm), over a 0.5 m wide and 12 m long area. Since these samples were composite and not replicated, no statistical analysis was carried out on the data.

Earthworm sampling was conducted to characterize potential differences in soil biological properties between open and wooded lawns. Earthworms were sampled in May 2022 using the spade test (49), by excavating 20 x 20 x 20 cm soil volumes. Three soil samples were taken *per* experimental block, i.e., a total area of 0.36 m^2 *per* lawn type. To avoid digging difficulties in dry soil, each location received a constant amount of water once a day for three days the week before sampling. The water amount, the irrigation time and the distance from the litterbags were adjusted to avoid unintended effects on soil moisture and decomposition. The samples were kept at 4 °C before analysis. Earthworms were extracted from each sample and separated according to their ecological category (50) – epigeic, anecic, or endogeic. Earthworm abundance was measured by counting the earthworms from each category *per* experimental block.

2.4 Litter collection and mass loss calculation

The decomposition rates of 8 litter types were measured using the litterbag method, with mesh sizes of 5 and 2 mm. The 2-mm-mesh size was used to exclude the relative contribution of macrofauna to decomposition. The 8 litter types were collected in Angers from common shrubs (*Cornus alba, Spiraea vanhouttei*) and trees (*Quercus robur, Quercus rubra, Fagus sylvatica, Platanus × acerifolia, Pinus radiata, Tilia × europaea*) of temperate UGSs, and dried at room temperature.

The litterbags were manufactured in the laboratory using white polyethylene grids of 16 x 16 cm side. Five g of air-dried litter were carefully inserted into the litterbags, which were closed using staples. A total of 384 litterbags (8 species x 2 mesh sizes x 2 lawn types x 3 replicates x 4 harvests) were installed in the field on October 1st, 2021 (beginning of the litterfall period, day 0) and monitored for 1 year until October 5th, 2022 (day 369). They were harvested 4 times to assess mass losses, on days 70, 152, 208 and 369. They were spaced 40 cm apart to avoid interaction effects within the same experimental block, and their position was randomized to mitigate the effect of microclimate variability and litter species proximity. At each harvest, litter was cleaned with a paintbrush and water if necessary. Then, it was dried in a ventilated oven at 60 °C for 48 hours. Mass losses were calculated after dry mass weighing, using the initial dry mass of litter.

2.5 Initial chemical and morphological characterization of litter

The initial litter properties are reported in Table 2. The C and N contents were determined by elemental analysis (Euro-EA,

Eurovector, Milan, Italy). The soluble compound, hemicellulose, cellulose and lignin contents were determined with the Van Soest fractionation procedure (51) using an Ankom 2000 Automated Fiber Analyzer (ANKOM Technology, Macedon, USA). The polysaccharide content was calculated as the sum of the cellulose and hemicellulose contents. The lignocellulose index (LCI) was calculated as follows (52) (Equation 1):

$$LCI = Lignin/(Lignin + Cellulose + Hemicellulose) \quad (1)$$

The specific leaf area or SLA ($m^2\ kg^{-1}$) of the litter types was calculated as follows (39) (Equation 2):

$$SLA = A/M \quad (2)$$

where A is the leaf area (m^2) and M is the dry mass (kg). Standard errors are not available because the SLA calculation method yielded a single SLA value for each selected leaf.

2.6 Microclimate and soil moisture measurements

Microclimate parameters (litter temperature, precipitation, global radiation) and soil physical parameters (soil temperature and volumetric moisture) were monitored every 5 minutes in the 3 experimental blocks (3 replicates) in the wooded and open lawns. Litter temperature was measured using sensors in the litterbags (Ibutton, Whitewater, USA), beneath the leaf litter. Global radiation was measured using pyranometers (RG100, Solems, France). Precipitation was measured with manual rain gauges after each rainy event. Soil surface temperature and volumetric moisture at 7.5 cm depth were monitored with TDR probes (CS 655, Loughborough, UK). As the open lawns received some tree shading at the end of the day, we quantified this effect by comparing the daily global radiation of the two lawn types with public weather data from Beaucouzé weather station (53), located in an unshaded environment 10 km from the study site.

2.7 Data processing and statistical analyses

The decomposition rates were quantified using the decomposition coefficient k calculated from a single-pool exponential model (54, 55) (Equation 3):

$$M_t = M_0 \times \exp(-kt) \quad (3)$$

where M_t is the mass of dry litter (g) at a given time t, M_0 the initial mass of dry litter (g), k the decomposition coefficient (y^{-1}) and t the incubation time (years).

Three-way repeated measures analysis of variance (rm-ANOVA, Supplementary Table 2) was used to analyze the 'lawn type', 'litter type' and 'litterbag mesh size' effects (between effects) over time (within effects) on the remaining litter mass. Student t-tests were used to determine differences in litter decay rates in the 5- and 2-mm-mesh litterbags *per* lawn type. A three-way ANOVA was performed to analyze the 'lawn type', 'litter type' and 'litterbag mesh size' effects on the litter decomposition coefficients (k). Then, Student t-tests were used to determine the differences between the decomposition coefficients according to the lawn type and the mesh

TABLE 2 Initial C and N contents (%), C:N *ratio*, biochemical fractions (%), soluble compound: N, polysaccharide: N, lignin: N *ratios*, lignocellulose index (LCI) and surface leaf area (SLA, $m^2\ kg^{-1}$) of 8 litters.

Litter	Reference	C (%)	N (%)	C:N	Sol. (%)	Hemi. (%)	Cell. (%)	Lignin (%)	Ash (%)	Poly. (%)	Sol.:N	Poly.:N	Lignin:N	LCI	SLA ($m^2\ kg^{-1}$)
Tree litter types															
Platanus × acerifolia	PLA	48.4 (0.09)	0.50 (0.00)	96.1 (0.39)	29.8 (0.52)	14.0 (0.41)	21.1 (0.20)	34.6 (0.24)	0.59 (0.07)	35.0 (0.27)	59.2 (1.31)	69.7 (0.18)	68.7 (0.35)	0.50 (0.00)	12.1
Fagus sylvatica	FAG	47.6 (0.26)	0.73 (000)	65.0 (0.58)	30.8 (2.25)	16.6 (1.37)	19.9 (0.38)	31.6 (0.65)	1.10 (0.18)	36.5 (1.43)	42.0 (3.02)	49.8 (2.04)	43.1 (0.96)	0.46 (0.00)	14.6
Pinus radiata	PIN	52.0 (0.19)	0.50 (0.02)	103.1 (4.60)	21.9 (0.03)	19.2 (0.21)	20.6 (0.26)	38.0 (0.26)	0.22 (0.02)	39.9 (0.23)	43.4 (2.09)	79.1 (3.54)	75.5 (3.95)	0.49 (0.00)	3.21
Quercus robur	QUR	49.2 (0.33)	1.09 (0.02)	45.0 (1.05)	31.5 (0.57)	19.4 (0.44)	18.8 (0.16)	29.6 (0.45)	0.83 (0.08)	38.1 (0.41)	28.8 (0.26)	34.8 (0.74)	27.1 (0.9)	0.44 (0.00)	11.6
Quercus rubra	QRA	51.2 (0.36)	0.75 (0.01)	68.7 (1.59)	29.1 (0.62)	20.6 (0.57)	19.3 (0.30)	30.6 (0.48)	0.44 (0.15)	39.9 (0.83)	39.0 (0.28)	53.5 (1.94)	41.0 (0.88)	0.43 (0.01)	15.4
Tilia × europaea	TIL	43.0 (0.36)	0.90 (0.01)	47.9 (0.48)	55.4 (1.04)	17.2 (0.89)	15.0 (0.16)	11.8 (0.22)	0.57 (0.18)	32.2 (0.90)	61.8 (1.85)	35.9 (0.64)	13.2 (0.18)	0.27 (0.00)	16.5
Shrub litter types															
Spiraea vanhouttei	SPI	48.5 (0.31)	0.89 (0.01)	54.7 (0.46)	41.3 (1.76)	28.0 (0.81)	14.2 (0.53)	16.1 (0.87)	0.30 (0.03)	42.3 (1.20)	46.5 (1.73)	47.6 (1.54)	18.2 (1.10)	0.28 (0.01)	17.0
Cornus alba	COR	43.6 (0.34)	1.29 (0.01)	33.9 (0.45)	63.8 (1.22)	17.4 (1.32)	11.3 (0.21)	7.00 (0.52)	0.42 (0.09)	28.7 (1.46)	49.6 (0.77)	22.3 (1.30)	5.5 (0.36)	0.20 (0.02)	12.7

Values are means (n=3) and standard deviations are in parentheses. Sol., soluble compounds; Hemi., hemicellulose; Cell., cellulose; Poly., polysaccharides.

size. We also used Student t-tests to determine the effects of the lawn type on the daily averaged climatic parameters (precipitation, global radiation, litter and soil temperature and moisture) and the soil properties (e.g., C:N *ratio*, pH, earthworm abundance). Data were log_{10}-transformed when necessary to meet the assumption of normality and homogeneity of variances. We performed Pearson correlations to test the links between the decomposition coefficients and the microclimate parameters. Linear regression models were used to analyze the links between the litter traits and the decomposition coefficients. Exponential regressions were chosen when they provided a better fit to the data, as indicated by higher R^2 values and significant P-values. All statistical analyses were conducted using R software, version 4.2.2 (56).

3 Results

3.1 Environmental context: effect of trees on soil properties, biomass production and microclimate

The C:N *ratio*, pH and microbial biomass C were significantly higher in the wooded lawns than in the open lawns (Table 1). Total organic C, the total N and total P contents and bulk density were not significantly different in the two lawns. Between May and December, the cumulative grass clipping retention was almost 9 times greater in the open lawn (114 g dry matter m^{-2}) than in the wooded lawn (13 g dry matter m^{-2}). The number of earthworms was 233 ± 65 individuals m^{-2} in the open lawn and 133 ± 121 individuals m^{-2} in the wooded lawn (Student t-test, P>0.05; Supplementary Figure 1). No significant difference in the proportion of each earthworm category was found between the two lawn types.

Global radiation, relative air humidity, precipitation, litter and soil temperature and moisture significantly differed in the wooded and open lawns from October 2021 to December 2022 (Figure 1). Average global radiation and precipitation were significantly lower in the wooded lawns (67 ± 17 W m^{-2} and 328 ± 11 mm, respectively) than in the open lawns (139 ± 10 W m^{-2} and 411 ± 11 mm, respectively). Average global radiation was 20% lower in the open lawns and 62% lower in the wooded lawns compared to the unshaded area of the weather station (174 W m^{-2}; Supplementary Figure 2). The open and wooded lawns were not significantly different as regards litter temperature (14.4 ± 0.2 °C and 13.1 ± 0.9 °C, respectively), soil temperature (14.4 ± 0.0 °C and 13.8 ± 0.6 °C, respectively) and moisture (13.3 ± 1.0% and 14.2 ± 0.7%, respectively). Nevertheless, daily litter and soil temperatures were significantly lower in the wooded lawns than in open lawns from May to October

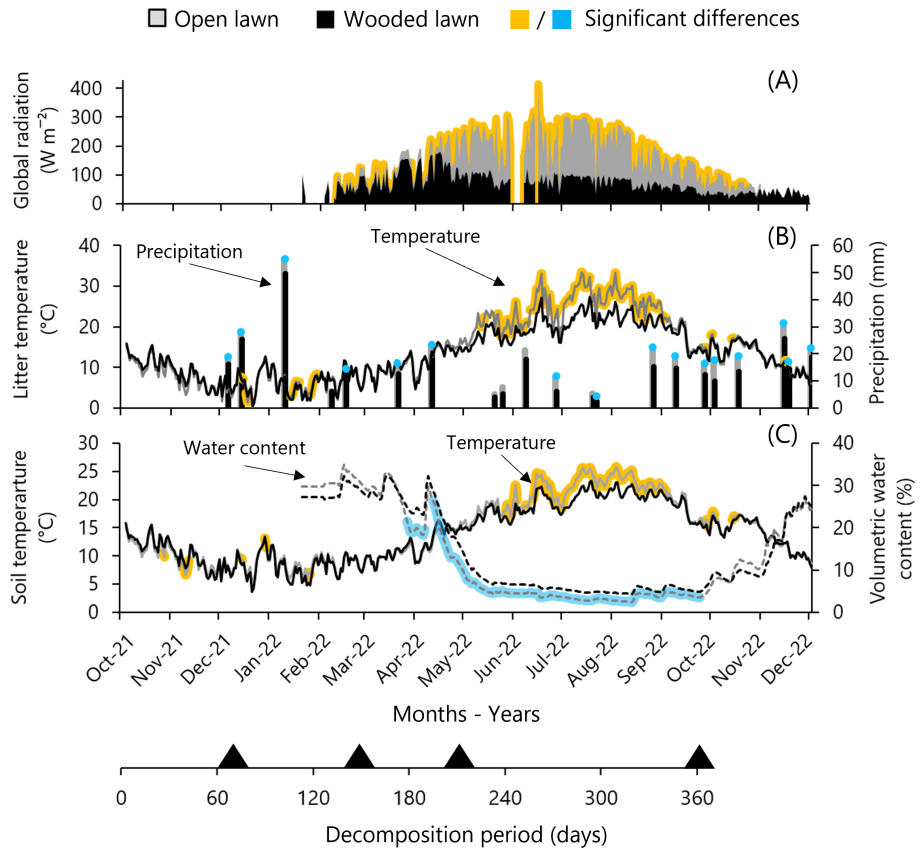


FIGURE 1
Microclimate data: **(A)** Mean daily global radiation (W m^{-2}), **(B)** litter temperature (°C) and precipitation (mm), **(C)** soil temperature (°C) and volumetric water content (%) in open (gray lines) and wooded (black lines) lawns from October 2021 to December 2022 (n=3, t-tests for lawn types). Yellow- and blue-colored areas, significant differences between the two lawn types for a given parameter and date. The litter decomposition period (days) is shown on the bottom horizontal scale. Black triangles correspond to the litterbag sampling points.

10.3389/fsoil.2026.1777598

(Figures 1B, C). These differences mostly occurred during daytime, when litter temperature at the open lawn soil surfaces sometimes exceeded temperature in the wooded lawns by 20 to 30 °C in summer. Moisture was also significantly lower in the open lawns from May to October (Figure 1C).

3.2 Litter decomposition patterns across lawn types, litter types, and mesh sizes

The remaining litter mass was significantly different depending on the litter type and litterbag mesh size over time ('litter type' x 'mesh size' x 'time' interaction, P<0.001; Supplementary Table 2). The litter types decomposed at significantly different rates depending on the lawn type, the litter type and the litterbag mesh size ('lawn type' x 'litter type' x 'mesh size' interaction, P<0.05; Supplementary Table 3). Regardless of the mesh size and lawn type, 5 litters decomposed slowly ($0.29 < k < 0.85$ y^{-1} for PLA, FAG, PIN, QUR and QRA litters), while 3 litters decomposed faster ($3.06 < k < 6.23$ y^{-1} for TIL, SPI and COR litters; Figures 2A, B). In the 5-mm mesh litterbags, decomposition was significantly faster in the wooded lawns than in the open lawns for QUR (k = 1.00 and 0.65 y^{-1}, respectively) and QRA (k = 1.51 and 0.82 y^{-1}, respectively; Figure 2A) litter. In the 2-mm mesh litterbags, decomposition was significantly slower in the wooded lawns than in the open lawns for FAG (k = 0.27 and 0.33 y^{-1}, respectively; Figure 2B) litter.

Decomposition was significantly slower in the 2-mm-mesh litterbags than in the 5-mm-mesh ones (Figure 3). This difference was particularly marked in both lawn types for TIL, COR, SPI, QUR and QRA (Figure 3). Decomposition of PLA, FAG and PIN litter was not affected by the lawn type (Figures 2A, B) or the litterbag mesh size (Figure 3).

3.3 Drivers of decomposition: microclimate and litter properties influences

No consistent significant correlation between the litter decomposition coefficients and the cumulative microclimate parameters was found (Figure 4). COR litter decomposition in the 5-mm-mesh litterbags was significantly and positively correlated with cumulative global radiation, litter temperature and soil temperature (Supplementary Figures 3A-C), while QUR litter decomposition was significantly and negatively correlated with cumulative precipitation, global radiation and litter temperature (Supplementary Figures 3D-F). FAG litter decomposition in the 2-mm-mesh litterbags was significantly and positively correlated with cumulative precipitation (Supplementary Figure 3G).

Significant regressions were found between the decomposition coefficients of the 8 litters and their initial biochemical properties (linear and exponential models, P<0.05; Figure 5, Supplementary Table 4). The linear model yielded the best coefficients of determination, except for the lignin:N *ratio* which fitted better with the exponential model. The decomposition coefficients were better explained by the litter biochemical properties in the 2-mm-mesh litterbags than in the 5-mm-mesh litterbags. The best relationships with the decomposition coefficients were given by the cellulose content, the lignin content, the LCI, and the lignin:N *ratio* (Figures 5C-J), which were negatively correlated with the decomposition coefficients whatever the litterbag mesh size. The decomposition coefficients were positively correlated with soluble compounds, but they were not significantly correlated with C, N, hemicellulose, the polysaccharide content, SLA or the C:N, soluble compound:N and polysaccharide:N *ratios* (Supplementary Table 4).

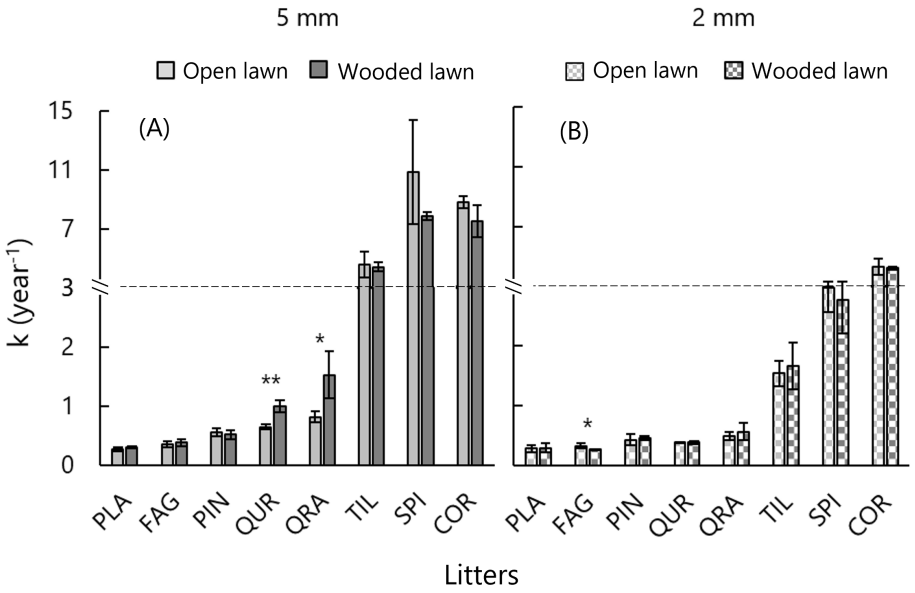


FIGURE 2
Annual decay rates (k, % y^{-1}) of 8 litters in 5-mm-mesh **(A)** and 2-mm-mesh **(B)** litterbags in open and wooded lawns after 369 days. PLA, *Platanus* × *acerifolia*; FAG, *Fagus sylvatica*; PIN, *Pinus radiata*; QUR, *Quercus robur*; QRA, *Quercus rubra*; TIL, *Tilia* × *europaea*; SPI, *Spiraea vanhouttei*; COR, *Cornus alba*. Dotted lines, scale change in the y axis. Error bars, standard deviations from the mean (n=3). *P<0.05; **P<0.01 (t-test for mesh sizes between two lawn types for one litter).



FIGURE 3
Annual decay rate (k, % y^{-1}) of 8 litters from open **(A)** and wooded **(B)** lawns in 5-mm-mesh and 2-mm-mesh litterbags after 369 days. PLA, *Platanus* × *acerifolia*; FAG, *Fagus sylvatica*; PIN, *Pinus radiata*; QUR, *Quercus robur*; QRA, *Quercus rubra*; TIL, *Tilia* × *europaea*; SPI, *Spiraea vanhouttei*; COR, *Cornus alba*. Dotted lines, scale change in the y axis. Error bars, standard deviations from the mean (n=3). *P<0.05; **P<0.01; ***P<0.001 (t-test for mesh sizes).

4 Discussion

4.1 Effect of litter quality on decomposition

Our results show that litter quality was the primary driver of decomposition, supporting our hypothesis (iii). The decomposition patterns of the 8 litters were similar in both lawn types, except for the *Q. robur* and *Q. rubra* litters which decomposed faster in the wooded lawns, probably revealing a preferential substrate for the macrofauna after labile litter decomposition. Five recalcitrant tree litter types (*Platanus*, *Fagus*, *Pinus*, *Q. robur* and *Q. rubra*) decomposed slowly compared to the 3 labile tree or shrub litter types (*Tilia*, *Spiraea* and *Cornus*) (Supplementary Figure 4). The litter decomposition coefficients recorded in the present study for *Fagus, Pinus, Q. robur, Q. rubra* (from 0.30 to 0.53 y^{-1}) *Tilia* and *Cornus* (from 1.60 to 4.27 y^{-1}) in the 2-mm-mesh litterbags (Supplementary Figure 5) are in the range of those recorded in various temperate and continental forest ecosystems [e.g (22, 57, 58)]. These results suggest an effective decomposing activity in lawns even when excluding macrofauna.

The significant negative relationships observed in our study between the decomposition coefficients and the initial cellulose and lignin contents and the lignocellulose index are in line with literature [(59, 60); Figures 5C-J], and show that these indicators are good predictors of litter decomposition. Cellulose and hemicellulose are energy-rich polysaccharides, while lignin is a polyphenolic compound that likely acts as an energy sink in catalytic processes. As cellulose is usually embedded in lignin in plant

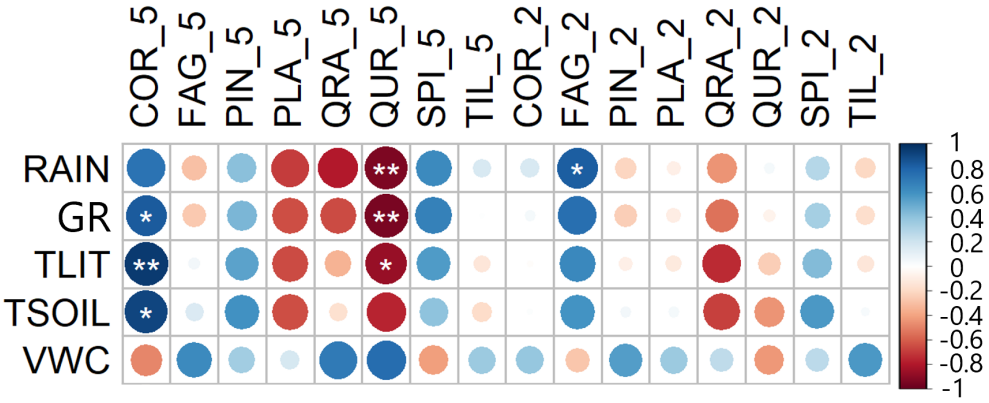


FIGURE 4
Pearson's correlation tests of the annual decomposition coefficients of 8 litters in 5-mm-mesh (e.g., COR_5) and 2-mm-mesh (e.g., COR_2) litterbags with daily cumulative microclimate parameters. RAIN, cumulative precipitation; GR, cumulative global radiation; TLIT, cumulative litter temperature; TSOIL, cumulative soil temperature; VWC, cumulative soil volumetric water content. PLA, *Platanus* × *acerifolia*; FAG, *Fagus sylvatica*; PIN, *Pinus radiata*; QUR, *Quercus robur*; QRA, *Quercus rubra*; TIL, *Tilia* × *europaea*; SPI, *Spiraea vanhouttei*; COR, *Cornus alba*. Blue, positive correlations; red, negative correlations. The circle diameter and the color intensity are proportional to the coefficient of correlation. *P<0.05; **P<0.01.

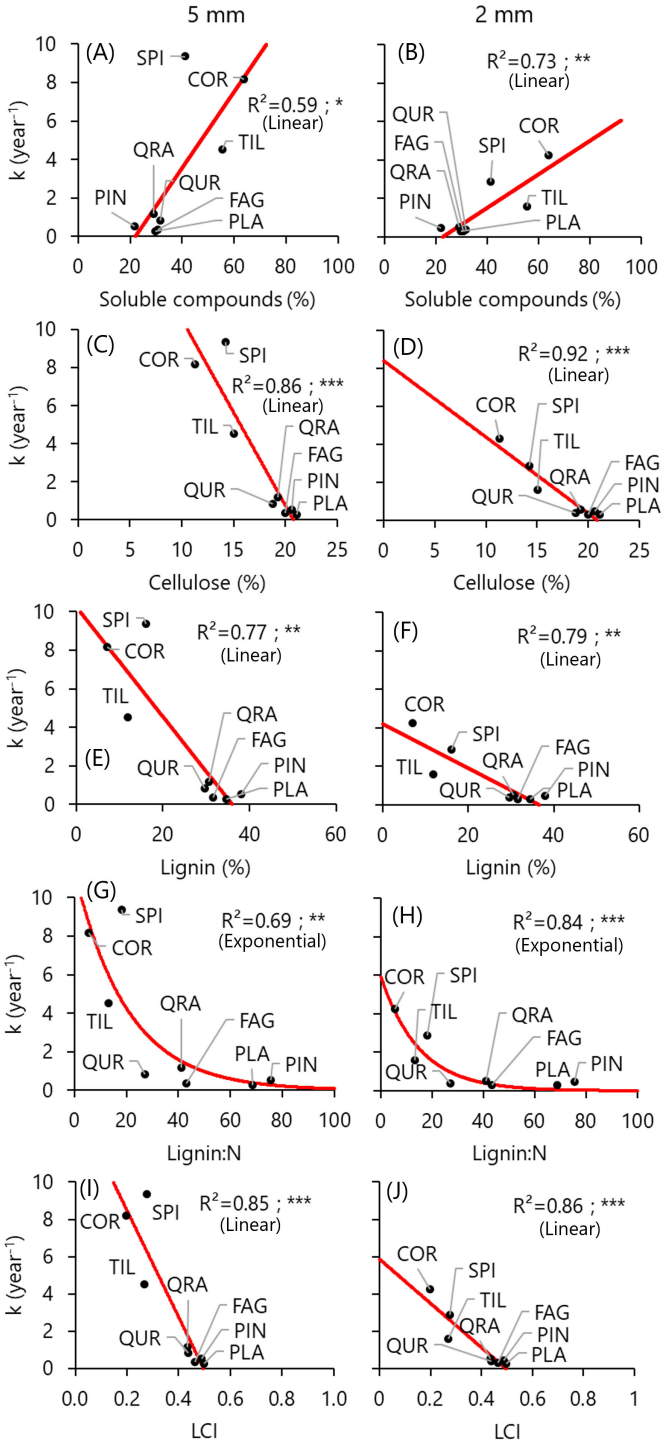


FIGURE 5
Significant linear and exponential regressions of the annual decay rates (k, % y^{-1}) of 8 litters with the initial soluble compounds (**(A, B)**; %), the cellulose (**(C, D)**; %) and lignin contents (**(E, F)**; %), the lignin:N *ratio* **(G, H)** and the lignocellulose index (LCI) **(I, J)** in 5- and 2-mm-mesh litterbags. PLA, *Platanus* × *acerifolia*; FAG, *Fagus sylvatica*; PIN, *Pinus radiata*; QUR, *Quercus robur*; QRA, *Quercus rubra*; TIL, *Tilia* × *europaea*; SPI, *Spiraea vanhouttei*; COR, *Cornus alba*. R^2, coefficient of determination. *P<0.05; **P<0.01; ***P<0.001.

litter, polysaccharide depolymerization by extracellular enzymes is slowed down proportionally to the lignin content (61, 62). Consequently, the LCI of different litter types was a good predictor of decomposition coefficients in our study because it influences the microbial allocation of extracellular enzymes that catalyze lignocellulose degradation (62). Thus, the high initial lignin content of *Platanus*, *Fagus*, *Pinus*, *Q. robur* and *Q. rubra* litters resulted in lower decomposition rates compared to *Tilia*, *Spiraea* and *Cornus* litters (Figures 5E-F). Similar trends are documented in literature for *Platanus*, *Fagus*, *Pinus*, *Q. robur*, *Q. rubra*, with 0.1 to 4.5 times lower decomposition rates than more labile litter types [e.g (25, 57, 63)]. Initial soluble compounds also explain the decomposition

process: their high proportions make litter more accessible to microbes and more palatable to invertebrates, and thus facilitate the initial decomposition stage [(13, 21); Figures 5A, B]. These results contribute to explain the faster decomposition of *Tilia*, *Spiraea* and *Cornus* litter compared to the recalcitrant tree litter types. Additionally, the good prediction of the decomposition coefficients by the lignin:N *ratios* is consistent with the findings of other studies (59, 64) and demonstrates that this indicator comprehensively reflects litter quality.

Although the C:N *ratio* has been widely recognized as one of the best predictors of litter decomposition, this indicator was poorly correlated to the decomposition coefficients in our study because the plant species with the lowest C:N *ratio* are not always those whose litter decomposes faster (21, 55). *Q. robur* was among the species whose litter decomposed slowly despite a relatively low C:N *ratio* (Supplementary Table 4). Conversely, *Q. rubra*, *Tilia* and *Spiraea* litter decomposed faster than *Q. robur* litter despite their higher C:N ratios. The morphological functional traits of litter (e.g., SLA) may also be predominant explanatory factors of the decomposition process (21, 65). However, the decomposition process did not appear to be positively influenced by SLA in our study because of the predominant influence of biochemical properties.

4.2 Effect of microclimate on decomposition

Contrary to our expectations, microclimatic variations induced by trees had limited influence on litter decomposition, thus not supporting hypotheses (i) and (ii). The trees above the lawns induced a microclimate from May to October (Figure 1), which matched with the optimal presence period of *Q. rubra* leaves that shadowed the soil surface. The effect of leaf growth, which started in mid-April, was clearly visible on global radiation, which started to decrease in May in the wooded lawns, whereas it increased continuously throughout spring and summer in the open lawns. Thus, global radiation was reduced by 63% in the wooded lawns compared to the open lawns from May to October, in line with (66) who found that an urban tree (*Malus coccinella*) can intercept up to 85% of global radiation. Consequently, litter and soil temperatures were also reduced in the wooded lawns by 3.6 °C and 1.6 °C, respectively (Figures 1B, C). A strong average precipitation interception of 30% by the tree canopy was measured in the wooded lawns from May to October (Figure 1B), and exceeded the 23% interception measured by (67) under urban deciduous trees (*Albizia procera*). Paradoxically, soil moisture was not reduced in the wooded lawns (Figure 1C), probably because of lower evapotranspiration than in the open lawns. These results are in line with (7), who found that the presence of trees in 12 lawns located in different UGSs decreased the soil surface temperature by 2 °C on average without influencing soil moisture.

In our study, litter decomposition was unrelated to the microclimate variations induced by lawn trees (Figures 4, S3). The potential positive effect of increased temperature on litter decomposition in the open lawns (+1.6 °C between May and October 2022) was likely constrained by limited moisture conditions (Figure 1). During this period, soil moisture was at its lowest in both the open and wooded lawns, while temperature differences between the two lawn types were greatest (Figure 1C). The lack of relationship observed here is consistent with previous studies, which show that soil moisture can have limiting effects on litter decomposition in temperate environments (28, 68, 69). They are also in line with (26), who showed that C quality (i.e., lignin content) drives decomposition more than microclimate does. In our study, litter quality and moisture appear to have strongly and jointly influenced the decomposition process (70, 71).

4.3 Effect of soil properties on decomposition

Our results indicate that macrofauna contributed more strongly to the decomposition of labile litter, supporting hypothesis (iv), while soil properties had a limited influence. The minor differences in soil physicochemical properties (Table 1) and earthworm abundances (Supplementary Figure 1) observed between open and wooded lawns suggest that variations in microclimate (Figure 1) and grass clipping retention (9 times higher in open lawns) had limited impact on soil conditions, despite the age of the trees in wooded lawns (around 45 years). Additionally, the regular practice of raking tree litter likely limited the extent to which trees could influence soil properties (6, 10). This disturbance can disrupt plant-soil interactions by preventing them from achieving a steady state even decades after tree planting (6). As a result, similar soil properties may persist across different vegetation types irrespective of park age.

As expected, our results show that macrofauna contributed more to the decomposition of labile litters (*Tilia, Spiraea, Cornus*), as decomposition was faster in the 5-mm-mesh litterbags than in the 2-mm-mesh litterbags [(36, 72); Figure 3]. The preference of macrofauna for labile litter may be explained by its high N and soluble compound contents (Table 2), hence greater palatability to earthworms. Literature results show that litter quality strongly drives the activity of earthworm populations, and consequently the decomposition process (57, 73). Unlike recalcitrant litter, labile litter does not require prior microbial or abiotic degradation before ingestion by earthworms (74).

In our study, the contribution of macrofauna to decomposition was greater in the *Q. robur* and *Q. rubra* litters, whose decomposition was accelerated in the 5-mm-mesh litterbags in both lawn types (Figure 3), but significantly more so in the wooded lawns (54% faster for *Q. robur* litter and 87% for *Q. rubra* litter; Figure 2A). As *Quercus* was the dominant provider of litter to the wooded lawns, our results may indicate an acceleration of litter decomposition on soils receiving their own litter (home-field advantage, HFA) thanks to decomposer specialization (75–77). showed that an HFA could accelerate litter mass loss by 9% on average, with a maximum of 20%. The higher mass losses of 15 and 22% recorded for *Q. robur* and *Q. rubra* litter in the wooded lawns, respectively (Supplementary Figure 5), could indicate a significant HFA. This potential HFA was observed despite the short residence time of native *Q. rubra* leaves on the soil surface because they were raked in fall. In boreal climate, the presence of trees on a lawn facilitated the decomposition of native tree litter despite the export

of aerial litter after two years of a litterbag experiment in parks > 70 years old (30). Nevertheless, the HFA hypothesis remains speculative, as no reciprocal transplantation experiments under other tree species have been carried out (76).

One limitation of our study is the lack of spatial replication of wooded and open grass areas within the urban landscape. Comparisons between these habitat types would ideally involve multiple paired replicates within the city, ensuring similar pedoclimatic contexts and orientations for valid and consistent ecological comparisons. However, replicating such comparisons in multiple UGSs posed significant logistical and methodological challenges. More specifically, ensuring comparable pedoclimatic conditions and securing permissions to instrument public spaces are significant obstacles. Instead, we focused on a large, secure study site that allowed for controlled experimentation, complemented by emphasizing a wide range of litter types to capture ecological variability. While future studies should aim to incorporate greater spatial replication, our approach represents a valuable contribution by prioritizing methodological rigor and addressing practical constraints inherent to urban research.

5 Conclusions

The aim of our work was to understand how the relative influence of litter quality, microclimate and soil properties can influence the decomposition process of frequently encountered tree and shrub litter types in lawns. Our results show that decomposition rates were primarily driven by litter quality, with recalcitrant litters (i.e., with a high lignocellulose content) decomposing more slowly than labile ones, supporting hypothesis (iii). In contrast, despite marked differences in microclimatic conditions between lawn types (reduced global radiation, reduced litter and soil temperatures and reduced precipitation), decomposition was unrelated to microclimate, thus not supporting hypotheses (i) and (ii). Similarly, soil properties showed limited variation between lawn types and had little effect on decomposition. Macrofauna contributed more to the decomposition of labile litter, supporting hypothesis (iv).

Overall, these findings indicate that litter quality outweighs microclimatic and soil effects as a driver of decomposition in urban lawns. They also suggest that wooded lawns can effectively decompose a wide range of litter types in a temperate UGSs. Maintaining litter under trees could therefore support soil C processes without impairing decomposition. Future research should investigate the long-term effects of litter retention on soil properties and C sequestration under contrasting management practices and soil conditions.

Data availability statement

The datasets presented in this study can be found in online repositories. The names of the repository/repositories and accession number(s) can be found below: https://data.mendeley.com/preview/x9rmm264df?a=82f65f3d-ad94-4651-b9a2-c2e7fc13f465.

Author contributions

TK: Conceptualization, Data curation, Formal Analysis, Investigation, Methodology, Resources, Software, Supervision, Validation, Visualization, Writing – original draft, Writing – review & editing. PC: Conceptualization, Funding acquisition, Methodology, Project administration, Supervision, Validation, Visualization, Writing – review & editing. VG: Conceptualization, Methodology, Supervision, Validation, Visualization, Writing – review & editing. RG: Conceptualization, Data curation, Formal Analysis, Funding acquisition, Investigation, Methodology, Supervision, Validation, Visualization, Writing – review & editing.

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Conflict of interest

The author(s) declared that this work was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fsoil.2026.1777598/full#supplementary-material

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