Soil CO_2, CH_4 and N_2O fluxes in urban forests, treed and open lawns in Angers, France

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Abstract

Urban green spaces (UGSs) are mostly represented by lawns and forests. These UGSs can store carbon in soil and above-ground biomass, potentially modulated by management intensity and vegetation cover (shading, rainfall intercept, litterfall, …). Trees in lawns can create a local microclimate modifying soil biogeochemical cycles affecting in turn greenhouse gas (GHG) emissions. The objective of this study was to assess the effects of trees on microclimate (temperature and moisture) influencing GHG in contrasted UGS types. We monthly monitored (from March to November 2021) and compared soil CO_2, CH_4 and N_2O fluxes simultaneously with surface temperature and moisture in treed lawns, open lawns and urban forests. Lawns included 4 different management intensities including mowing, irrigation and fertilization practices. Temperature was the best predictor of soil respiration in all UGS types studied and was the highest in open lawns. We showed that moisture reflected by the water filled pore space (WFPS) significantly added on variation explanation. The shading of trees strongly decreased soil respiration in treed lawns while soil properties were similar indicating a straightforward effect of lowering temperature. On the contrary, forests deeply changed soil properties as well as decreased soil temperature resulting in the lowest rates of soil respiration. Urban forests are a sink for CH_4 throughout the year. Lawns were weak to mitigate CH_4 and a source of CH_4 in irrigated parks where WFPS overpassed 75%. N_2O fluxes were weak probably reflecting the transition already made from mineral to organic fertilization limiting N availability.

Introduction

Urban areas cover 3% of the Earth's land surface and may contain 10% of organic carbon stocks contributing to the mitigation of increasing urban greenhouse gas (GHG) emissions (Churkina et al. 2010; Lal 2012; Canedoli et al. 2020). To counterbalance these emissions, it is important to maintain and even increase C stocks in urban areas, especially in soil and vegetation that contribute the most to this process (Churkina et al. 2010; Churkina 2012). Urban green spaces (UGS) could be therefore an important lever to this mitigation (Lal 2012) but these processes need to be evaluated.

Lawns are the most representative in terms of vegetation cover and management in UGSs (Churkina et al. 2010). In these areas, C and N inputs are made of organic compounds (grass clippings, dead roots, root exudates…) or minerals (fertilization, atmospheric deposition…). Despite the apparent spatial homogeneity of these herbaceous ecosystems, lawns are often contrasted in terms of vegetation cover as they also contain trees with a heterogeneous spatial distribution. This difference of vegetation cover is distinguished in this study as open and treed lawns. Open lawns are almost permanently exposed to the sun with inputs mostly coming from mowing and root grass turnover. Treed lawn formed at least by several trees, can create a microclimate by lowering soil temperatures and moisture, but also by locally modifying biogeochemical cycles and litter decomposition processes (Livesley et al. 2016; Nidzgorski and Hobbie 2016). Moreover, under deciduous trees, the litterfall in parks is regularly raked in autumn, thus litters do not decompose under trees or in the contrary, is concentrated under few trees for commodity. By controlling these environmental factors, treed lawns can therefore modify the C and N cycles and thus the driving processes of CO_2, CH_4 and N_2O emissions from root and soil microbial activities. Studying the influence of treed lawns adjacent to open lawns on soil C and N outputs is important to understand the potential contribution of UGSs functioning and management types to GHG emissions in ever-expanding cities.

Herbaceous and tree-dominated UGSs have been well studied in urban areas, through the study of lawns (Livesley et al. 2010; Qian and Follett 2012) and urban forests (Yesilonis and Pouyat 2012). Intensively managed lawns (irrigated and fertilized with mulching of mowing residues) have been identified as systems with high C sequestration potential (Pouyat et al. 2009; Qian and Follett 2012) with important indirect CO_2 emission as well as a potential negative impact with an increase in N_2O emissions whether mineral fertilization and irrigation is not adjusted to plant needs (Townsend-Small and Czimczik 2010; Livesley et al. 2010). Urban forests, that rarely resist to urban sprawl, provide great services such as storing more C than lawn soils due to high C inputs from litter (Yesilonis and Pouyat 2012). Moreover, soil studies rarely take into account the huge C biomass stock in woody plants leading to missing component to compare ecosystems processes (Sun et al. 2019). Furthermore, forest soils are a low source of CO_2 and N_2O emissions (Weissert et al. 2016; van Delden et al. 2018) as well as a strong CH_4 sink (Costa and Groffman 2013; van Delden et al. 2018) due to the high C:N of litterfall favoring N immobilization and lower bulk density as well as lower moisture content that increase the rate of diffusivity of CH_4 into the soil.

The biogeochemical processes in UGSs have barely been investigated when herbaceous and trees are interacting as a treed lawn. It is necessary to investigate the impact of this UGS type to decide whether it should be favored or not under temperate climate. The influence of trees on CO_2 and CH_4 fluxes in lawns has been studied recently by Lu et al. (2021) under Boreal climate and they found that CO_2 fluxes were twice lower in deciduous treed lawns compared to open lawns and they attributed this effect to the lower decomposability of tree litter (i.e. mainly fine roots). These authors also found higher CH_4 consumption in treed lawns than in open lawns but the potential reasons for these differences were not discussed. Literature has not yet assessed the microclimatic effects of lawn trees on the soil respiration process in an urban environment, and how much shading can drive this soil respiration lowering. We suspected that microclimatic effects of trees inducing changes in soil temperature and moisture (Nidzgorski and Hobbie 2016), could explain soil respiration variations (Oertel et al. 2016). In a non-urban environment, Smith and Johnson (2004) showed that the microclimatic effect of tree in a forest could reduce soil respiration by 38% compared to a grassland. This reduction was mainly attributed to the shading of the trees decreasing soil temperature. Thus, there is a need to address the question of how the presence of trees in lawns can change the soil surface temperature and moisture influencing soil respiration and C stocks.

Studies on UGS have mostly focused on intensively managed lawn (i.e. irrigated, fertilized and mulched) systems (Frank et al. 2006; Qian and Follett 2012) thus, with strong stimulation of C and N cycling. However, municipalities worldwide are currently changing the way they manage UGSs, by selecting sustainable action plans preventing to waste natural resources (Ignatieva et al. 2020; Pantaloni et al. 2022), such as soil, water and nutrients use. Specific management practices (irrigation, organic fertilization, varying mowing frequencies, mulching, etc.) are therefore implemented to obtain the desired landscape and maintain it, in the long term. Thus, UGSs can be classified into increasing levels of intensity management depending on several factors potentially nested (Table 1): the size of the UGS, plant cover type, UGS functions (e.g. leisure, aesthetic, environmental or ecological), location in the urban area (e.g. city center or periphery) and their landscape impact. Data of the impact of management practices on soil properties leading to GHG emissions are still lacking (Thompson and Kao-Kniffin 2019).

The objective of this study was to measure soil GHG emissions in treed lawns and open lawns under gradient of management intensity, and to relate the microclimatic effect of trees (soil temperature and moisture) to these gas emissions in a temperate urban context. We mainly hypothesized that (1) shading of tree in lawns limits soil respiration by lowering soil surface temperature (2) change in soil biochemical properties under trees can slow down soil respiration; (3) urban forest and treed lawns with lower bulk density and both soil resource and moisture content exhibit higher CH_4 consumption; (4) N_2O emissions increase with management intensity of lawns because of fertilization and irrigation that increase both nutrient availability and soil water content.

Materials And Methods

Study area and sites

Our study was conducted in 15 UGSs distributed in the city of Angers, France (47° 28' 25" N, 0° 33' 15" W) (Table S1). According to the Köppen-Geiger classification (Kottek et al. 2006), Angers' climate is warm and temperate (type Cfb, warm temperate - fully humid - warm summer). Over the period 1981–2010, Angers had an average annual temperature of 11.5°C. Annual rainfall averaged 693 mm and was well distributed throughout the year.

Ugs Types: Urban Forests, Open And Treed Lawns

In 2021, the city presented 1 500 ha of UGSs, among which 49% (735 ha) included grasslands mainly composed of lawns and 17% (225 ha) are wooded areas mainly composed of urban forests more or less artificial. The selected 27 sites within the 15 UGSs (Table S1), included 3 urban forests, 12 treed lawns associated to 12 open lawns. All sites were established for at least 18 years. Within each site, we chose to focus mainly on deciduous treed lawns because they are predominant in Angers.

Ugs Management

The 12 treed and open lawns presented 4 increasing management intensities regarding of mowing frequency and cutting height, with or without restitutions, irrigation and organic fertilization (Table 1). In treed lawns litterfalls are raked every 15 days during the defoliation period (from October to January). Urban forests were not managed.

Table 1
Management practice types and frequency carried out for management intensity on treed and open lawns in the 24 sampled lawns. Urban forests correspond to no management at all

UGS types in Angers	**Urban forests**	**Lawns**			
	(n = 3)	Intensity 1 (n = 3)	Intensity 2 (n = 11)	Intensity 3 (n = 2)	Intensity 4 (n = 8)
Management practices					
Restitution of grass clippings	No	Yes	Yes (n = 8) or no (n = 3)	No	No
Cutting frequency	No	2–3/year	2/month	2/month	4/month
Cutting height (cm)	No	10	7	7	7
Irrigation	No	No	No	Irrigation (2mm, 3/week)	Irrigation (2mm, 3/week)
Fertilization	No	No	No	No	Organic fertilization (3/year)
% sites (n = 694)	1	9	73	16	1
% total UGS surface area (%, 1 500 ha)	20	41	32	3	4

Soil Properties

According to the French soil classification (Baize and Girard, 2009), soil analyses of the 15 UGS soils were classified into 4 soil types (Table S1). These soils had mainly a sandy loamy and loamy texture. We analyzed soil biophysicochemical properties (total organic C, total N, total P, microbial biomass, pH, bulk density, soil texture, CEC, EC and exchangeable elements). Soils were sampled between 0 and 10 cm depth with an auger (Ah horizon). We verified that the level of urbanization, defined as difference in temperature between the rural and urban environment (Δ T_{urban} - T_{rural}) (Peng et al. 2012; Heisler and Brazel 2015), was not related to soil surface temperature or soil N content (Table S2).

In situ **GHG fluxes, soil temperature and moisture measurements**

The soil CO_2, CH and N O flux were monthly measured in 2021 (i.e. March, April, May, June, July and November) in the 27 sites. As Smith and Johnson (2004), we have defined the growing season (GS) as the period of the year when the air temperatures recorded by the local weather station are no longer negative. GS therefore extended from April 13th 2021 to November 05th 2021. The sampling period corresponding to GS was therefore April, May, June and July. The non-growing season (NGS) corresponded to the sampling months of March and November.

The sampling spots were selected to be a representative area of the UGS types (slope, vegetation, …). For the measurement of CO , we used 2 automated infrared analyzers CFLUX-1 (PP Systems, Amesbury, USA) with a sampling surface of 0.032 m^2 and volume of 2.3 L. For the measurement of CH and N O, we used a Fourier transform infrared analyzer DX4040 (Gasmet Technologies Oy, Helsinki, FINLAND) associated with a manual dynamic closed chamber system (6.6 L) with a sampling surface of 0.042 m^2. All gas samples were taken between 09:00 and 13:00 because this time was considered to be representative of daily fluxes (Kaye et al. 2004, 2005). CO measurements were made every 15 min. For the measurement of CH and N O, two measurements were made per spot (between 9:00 and 11:00 and between 11:00 and 13:00) to cover the daily temporal increase in GHG fluxes. In order to obtain linear increases of GHG fluxes in the measurement chamber, the time of chamber closure was 5 min for CO_2 measurements and 15 min for CH_4 and N_2O measurements.

Soil surface temperature and the volumetric humidity at 5 cm depth were monitored for each sampling point with a TRIME-PICO 32 TDR probe (IMKO Micromodultechnik GmbH, Ettlingen, GERMANY). Water-filled pore space (WFPS) was calculated using the following equation (Robertson and Groffman 2015):

$$WFPS = \frac{SWC}{1 - \frac{BD}{PD}} \times 100\,(1)$$

where SWC is volumetric soil water content (vol. %), BD is bulk density ($g.cm^{-3}$) and PD is particle density (2.65 $g.cm^{-3}$).

Data Processing And Statistical Analysis

GHG fluxes were calculated from linear increase (CO_2 and N_2O) or decrease (CH_4) in gas concentration per unit of time (Butterbach-Bahl et al. 2016), corrected for chamber volume, sampled surface area, air temperature and atmospheric pressure (Barton et al. 2007). The atmospheric pressure could not be assessed in the manual chamber so we did not integrate it in the CH_4 and N_2O flux calculation. GHG fluxes, soil surface temperatures and WFPS collected between 09:00 and 13:00 were averaged to obtain the mean daily values. These daily values were averaged for each UGS type or management intensity and were used as the average values of the month.

One-way-repeated measures analysis of variance (rm-ANOVA) was used to determine the effects of UGS type and management intensity across time on GHG fluxes, soil surface temperature and WFPS. When a significant interaction was found, we separately analyzed the effects of UGS type (urban forest, treed lawn, open lawn) or management intensity (intensity 1, 2, 3, 4) for each sampling time by one-way ANOVA followed by the Tukey post-hoc tests (at $P < 0.05$) to analyze in detail the variations between each UGS type or management intensity and for each sampling time. One-way ANOVA followed by Tukey post-hoc tests (at $P < 0.05$) was also used to determine the effects of UGS type or management intensity on soil properties. Microbial biomass was 'n + 3' log normal transformed prior to statistical analyses to meet the assumption of normality and homogeneity of variances. For CH fluxes, N O fluxes, organic C, total N and total P contents data normality could not be obtained. Thus, a Kruskal-Wallis test followed by Mann-Whitney post-hoc tests were carried out to test the effects of UGS type and management intensity on the non-normal datasets. To test the effect of restitution (only within management intensity 2) on GHG fluxes, total N and microbial biomass contents we used a homoscedastic student t-test if normality of datasets could be obtained. For non-normal datasets (N_2O fluxes of June, organic C content) we used a Wilcoxon rank-sum test.

Simple linear and polynomial regression models were used to analyze the effects of temperature or WFPS on GHG fluxes. To model the soil respiration with the specific effect of temperature and with the combined effects of temperature and WFPS, we compared 9 models referenced by Weissert et al. (2016) (Table S4) based on statistical criteria such as R^2 and RMSE.

Annual net losses of C as CO_2 were estimated on the basis of measured soil surface temperatures. To simulate the cumulative C fluxes from urban forests, treed lawns and open lawns, air temperatures recorded at the weather station of Beaucouzé (Pays de la Loire, FRANCE) were used. The regression equations of soil surface temperature with air temperature (Fig. S1) were then used to predict the daily mean soil surface temperature in 2021. The equations of the best fit soil surface temperature only model (Logistic model, Table S4) of the 3 UGS types were used to predict the average daily CO_2 fluxes in urban forests, treed lawns and open lawns. The daily CO_2 fluxes were then summed for GS and NGS and converted into C fluxes. The duration of soil C turnover was calculated on the basis of C stock and heterotrophic soil respiration. In line with several studies in forest and grassland environments, we hypothesized an average annual root respiration of 50% of the total respiration (Hanson et al. 2000; Byrne and Kiely 2006).

We performed Spearman correlation to test the links between GHG fluxes and soil properties as well as between GHG fluxes, soil surface temperature and urbanization level. All statistical analyses were conducted on R software, version 4.1.3.

The sensitivity of CO_2 fluxes to temperature variation can be described by the Q_{10} factor (Oertel et al. 2016). The following equation was used to calculate Q_{10} values in the three studied UGS types:

$$Q_{10} = \frac{R_{T+10}}{R_T} (2)$$

where R_{T+10} is the soil respiration at the initial soil temperature ($R_T, T = 15^{\circ}C$) plus 10°C (Smith et al. 2003; Brisson and Launay 2008). We calculated these values using the linear equations from the regression of soil respiration with soil surface temperature.

Results

CO_2, CH_4 and N_2O fluxes according to the type of UGS or management intensity

CO_2 fluxes were significantly affected by the UGS type depending on the time (Repeated measure ANOVA, F = 2.22, $p < 0.05$, Table 2). Open lawns had systematically higher fluxes than soil from treed lawns and urban forests, the difference increasing with time during the growing season (GS) (Fig. 1a). CO_2 fluxes in treed lawns did not change from urban forests except in March. On average, the CO_2 fluxes in open lawns (1 091 ± 61 mg CO_2 m^{-2} h^{-1}) were 34% higher than CO_2 fluxes from treed lawns (715 ± 46 mg CO_2 m^{-2} h^{-1}) and 52% higher than CO_2 fluxes from urban forests (521 ± 59 mg CO_2 m^{-2} h^{-1}) (Rm-ANOVA main effect, F = 18.8, $p < 0.001$, Table 2).

Table 2
Results of one-way-repeated measures ANOVA testing UGS type or management intensity (Management) and time on CO_2 fluxes, soil surface temperature and WFPS. F and p values are provided for each property for the significance of modalities

		CO_2 fluxes (mg m⁻² h⁻¹)		**Soil surface temperature (°C)**		**WFPS (%)**	
Sources	df	F	p	F	p	F	p
Between-subjects							
UGS Type	2	**18.8**	**< 0.001**	**18.1**	**< 0.001**	0.14	0.870
Within-subjects							
Time	5	**37.3**	**< 0.001**	**98.4**	**< 0.001**	**8.23**	**< 0.001**
UGS Type x Time	10	**2.22**	**0.035**	1.80	0.098	1.31	0.280
Between-subjects							
Management	3	0.93	0.444	0.86	0.4786	**8.38**	**< 0.001**
Within-subjects							
Time	5	**28.8**	**< 0.001**	**158.1**	**< 0.001**	**21.03**	**< 0.001**
Management x Time	15	0.91	0.552	1.50	0.140	**2.35**	**0.011**

The CH_4 fluxes were significantly higher in urban forests than in open lawns and in treed lawns (Fig. 1b) during the 3 consecutive months recorded (June, July and November). On average, the CH_4 fluxes in urban forest (− 0.19 ± 0.04 mg m⁻² h⁻¹) were higher than treed and open lawns (− 0.019 ± 0.02 mg m⁻² h⁻¹ and − 0.033 ± 0.01 mg m⁻² h⁻¹, respectively).

No significant differences of N_2O fluxes were found between the 3 UGS types (Fig. 1c). On average, N_2O fluxes in this study reached 0.014 ± 0.002 mg m⁻² h⁻¹. Nevertheless, the variability of N_2O fluxes was high with rates as high as 0.087 mg m⁻² h⁻¹, especially in June.

Management intensity had no significant effect on CO_2 and CH fluxes (Fig. S2). In July, N O fluxes in management intensity 2 (0.006 ± 0.001 mg m⁻² h⁻¹) were significantly lower to those of management intensity 3 and 4 (0.04 ± 0.003 mg m⁻² h⁻¹ and 0.024 ± 0.003 mg m⁻² h⁻¹ respectively) (Fig. 2). In June, several high fluxes were found especially in management intensity 1 reaching 0.087 mg m⁻² h⁻¹.

Effect Of Ugs Type And Management Intensity On Soil Temperature And Wfps

The temperature increased significantly from March to July (Rm-ANOVA main effect, $F = 98.4$, $p < 0.001$, Table 2 and Fig. 3). The UGS type had a significant effect on soil surface temperature (Rm-ANOVA main effect, $F = 18.1$, $p < 0.05$, Table 2) but with no influence of time (UGS x time interaction, Rm-ANOVA, $F = 1.80$, $P = 0.098$) (Fig. 3). The main effect of UGS type indicated a higher averaged temperature in open lawns (16.8 ± 0.3°C, min = 15.2°C, max = 18.4°C) than in treed lawns (14.9 ± 0.3°C, min = 13.6°C, max = 16.6°C) and urban forests (13.6 ± 0.8°C, min = 12.1°C, max = 13.8°C). No significant differences of temperatures were found between management intensities (Table 2).

The water filled pore space (WFPS) significantly changed with time (Rm-ANOVA main effect, $F = 8.23$, $p < 0.001$, Table 2) but no influence of UGS type was found. The WFPS was significantly affected by the management intensity depending on the time (Rm-ANOVA, $F = 2.35$, $P = 0.011$, Table 2). During April, May and July, the lawns managed in intensity 4 had a significantly higher WFPS than management intensity 1, 2 and management intensity 3, but only on July (Fig. 4).

Relationships Between Temperature, Wfps And Soil Properties With Ghg Fluxes

Different models were computed and parameters and performances are reported in Table S4.

CO_2 fluxes

CO_2 fluxes were significantly correlated to soil temperature with 54 to 58% of the variation explained in the 3 UGS types, urban forests with the lowest and open lawns with the highest slopes (Fig. 5a, b, c). The WFPS did not correlate with CO_2 fluxes (Fig. 6a, b, c) but a site effect was found showing either positive or negative correlations ($r^2 = 0.68$ to 0.80) for each urban forest with unique behaviors. Treed lawns and open lawns had a site effect showing negative links between WFPS and CO_2 fluxes with highly different slopes, regression types and variation explained ($r^2 = 0.36$ to 0.96).

CH_4 fluxes

CH_4 fluxes showed no clear link with soil surface temperature in urban forests and treed lawns (Fig. 5d, e, f) but a slight negative correlation was found in open lawns ($r^2 = 0.14$, $p < 0.05$). More specifically, in urban forests, one site showed a strong positive correlation between temperature and CH_4 fluxes ($r^2 = 0.72$, $p < 0.001$, polynomial regression) whereas the other sites showed negative links ($r^2 = 0.20$ and r2 = 0.60, $p < 0.05$, polynomial regression).

CH_4 fluxes were positively correlated to the WFPS (Fig. 6d, e, f) with the highest variation explained in urban forests (r^2 = 0.51, $p < 0.001$) intermediate in treed lawns (r^2 = 0.45, $p < 0.01$) and weak in open lawns (r^2 = 0.21, $p < 0.05$), all revealing different UGS type behaviors but all regressions increasing.

N_2O fluxes

N_2O fluxes and soil surface temperature showed no general correlation for treed and open lawns (Fig. 5g, h) and a significant and positive correlation in urban forests (Fig. 5i) was found (r^2=0.14, $P < 0.05$, polynomial regression). Individual correlations by sites showed very contrasted behaviors with positive or negative correlations from non-significant to very significant correlations (r^2 = 0.01 to 0.96) in both treed and open lawns. During November, data under 10°C showed a significant and negative correlation between temperature and N_2O fluxes (r^2 = 0.40, $p < 0.05$). During June and July, after removing 4 extreme values of N_2O (above 0.040 mg N_2O m^2 h^{-1}, see discussion) among 46 measurements, a general increase in N_2O fluxes positively correlated with WFPS (r^2 = 0.35, $p < 0.01$).

Selected UGS types (number 14 and 15, Table S1) presenting WFPS higher than 75% showed a significant increase in N_2O fluxes with temperature (r^2 = 0.71, $p < 0.001$). When temperature ≥ 20°C (i.e. during June and July) we found a significant increase in N_2O with WFPS (r^2 = 0.68, $p < 0.001$).

Modelling of soil respiration with the combined effects of soil surface temperature and WFPS.

Among the 9 models tested to predict CO_2 fluxes (Table S4), the power-logistic model, integrating the specific effects of temperature and WFPS, gave the best fits for urban forests (r^2 = 0.61, RMSE = 123 mg CO_2 m^{-2} h^{-1}), treed lawns (r^2 = 0.55, RMSE = 206 mg CO_2 m^{-2} h^{-1}), and open lawns (r^2 = 0.67, RMSE = 280 mg CO_2 m^{-2} h^{-1}).

By separately analyzing irrigated and non-irrigated lawns (Table S4), the best modelling of CO_2 fluxes in irrigated lawns were also obtained with the power-logistic equation (r^2 = 0.65, RMSE = 200 mg CO_2 m^{-2} h^{-1} for treed lawns and r^2 = 0.78, RMSE = 249 mg CO_2 m^{-2} h^{-1} for open lawns). CO_2 fluxes of non-irrigated lawns were better modelled with the power-logistic equation (r^2 = 0.55, RMSE = 200 mg CO_2 m^{-2} h^{-1} for treed lawns and r^2=0.70, RMSE = 248 mg CO_2 m^{-2} h^{-1} for open lawns).

Predicted Cumulative C Flux From Soils

Soil surface and air temperatures were strongly correlated (r^2=0.88, $p < 0.001$ in urban forest, r^2=0.88, $p < 0.001$ in treed lawns and r^2=0.83, $p < 0.001$ in open lawns, Fig. S1). Annual predicted cumulative C fluxes of urban forests (1 136 g C m^{-2}) and treed lawns (1 457 g C.m^{-2}) were 53.2 and 37.5% lower, than open lawns (2 295 g C m^{-2}) (Fig. 7).

Soil properties and GHG fluxes depending on UGS types and management intensities.

Soil properties were significantly different depending on UGS types and management intensities: total P, pH and bulk density were significantly lower in urban forests than in treed and open lawns (Table 3). Total organic C, total N, total P were significantly lower and bulk density was significantly higher in management intensity 2 than management 4 (Table 3).

Table 3
Soil properties shown according to either UGS types or management intensities. Values are means and standard errors in parentheses. Bold and different letters indicate significant differences between modalities.

	Total organic C (%)	**Total N (%)**	**C:N**	**Total P (%)**	**Microbial biomass (mg kg^{-1})**	**pH**	**Bulk density (g cm^{-3})**
UGS type							
Urban forest (n = 3)	7.76 (3.33) a	0.48 (0.21) a	12.65 (2.68) a	**0.05 (0.01) a**	1 003 (326) a	**4.62 (0.39) a**	**0.58 (0.15) a**
Treed lawn (n = 12)	4.58 (0.59) a	0.32 (0.03) a	11.65 (0.34) a	**0.11 (0.01) b**	696 (45) a	**6.91 (0.18) b**	**0.96 (0.03) b**
Open lawn (n = 12)	4.17 (0.41) a	0.34 (0.03) a	11.13 (0.46) a	**0.11 (0.02) b**	801 (50) a	**6.61 (0.19) b**	**0.98 (0.07) b**
Management intensity							
1 (n = 3)	**5.24 (1.73) ab**	**0.31 (0.03) ab**	12.41 (1.02) a	**0.07 (0.02) ab**	873 (137) a	6.68 (0.35) a	**0.92 (0.04) ab**
2 (n = 11)	**3.49 (0.35) a**	**0.29 (0.02) a**	11.15 (0.31) a	**0.09 (0.02) a**	681 (37) a	6.58 (0.20) a	**1.10 (0.05) a**
3 (n = 2)	**3.77 (0.20) ab**	**0.31 (0.02) ab**	11.43 (0.30) a	**0.09 (0.00) ab**	791 (50) a	6.29 (0.13) a	**1.01 (0.13) ab**

	Total organic C (%)	Total N (%)	C:N	Total P (%)	Microbial biomass (mg kg⁻¹)	pH	Bulk density (g cm⁻³)
4 (n = 8)	**5.41 (0.56) b**	**0.40 (0.04) b**	11.33 (0.64) a	**0.15 (0.04) b**	782 (73) a	7.15 (0.20) a	**0.82 (0.08) b**

Organic C, Total N (ISO 10694); Total P (ISO 11263); Microbial biomass (ISO 14240-2); pH (ISO 10390); Bulk density (ISO 11272)

Discussion

GHG emission overview in urban green spaces (UGS)

In the current study, CO_2 fluxes recorded in open lawns were in the upper range (from 118 to 1 649 mg CO_2 m^{-2} h^{-1}) of the those recorded in various contexts (e.g. Livesley et al. 2010; Christen et al. 2011; Ng et al. 2015; Shchepeleva et al. 2019) and higher than those recorded in urban forests by Groffman et al. (2009) and Chen et al. (2013) (366 and 199 mg CO_2 m^{-2} h^{-1}, respectively). CO_2 fluxes in treed lawns, were only studied by Lu et al. (2021) under boreal climate (from 35 to 80 mg CO_2 m^{-2} h^{-1}) and were largely lower than those observed in the current study under a warm temperate climate. Furthermore, CH_4 consumption (i.e. negative fluxes) in open and treed lawns was in the upper range (from 0.000 to -0.027 mg CH_4 m^{-2} h^{-1}) of fluxes in literature (e.g. Kaye et al. 2004; Groffman and Pouyat 2009; van Delden et al. 2018; Shchepeleva et al. 2019) and largely above those found in urban forests (from − 0.013 mg CH_4 to -0.158 mg CH_4 m^{-2} h^{-1}) by Goldman et al. (1995), Groffman and Pouyat (2009) and Zhang et al. (2014). Urban forests were an important CH_4 sink as its consumption was 4 times higher than the average consumption rate attributed to forest systems (-0.04 mg m^{-2} h^{-1} according to Le Mer and Roger, 2001). With an average N_2O flux of 0.014 mg m^{-2} h^{-1} in the current study, these rates were weak and well below the values reported for open lawns (from 0.031 to 0.276 mg N_2O m^{-2} h^{-1}) (e.g. Kaye et al. 2004; Groffman et al. 2009; Livesley et al. 2010; Gillette et al. 2016) and for urban forests (0.044 mg N_2O m^{-2} h^{-1}) by Groffman et al. (2009).

The contribution of non-CO_2 GHG (i.e. CH_4 and N_2O expressed on a CO_2 equivalent basis) to the GHG balance were negligible (120 to 1 900 times smaller than average CO_2 fluxes).

Shading Reduced Temperature And Soil Respiration In Treed Lawns And Urban Forests

In treed lawns and urban forests, the canopy of woody species (deciduous trees) limited soil warming through shading of the soil surface and consequently reduced temperature and soil respiration (Wan and Luo 2003; Smith and Johnson 2004) and this effect have been found lasting but least intensive, during the non-growing season with evergreen trees (Lu et al. 2021). The sharpened correlations found in the current study, between soil respiration and soil surface temperature (Fig. 5a, b, c) with and without shading, confirmed that soil temperature is a dominant driver explaining CO_2 flux variations (Oertel et al. 2016) in urban lawns (Shchepeleva et al. 2019) and urban forests (Chen et al. 2013).

More specifically, during the growing season (GS), the presence of trees in lawns reduced soil respiration by 36% (Fig. 1a) paralleling with a decrease in temperature of 2.7°C (-20%) for the same period. The close Q_{10} values (Fig. 5a, b) of treed (1.88) and open lawns (1.90) indicated similar temperature dependence and confirm our hypothesis that tree shading in lawns plays as a limiting factor of soil respiration. Furthermore, moisture was found to be particularly improving soil respiration variation in lawns (Table S4) and even better, by distinguishing irrigated and non-irrigated lawns, revealing a significant contribution of WFPS into modelling. Annual variation of temperature and WFPS were negatively correlated during the year (data not shown). However, this link can be disconnected in urban ecosystems because of irrigation during summer in several UGS, leading to both high moisture and temperature. Moreover, we found that these artificial conditions are not always as they were supposed to be. For example, irrigation supposedly maintaining water supply in management intensity 4 (i.e. patrimonial UGS), reached very contrasted moisture levels leading to different respiration response (see management intensity section below). Soil moisture influences CO_2 production directly by regulating physiological processes of roots and microorganisms, and indirectly via the diffusion of nutrients and O_2 in the soil (Luo and Zhou 2006). It is estimated that the maximum of CO_2 production in soil by heterotrophic respiration is reached when macro pores are filled with air and micro pores are filled with water, i.e. WFPS of the soil is 50–60% (Linn and Doran 1984; Luo and Zhou 2006). Xu et al. (2004) observed that ecosystem respiration is slowed down at a WFPS below 30% whereas above a WFPS of 80%, soil O_2 becomes limiting for soil biological activity. Other studies (e.g. Smith and Johnson 2004; Shchepeleva et al. 2019), concluded that soil moisture was not a determining factor to explain variations of soil respiration in UGS and did not show significant differences. We demonstrated in the current study that considering the artificial conditions of temperature and WFPS during growing-season separating irrigated and non-irrigated lawns led to better predict soil respiration. Whether we cannot exclude a potential role of soil properties on soil respiration, the lack of differences between open and treed lawns (Table 3) and no correlations found (data not shown) could confirm the almost exclusive driver force of soil respiration by temperature and moisture in urban lawns (Oertel et al. 2016).

Trees in urban forests intercept the sunlight and resulted into a temperature decrease of 4.3°C (-32%) compared to open lawns paralleling with a 50% decrease of soil respiration (Fig. 1a). The lower Q_{10} in urban forests (1.76) than lawns (Fig. 5a, b, c) this time clearly showed a temperature sensitivity influenced by other factors, such as soil properties such as organic matter content and quality (Davidson et al. 2006; Conant et al. 2008). This was confirmed in the current study, especially with the lower bulk density, pH, CaO and phosphorus content (Table 3 and Table S3). The differences of soil respiration between urban forests and lawns, probably resulted from combined effect of shading and soil biochemical differences. Indeed, the literature has also shown that the recalcitrant forest litter can slow down the C cycle and thus could limit the observed CO_2 fluxes (Livesley et al. 2016; van Delden et al. 2018). The lower soil respiration in urban forests could also be attributed to the dense surface litter layer inducing an additional shading effect (Sayer 2005).

Lawns Are Weak Ch Sinks Compared To Urban Forests

The lower CH_4 consumption in lawns compared to forests (Fig. 1b) could be explained by the contribution of vegetation cover (litter types, moisture, …) and management practices, the latter affecting nutrient resources, bulk density and WFPS (Table 3). In our study, positive correlation between WFPS and CH_4 fluxes in the 3 UGS types (Fig. 6d, e, f) suggest that CH_4 consumption was favored by the low WFPS values (i.e. strong aerobic conditions). The decrease in WFPS is related to an increase in gas diffusion (CH_4 used as C energy source and O_2 as electron acceptor) and is necessary to methanotroph bacteria oxidizing CH_4 into CO_2 (Serrano-Silva et al. 2014). In urban forest soils, the generally high rate of gas diffusion due to low bulk density and low moisture favor CH_4 consumption (Costa and Groffman 2013; van Delden et al. 2018). Indeed, at the drier conditions of the study (i.e. 20% of WFPS) forest soils consumed up to 6 times more CH_4 than lawns (Fig. 6d, e, f). Other factors such as nature of the herbaceous vegetation whose litter is rich in labile N (Li et al. 2013; Nataningtyas et al. 2017), fertilization and irrigation (van Delden et al. 2018) could have limited CH_4 consumption in lawns. Further analyses should confirm the higher abundance and activity of methanotroph in urban forests and lawns to verify whether the function is limited by these main characters or by environmental conditions or both.

The similar CH_4 consumption in treed and open lawns showed that tree presence affecting microclimatic (i.e. temperature and moisture effects) but not soil properties had little influence on the CH_4 consumption process. One explanation could come from the regularly raked litterfall under deciduous trees would slow down soil enrichment and thus would limit the methanotrophic activity (Le Mer and Roger 2001). In line with that, Lu et al. (2021) showed that coniferous treed lawns, but not deciduous treed lawns, had higher soil organic matter than open lawns and exhibited higher CH_4 consumption and could confirm this hypothesis.

In a less obvious way, we found that some irrigated treed lawns presenting a WFPS exceeding 75% were a source of CH_4 (Fig. 6e), whereas open lawns especially in dry conditions (20% WFPS), could reach similar levels CH_4 consumption of forests (Fig. 6d, f).

Management Intensity Induced Changes In No Fluxes In Lawns

We observed a general decrease in N_2O fluxes from June to November with no effect of management intensity except in July, showing a cut off separating intensity 1 and 2 from intensity 3 and 4 with higher N_2O fluxes (Fig. 2). Despite fertilization events (April, June and October) in intensity 4, no burst of increase was detected. Rather, a specific site management history could have some importance as we found individual and atypical behaviors strongly related to N_2O fluxes modulated by moisture first (above 75% WFPS) and then by temperature with different behaviors whether temperature was below 10°C (i.e. negative correlation) and or above 20°C (i.e. positive correlation).

Due to the unbalanced number of UGS in each management intensity (Table 1), we suggest that interpreting the difference between intensity 2 and 4 (n = 11 and n = 8, respectively) was the most meaningful in this study. N mineral fertilization associated with irrigation has been identified as practices promoting N_2O fluxes in urban lawns by denitrification (Kaye et al. 2004; Livesley et al. 2010) indicating the role of N availability. We did not follow N mineral in the current study, but lawns intensively managed (i.e. intensity 4: fertilized and irrigated) presented greater N_2O fluxes paralleling with soil C, N and P content than intensity 2. However, N_2O in this study were rather low compared to other studies. Soil organic C has been reported to create a C-based sink for inorganic N and thus, limiting N mineral availability and then denitrification process (Qian and Follett 2012). The solid organic fertilizer used in urban parks could have reduced the immediate N availability and thus have alleviated N_2O fluxes (Gregorich et al. 2005). Nevertheless, the combination of high N content and WFPS above 75% in June and July in some irrigated parks resulted into high N_2O fluxes confirming that irrigation was the primary factor promoting N_2O emissions (Livesley et al. 2010). Moreover, high P content was related to high N_2O emissions by stimulating mineralization and nitrification (Mori et al. 2010) and would need to be followed by anoxic conditions for denitrification potentially favored by heterotrophic respiration (Mori et al. 2013). The lower soil resources in management intensity 3 along with a weak WFPS while still presenting strong N_2O fluxes (Fig. 2), could not be explained in this study. The preferential emission of urine by domestic animals at the sampling point could also have been a significant contributor to occasionally high N_2O fluxes by influencing the available N in soil (Allen et al. 2020). Literature also showed that mulching of grass could also explain higher N_2O fluxes in lawns, because grass clipping are made of N-rich organic matter that could be easily mineralized and integrated into denitrification process (Li et al. 2013; Nataningtyas et al. 2017). In our study, the restitution of grass clippings in lawns (management intensity 2) had no effect on N_2O fluxes (data not shown) as well as on total C, N and microbial biomass in soil.

Forests are not N_2O sources as they present aerobic conditions, low pH and N availability necessary for N_2O and N_2 produced during denitrification (Van Den Heuvel et al. 2011). Moreover, the strong C content (7.8%) in forest soils could play as a C-based sink for inorganic N, limiting N mineral availability and thus denitrification process (Qian and Follett 2012).

Carbon Footprint Simulation Of Ugs Types

Simulation of annual C fluxes as CO_2 (CH_4 being negligible in this study) in treed lawns were 838 g C.m⁻² lower than C fluxes in open lawns (Fig. 7). The city of Angers has 735 ha of lawns and 225 ha of urban forests. Based on proportion of treed and open lawns in the 15 UGS, we estimated the surface of open lawns as 75% (551 ha) and the surface of treed lawns as 25% (184 ha). Based on these proportions, urban forests in Angers could have emitted 2 555 Mg C, treed lawns 2 682 Mg C and open lawns 12 645 Mg C thus, for a total of 17 882 Mg C yearly. Whether 100% of the open lawns were converted into treed lawns then a loss of 4 614 Mg C as CO_2 would be avoided. The C losses avoided by the conversion of open lawns into treed lawns would be equivalent to the annual C footprint of 4 227 inhabitants of Angers, i.e. 2.8% of the population of Angers (Pôle métropolitain Loire Angers 2019). Furthermore, this estimate does not consider the C storage in the above-ground biomass of trees and would add up to this budget of C sequestration. Moreover, simulated data (Table 4) showed that the duration of soil C turnover would double in urban forests and treed lawns (plus 8 and 7 years respectively) compared to open lawns. Treed lawns are therefore interesting systems that should be further developed in future to contribute to climate-neutral cities. Nevertheless, the extension of treed lawns must be put into perspective with the technical and budgetary feasibility for managers (e.g. litterfall raking, tree breeding, C costs…) as well as the expectations of the population. The introduction of new methods of UGS management (no raking, less frequent mowing…) or introducing herbaceous species that are more resistant to shading will certainly be necessary to manage these new areas in a sustainable way. Water ecosystems such as river, lakes, and transient watered zones during winter should be included in studies (i.e. sources of CH_4 and N_2O) as they represent significant surface area in green cities such as in Angers.

We expected to find significantly higher C stocks in urban forests than in lawns due to the high restitution of recalcitrant aerial litter to the soil and the low net losses of C as CO_2 (Yesilonis and Pouyat 2012; Livesley et al. 2016). However, C stocks (0-30cm) did not show significant differences (Table 4). The C stock of urban forests (9 046 ± 2 575 g C m²), showed high variability between forest sites, which may be explained by an insufficient number of sampled sites and strong heterogeneity in UGS history management. The C stock of treed lawns 0–30 cm (10 590 ± 1 564 g C m²) also showed higher variability than those of open lawns (9 164 ± 808 g C m²). The woody vegetation species may have contributed to the high variability of C stocks in treed lawns that showed significant potential for C storage.

Table 4
Annual soil carbon turnover calculated from carbon stocks and annual soil respiration in urban forests (n = 3), treed lawns (n = 12) and open lawns (n = 12).

Measure	Urban forest	Treed lawn	Open lawn
C stocks 0-30cm (g C m⁻²)	9 045	10 590	9 164
Annual soil C from respiration (g C m⁻² yr⁻¹)	568	729	1 147
Soil C turnover (yr)	16	15	8

Conclusions

Temperature was the best predictor of soil respiration in all the studied UGS types (forests, treed lawns and open lawns). We showed that moisture reflected by the water filled pore space significantly added on variation explanation. However, we demonstrated that artificial conditions maintained by irrigation especially in summer made us to separate irrigated and non-irrigated data to improve modelling of soil respiration. The shading of trees strongly decreased soil respiration in treed lawns while soil properties were similar indicating a straightforward effect of lowering temperature. On the contrary, forests deeply changed soil properties as well as decreased soil temperature resulting in the lowest rates of soil respiration, making these systems the most conservative for C emissions, whereas above-ground biomass is an additional C sequestration pool. Open lawns, the most representative UGS type in term of surface area in Angers, was probably a source of CO_2 rather than a sink and converting them into treed lawns would result in a significant C preservation from soil respiration. Forests are strong sink for CH_4 throughout the year. On the contrary lawns, both open and treed were weak to mitigate CH_4 and can even be a source of CH_4 in irrigated parks where WFPS overpassed 75%. The rather weak N_2O emissions in this study probably reflected the transition already made from mineral to organic fertilization limiting N availability (microbial immobilization) whereas soil water logged was almost never met even in UGS with irrigation.

Further research is necessary to understand the soil contribution to GHG fluxes: We need to better take into consideration the C sequestration of trees as well as above and below ground litter and its turn-over and this should be done i) regarding the N cycle especially kinetic N availability along with GHG emissions; ii) by improving of the comprehensive contribution of living roots to soil respiration; and iii) by considering the role of soil microbial community and functions (catabolic and enzymatic) and should be measured under exotic plant species frequently met in UGSs.

Declarations

Competing interests

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

Authors' contributions

All authors contributed to the study conception and design. Preparation of the material was performed by TK, RG and VG. Data collection was performed by TK and RG. Data analysis was performed by TK and RG. The first draft of the manuscript was written by TK and all authors commented on previous versions of the manuscript. All authors read and approved the final manuscript.

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Availability of data and material

Datasets are available by requesting the authors.

The **ethics approval** declaration is not applicable to the current study.

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Figures

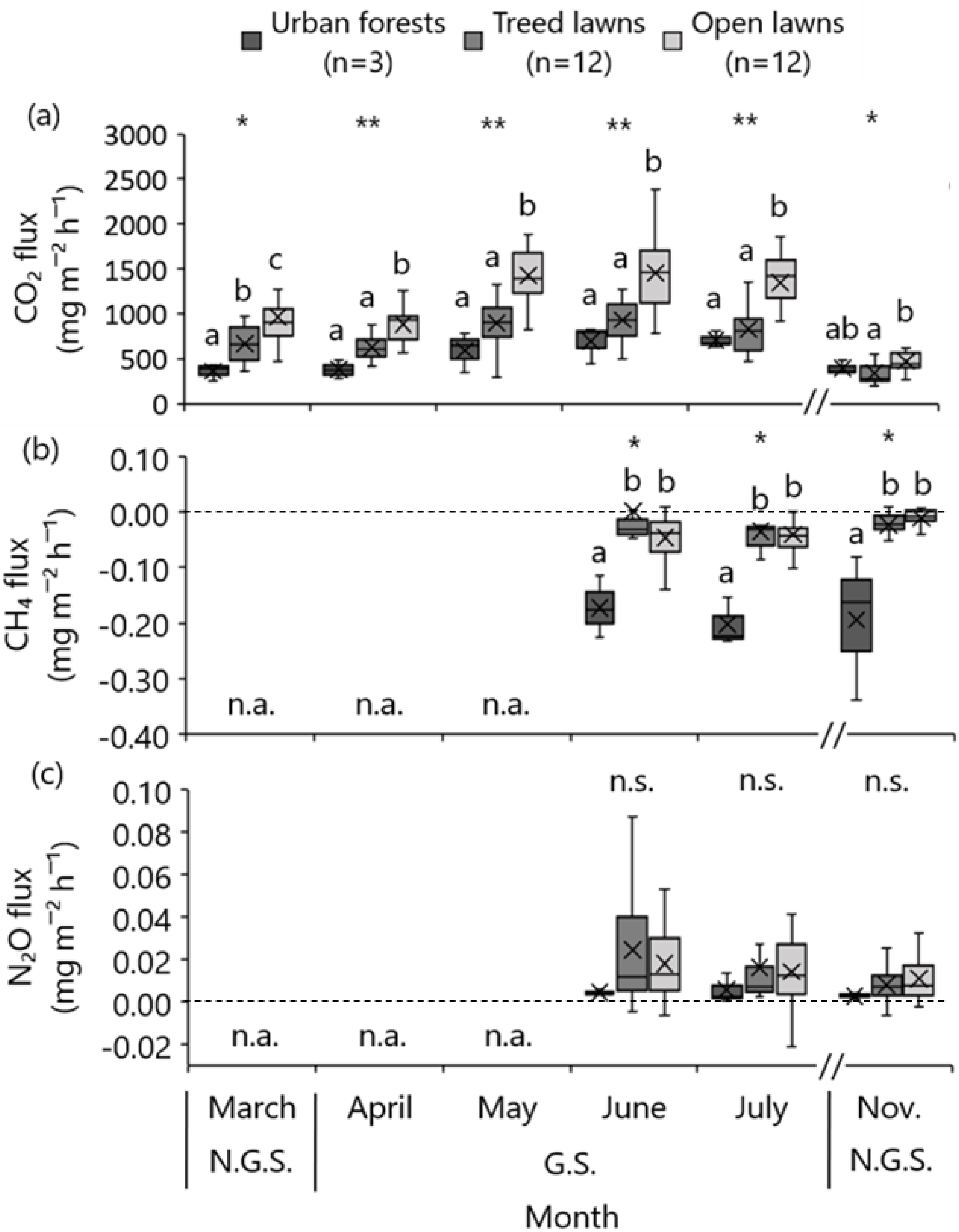

Figure 1

Monthly (a) CO , (b) CH and (c) N O fluxes (mg m⁻² h⁻¹) in urban forests, treed lawns and open lawns. GS and NGS indicate months during the growing season and non-growing season, respectively. In each box-plot the central bar of the graph is the median. The cross is the mean value and upper and lower edges are the quartiles. Letters indicate significant differences between treatments for a given month

(Tuckey post-hoc tests for CO fluxes and Mann-Whitney post-hoc tests for CH and N O fluxes); n.s., not significant; *P<0.05; **P<0.01; n.a., data not available

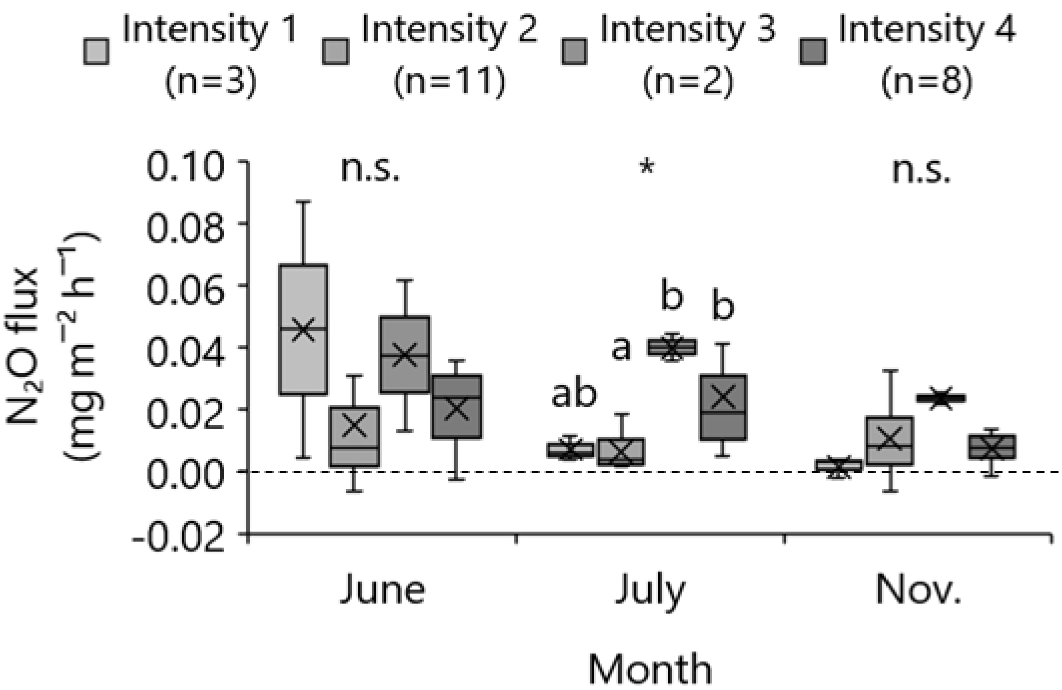

Figure 2

Monthly N O fluxes (mg m⁻² h⁻¹) for lawns distributed in 4 management intensities (Table 1). In each box-plot the central bar of the graph is the median. The cross is the mean value and upper and lower edges are the quartiles. Letters indicate significant differences between treatments (Mann-Whitney post-hoc tests); n.s., not significant; *P<0.05

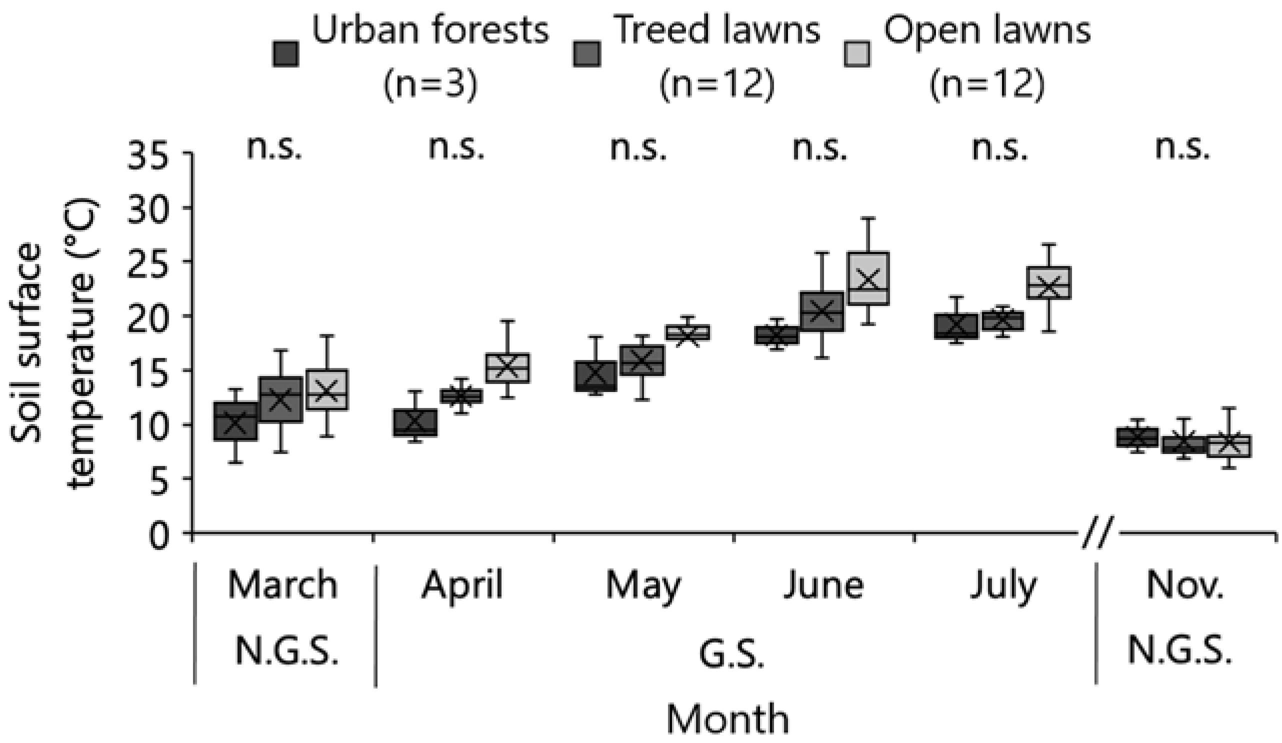

Figure 3

Monthly soil surface temperature (°C) in urban forests, treed lawns and open lawns. GS and NGS indicate months during growing season and non-growing season, respectively. In each box-plot the central bar of the graph is the median. The cross is the mean value and upper and lower edges are the quartiles. n.s., not significant

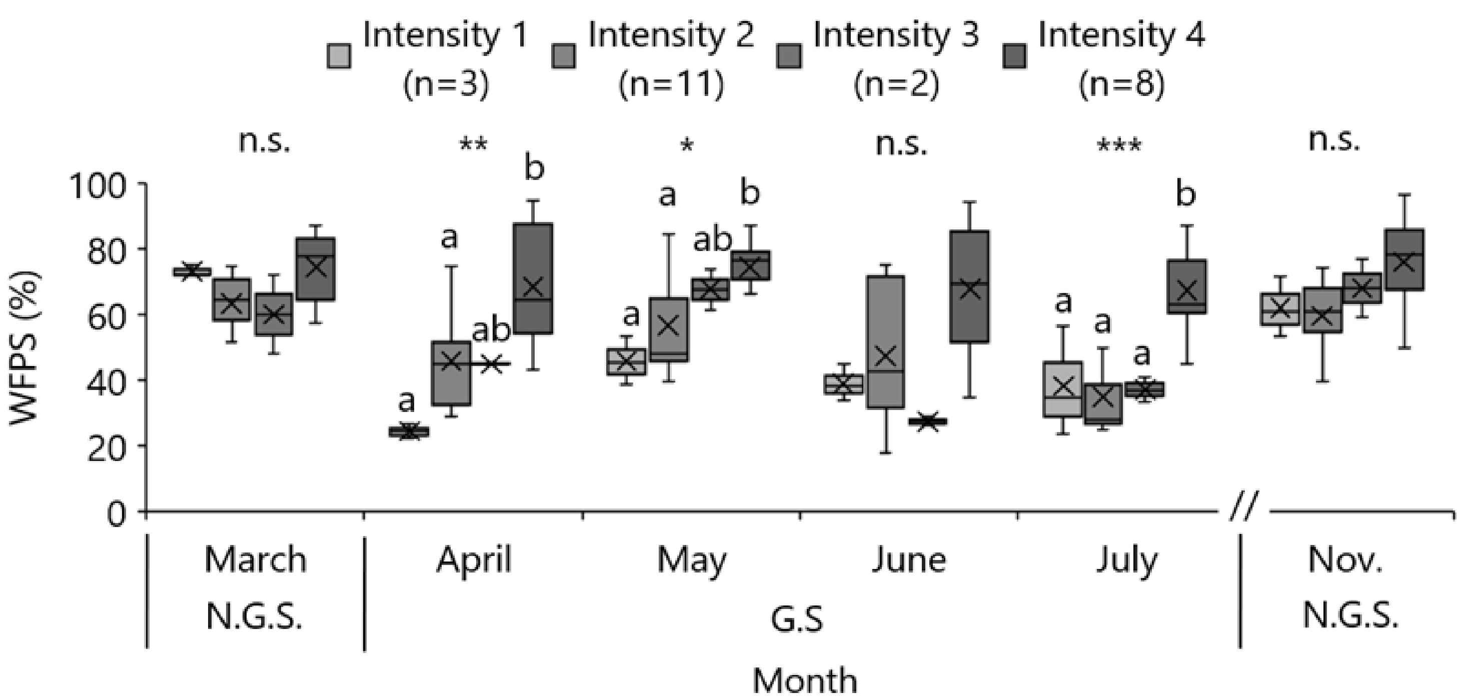

Figure 4

Monthly WFPS (%) for lawns distributed in 4 management intensities (modalities details are in Table 1). GS and NGS indicate months during growing season and non-growing season, respectively. In each box-plot the central bar of the graph is the median. The cross is the mean value and upper and lower edges are the quartiles. Letters indicate significant differences between treatments (Tuckey post-hoc tests for the 4 management intensities).; n.s., not significant; *P<0.05; **P<0.01; ***P<0.001

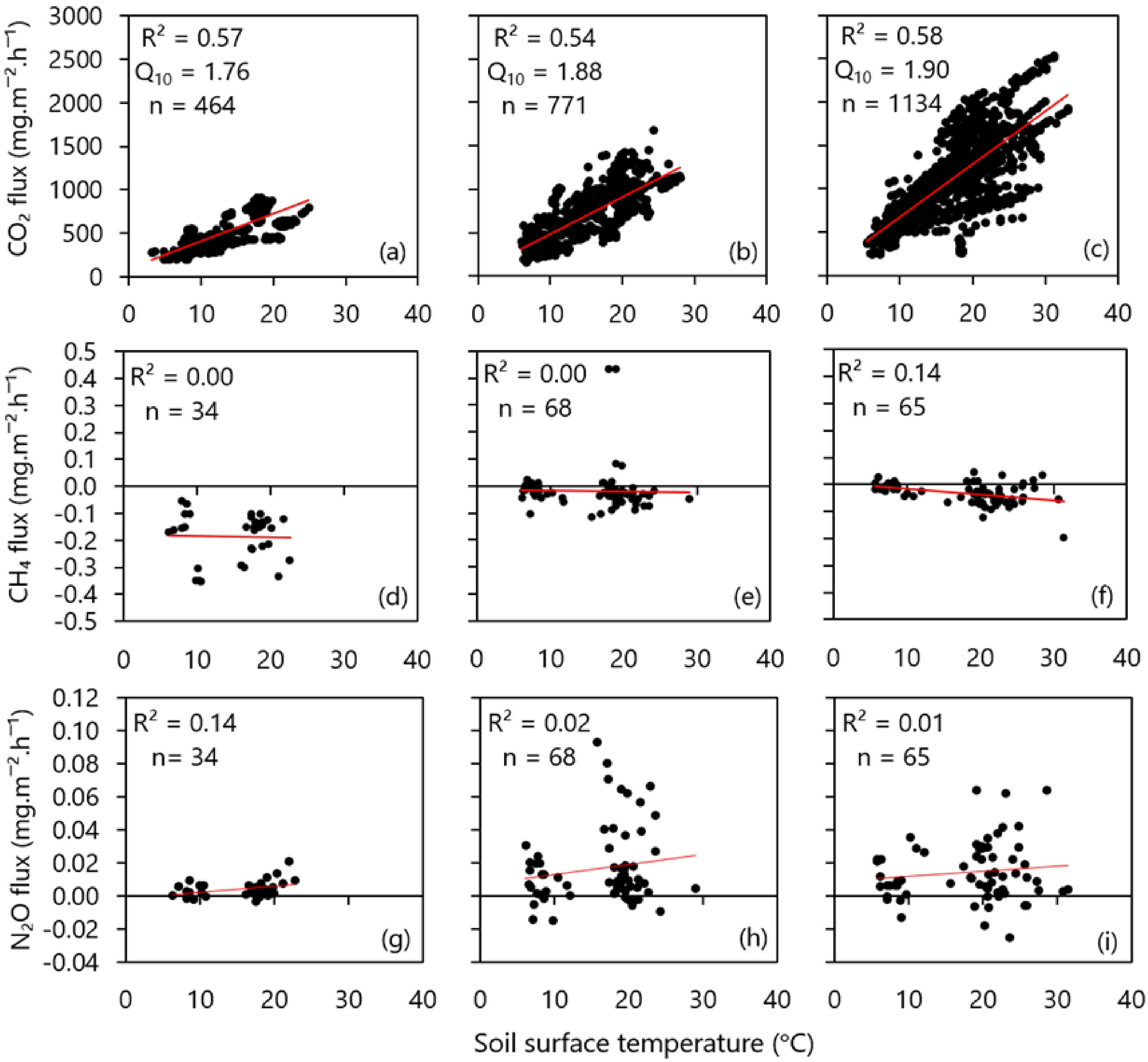

Figure 5

Linear regression of CO_2 (a, b, c), CH_4 (d, e, f) and N_2O (g, h, i) fluxes (mg m^{-2} h^{-1}) with soil surface temperature (°C) in urban forests (a, d, g), treed lawns (b, e, h) and open lawns (c, f, i). R^2 and Q_{10} refer to the coefficient of determination and sensitivity of CO_2 fluxes to temperature variation

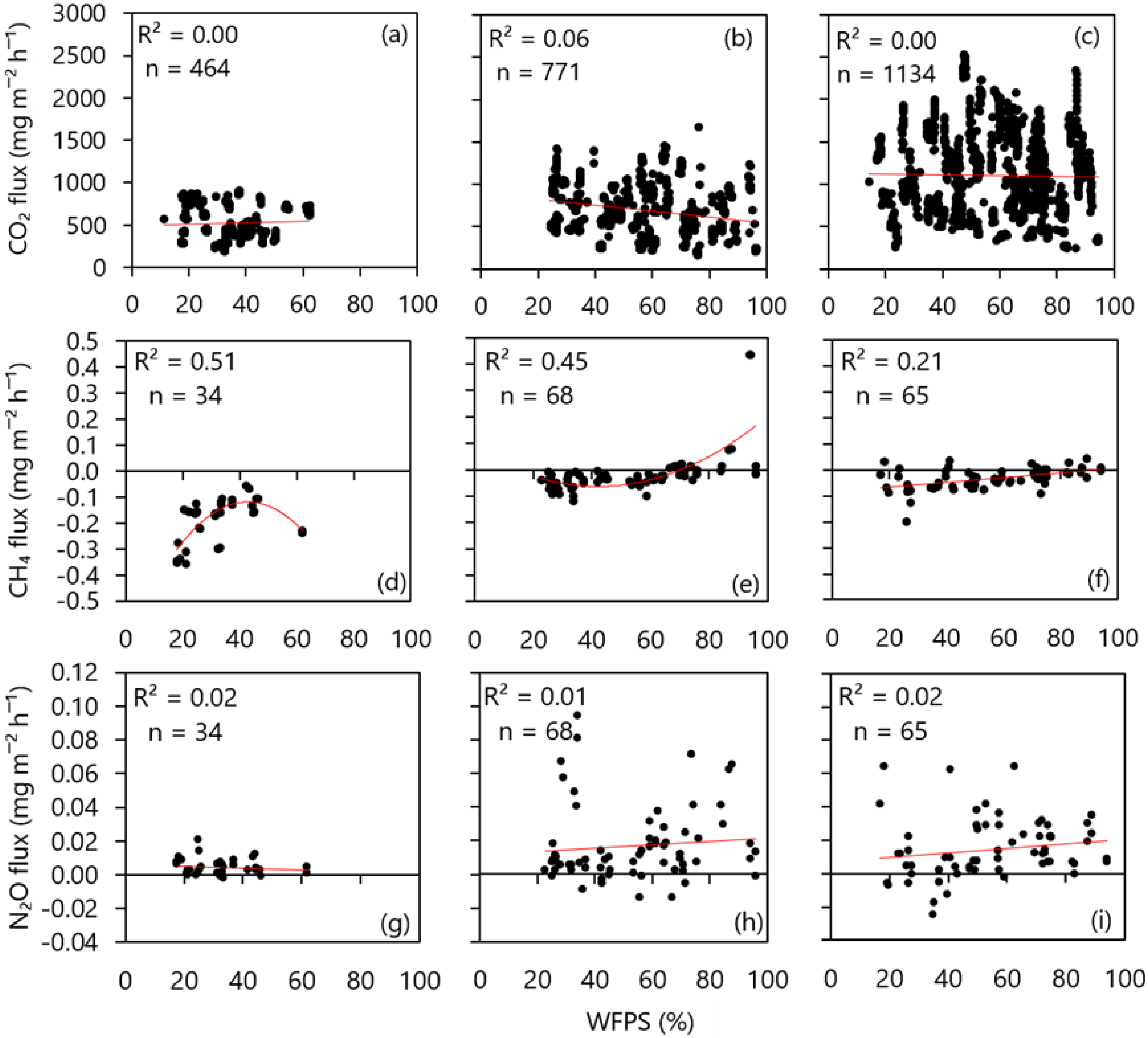

Figure 6

Linear or polynomial regressions of CO_2 (a, b, c), CH_4 (d, e, f) and N_2O (g, h, i) fluxes (mg m^{-2} h^{-1}) with WFPS (%) in urban forests (a, d, g), treed lawns (b, e, h) and open lawns (c, f, i). R^2 and Q_{10} refer to the coefficient of determination and sensitivity of CO_2 fluxes to temperature variation, respectively

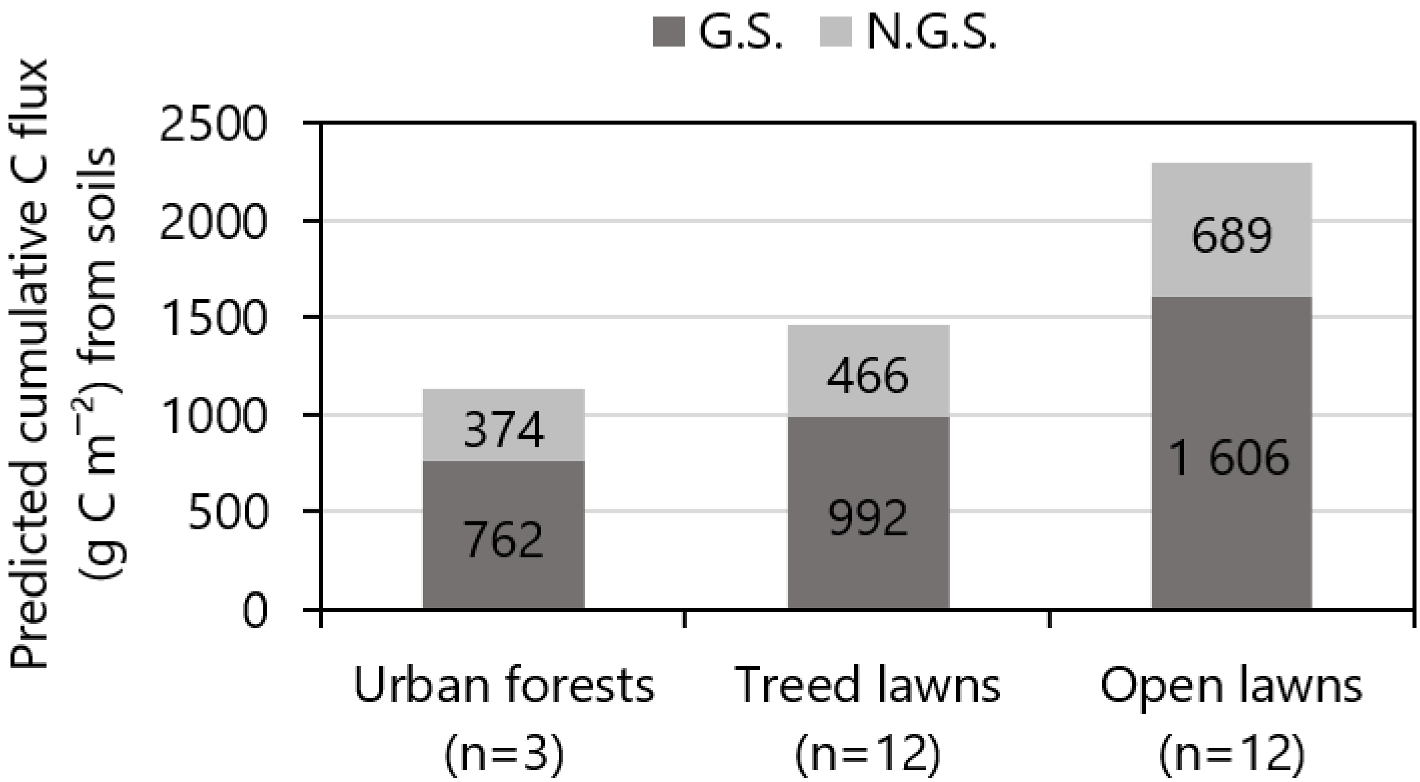

Figure 7

Predicted annual cumulative C flux (g C m^{-2}) from soils in urban forests (n=3), treed lawns (n=12) and open lawns (n=12). G.S.: growing season; N.G.S.: non growing season

Supplementary Files

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- SupplementaryInformationfinal.pdf