



1 **CO₂, CH₄, and N₂O Emissions from Urban Lawns in Southern**
2 **China: Characteristics and Driving Mechanisms**

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4 **Ziyan Wu^{1,2}, Shaohui Li^{1,3}, Zihan Fan^{1,3}, Fuhao Li^{1,3}, Cunxin Hu^{1,3}, Kunpeng Zang¹, Shuangxi**
5 **Fang^{1,4*}, Shuo Liu^{1,2,4*}**

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7 ¹ Zhejiang Carbon Neutral Innovation Institute & Zhejiang International Cooperation Base for
8 Science and Technology on Carbon Emission Reduction and Monitoring, Zhejiang University of
9 Technology, Hangzhou 310014, China

10 ² Shaoxing Research Institute, Zhejiang University of Technology, Shaoxing 312077, China

11 ³ College of Environment, Zhejiang University of Technology, Hangzhou 310014, China

12 ⁴ State Key Laboratory of Green Chemical Synthesis and Conversion, Zhejiang University of
13 Technology, Hangzhou 310014, China

14

15 **Correspondence authors:**

16 **Shuo Liu, E-mail: shuoliu241@zjut.edu.cn; Shuangxi Fang, E-mail: fangsx@zjut.edu.cn**



17 Abstract

18 Urban lawns have expanded rapidly with urbanization, substantially influencing
19 regional carbon budgets. However, research on their greenhouse gas (GHG) source-sink
20 dynamics remains scarce and the driving mechanisms are poorly understood. This study
21 investigated CO₂, CH₄, and N₂O fluxes from urban lawns beneath typical park-planted
22 broad-leaved (BF) and coniferous (CF) tree stands in a major city in the Yangtze River
23 Delta (YRD) region, China. The seasonal and diurnal patterns of GHG fluxes were
24 characterized, and the roles of environmental factors and mowing management were
25 analyzed. Results showed that CO₂ fluxes from both lawn types exhibited pronounced
26 seasonal fluctuations, with higher emissions in summer and lower in winter. The mean
27 annual CO₂ flux from CF lawns ($217.74 \pm 15.58 \text{ mg C m}^{-2} \text{ h}^{-1}$) was significantly higher
28 than that from BF lawns ($94.64 \pm 10.12 \text{ mg C m}^{-2} \text{ h}^{-1}$). Diurnally, CO₂ fluxes displayed a
29 "daytime low, nighttime high" pattern, contrasting with the typical "daytime high,
30 nighttime low" unimodal pattern commonly observed in grassland soils. Annual mean CH₄
31 fluxes for the BF and CF lawns were $-20.29 \pm 1.22 \text{ ug C m}^{-2} \text{ h}^{-1}$ and $-25.39 \pm 1.59 \text{ ug C}$
32 $\text{m}^{-2} \text{ h}^{-1}$, respectively, indicating net CH₄ uptake throughout the year, with soil volumetric
33 water content (VWC) emerging as the key common regulatory factor. Diurnal CH₄
34 variations followed a "daytime high, nighttime low" pattern in spring and summer but
35 showed no clear diurnal rhythm in autumn and winter. Annual mean N₂O fluxes were 8.11
36 $\pm 0.75 \text{ ug N m}^{-2} \text{ h}^{-1}$ and $6.58 \pm 0.85 \text{ ug N m}^{-2} \text{ h}^{-1}$ for the BF and CF lawns, respectively,
37 with higher emissions in spring-summer and lower in autumn-winter, indicating net N₂O
38 emission overall. N₂O also deviated from the typical diurnal pattern, likely due to
39 pronounced fluctuations in soil water content. Mowing management showed that spring
40 mowing caused the least disturbance to GHG fluxes, summer mowing significantly
41 enhanced CO₂ emissions, and autumn mowing promoted CH₄ uptake and N₂O emission.
42 By combining structural equation modeling (SEM), random forest (RF) importance
43 ranking, and SHAP analysis, we identified fundamentally distinct environmental driver
44 regimes between the two canopy types: the BF lawn exhibited a "temperature–inorganic
45 nitrogen dominated" pattern, whereas the CF lawn showed a "moisture–microbial–carbon
46 pool co-regulation" pattern. Overall, urban lawns exhibit a unique GHG source–sink
47 pattern and regulatory regimes distinct from those of natural grasslands. Quantifying their



48 GHG budgets is of considerable scientific value for optimizing urban carbon management
49 and advancing low-carbon city development.
50 **Keywords:** urban lawn, greenhouse gas flux, mowing, driving mechanism



1. Introduction

Anthropogenic greenhouse gas (GHG) emissions are the principal driver of global warming (IPCC, 2023). Carbon dioxide (CO₂), methane (CH₄), and nitrous oxide (N₂O) represent the three most important GHGs, accounting for approximately 66%, 16%, and 6% of the global greenhouse effect, respectively (WMO, 2025). Although CH₄ and N₂O are present at relatively low atmospheric concentrations, their warming potentials are substantially higher. Over a 100-year time horizon, their global warming potentials (GWP) are 28 and 265 times that of CO₂ (IPCC, 2023). Studies indicate that more than 60% of GHG emissions originate from urban areas (Han et al., 2018). Urban lawns, as integral components of urban ecosystems (Aronson et al., 2017), serve both as important reservoirs of carbon and nitrogen and as non-negligible sources of GHG emissions (Trémeau et al., 2024). Their source–sink dynamics thus exert a significant influence on regional and even global climate change.

Urban lawn areas have expanded steadily alongside urbanization over recent decades. In China, the national urban green space area had reached $3.31 \times 10^{10} \text{ m}^2$ by 2024, with a built-up area greening coverage rate of 43.2%, representing an increase of nearly 40% compared to 2015 (NBSC, 2024). This expansion not only alters urban surface characteristics but also affects gas exchange processes within urban ecosystems (Ng et al., 2015). On the one hand, lawn plants sequester CO₂ through photosynthesis, mitigating urban heat islands and reducing atmospheric GHG concentrations (Tolamariam, 2023). On the other hand, management practices such as mowing, trampling, and irrigation can alter GHG fluxes from urban lawns (Gu et al., 2015; Pengthamkeerati et al., 2005), making their net role as either sources or sinks inherently complex.

Nevertheless, research focusing on urban lawn ecosystems remains comparatively scarce, while the vast majority of grassland-targeted GHG studies have concentrated on natural ecosystems. For natural-system studies, Wang et al. (2024b) reported that CO₂ emissions from alpine grasslands are primarily controlled by soil water content, nutrient availability, pH, and fungal diversity, whereas CH₄ uptake is governed by methanotroph community richness, and N₂O emissions are negatively correlated with soil pH. Zou et al. (2025) demonstrated that drought followed by rewetting significantly increases CO₂ and N₂O emissions in grasslands, and that soil moisture exerts a dual influence on CH₄ flux:



increased moisture promotes CH₄ emission under high initial water content but suppresses it under low initial water content. Ma et al. (2021) found that CH₄ uptake rates in northern temperate grasslands are negatively correlated with soil water content and bulk density, while the influence of soil organic carbon, texture, and bulk density on CO₂ flux is relatively minor compared with that of soil water content. Beyond abiotic factors, anthropogenic disturbances further modulate GHG fluxes. Heavy grazing increases CO₂ and positively correlates with CH₄ and N₂O emissions (Cao et al., 2025) Yu et al. (2025) examined the effects of mowing intensity on GHGs in alpine meadows, finding that light mowing increased soil CO₂ emissions, heavy mowing decreased them, and soil CH₄ uptake declined with increasing mowing intensity.

Existing studies on GHG emissions from urban lawns have largely concentrated on soil organic carbon stocks, above- and below-ground biomass carbon pools, and overall carbon sink potential. For example, Wang et al. (2024a) reported that the topsoil (0–30 cm) carbon stock of lawns is approximately twice that of trees, yet even if all urban green spaces were converted to forestland, only 1% of annual emissions could be offset. However, these studies have predominantly emphasized carbon sequestration, with few simultaneously monitoring N₂O and CH₄ fluxes. Guo et al. (2024) showed that moderate management (e.g., irrigation and fertilization) can enhance lawn carbon stocks, though soil organic carbon in lawns remains significantly lower than in other green space types. Trémeau et al. (2024) conducted a comparative study of irrigated lawns, non-irrigated lawns, and meadows, but their primary objective was to assess biodiversity and drought resilience, with GHG fluxes treated as secondary variables. Consequently, the GHG source–sink dynamics of urban lawns currently lack robust in situ data, and the mechanisms governing their multi-gas fluxes are yet to be resolved.

As a major global emitter of GHGs, China has placed increasing emphasis in recent years on enhancing carbon sinks and reducing emissions in urban green spaces. In 2025, the National Forestry and Grassland Administration (NFGA) launched the "Beijing Sub-center Landscape Green Space Carbon Credit Pilot," which explicitly incorporates lawns into the carbon sink accounting of tree–shrub–grass–wetland composite ecosystems, thereby monetizing ecological benefits through carbon trading to support low-carbon maintenance practices (NFGA, 2025). The effectiveness of such policy initiatives, however,



hinges critically on a robust scientific understanding of the GHG source–sink mechanisms of urban lawns. To address this gap, we conducted in situ monitoring of CO₂, CH₄, and N₂O fluxes using the static chamber–gas chromatography method in urban lawns beneath BF and CF in a representative southern Chinese city. Coupled with environmental factor analyses, we explored seasonal and diurnal emission dynamics and the mechanisms driving GHG flux variations. The findings aim to provide a scientific basis for evaluating the carbon sink function of urban lawns, informing evidence-based urban greening strategies, and advancing the development of low-carbon cities.

2. Materials and Methods

2.1 Study Area

The experimental site was located in Hangzhou, Zhejiang Province, China (120.17°E, 30.29°N, 10 m a.s.l.) (Fig. 1), a major city in the Yangtze River Delta (YRD) region characterized by intensive urbanization and dense economic activity. The experimental plots are park-planted tree-lawn communities within an urban park, comprising two stand types: broad-leaved (BF) stands dominated by *Ginkgo biloba* and *Osmanthus fragrans*, and coniferous (CF) stands dominated by *Metasequoia glyptostroboides*. These tree stands were established for urban landscaping during the 1980s–1990s, with a current stand age of ~40 years. The CF stand was planted with a tree density of 220–280 stems·hm⁻² (row spacing of 4–5 m) and the annual tree litter input was 2.2–3.4 t·hm⁻²·yr⁻¹. The BF stand was planted with a relatively lower tree density of 180–240 stems·hm⁻² and the annual tree litter input was 2.8–3.6 t·hm⁻²·yr⁻¹. The soil type (0–10 cm) of the two stands is loam with balanced sand. The understory lawn was regularly mowed, with nearly all of grass clippings removed. No artificial fertilization or supplementary irrigation was applied to either trees or lawn vegetation throughout the measurement period. The area features a typical subtropical monsoon climate, with four distinct seasons and concurrent rainfall and high temperatures. Winters are mild with low precipitation, while summers are hot and rainy. The mean annual temperature is 22.3 °C, the hottest month being August (monthly mean 33.1 °C) and the coldest February (monthly mean 5.2 °C). Annual precipitation reaches 1500.6 mm, with rainfall concentrated from June to September. The BF lawn was dominated by *Brachythecium* spp., *Carex* spp., *Digitaria sanguinalis*, and *Oxalis*



144 *corniculata*, whereas the CF lawn is dominated by *Persicaria virginiana*, *Oxalis*
 145 *corniculata*, *Clinopodium chinense*, and *Rostellularia procumbens*.

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147 **2.2 Gas Sampling and Analysis**

148 The experimental plots comprised two urban lawn types, i.e., lawns beneath BF and
 149 CF. Three independent randomized replicates were established for each lawn type, yielding
 150 a total of six observation sites. Gas sampling and analysis were conducted using the static
 151 chamber–gas chromatography method. Each flux chamber had a height of 60 cm and a
 152 base area of 40 cm × 40 cm. The outer surfaces were covered with reflective aluminum foil
 153 to minimize the greenhouse effect during enclosure. Two fans powered by an external 12
 154 V battery were mounted on the inner ceiling to ensure thorough gas mixing, and a
 155 thermometer was installed at the top to monitor internal temperature changes.

156 The experimental period was from May 2022 to May 2023. One month prior to the
 157 first sampling campaign, chamber base frames were inserted into the soil to a depth of 10
 158 cm and allowed to stabilize. During each sampling event, the chamber body was vertically
 159 placed into the water-filled groove of the base frame and sealed with water to isolate the
 160 chamber interior from the ambient atmosphere. The fans were then activated to
 161 homogenize the air inside. Gas samples (1 L each) were collected into aluminum foil
 162 sampling bags at 0, 5, 10, and 15 min after chamber closure. Seasonal sampling was
 163 conducted between 9:00 and 11:00 local time, twice per week under clear weather
 164 conditions. Diurnal sampling was performed at 3-hour intervals from 9:00 on the sampling
 165 day to 9:00 the following day (eight sampling periods per cycle), with three replicate
 166 diurnal cycles per season (spring, summer, autumn, and winter). Gas samples were
 167 transported to the laboratory within 24 h and analyzed on an Agilent 7890B gas
 168 chromatograph (Agilent, Santa Clara, CA, USA; three-channel system; Hong et al., 2022),
 169 with CO₂ and CH₄ measured by FID (CO₂ after conversion to CH₄) and N₂O by μ -ECD.
 170 Against WMO/GAW-traceable standards, the precision (RSD), accuracy, and linearity (R²)
 171 were: CO₂ 0.05% (1 σ = 0.20 ppm), 0.37 ppm, 0.9999; CH₄ 0.08% (1 σ = 1.70 ppb), 0.15
 172 ppb, 0.9999; N₂O 0.08% (1 σ = 0.26 ppb), 0.35 ppb, 0.9999. The repeatability (1 σ) was 0.20
 173 ppm, 1.70 ppb, and 0.26 ppb, respectively, corresponding to minimum detectable changes
 174 ($\approx$ 3 σ) of $\sim$ 0.6 ppm, $\sim$ 5 ppb, and $\sim$ 0.8 ppb. During 15-min chamber closures, headspace



175 concentrations ranged within 437–729 ppm CO₂ (median 519 ppm), 2.04–2.47 ppm CH₄
176 (2036–2468 ppb), and 0.325–0.350 ppm N₂O (325–350 ppb). Each individual flux
177 measurement was calculated as the mean of three replicate samples. All reported mean
178 fluxes are expressed as mean ± standard deviation (SD).

179 Meteorological variables, including atmospheric temperature (AT), relative humidity
180 (RH), and precipitation (PI), were recorded synchronously with each gas sampling event.
181 Soil samples were collected monthly using a five-point sampling method: within a 50-cm
182 radius of each observation site, topsoil (0–10 cm) was collected and homogenized to form
183 a representative composite sample. Soil samples were passed through a 2-mm sieve prior
184 to analyses. Microbial biomass carbon (MBC) and nitrogen (MBN) were determined via
185 chloroform fumigation–extraction followed by TOC analysis and Kjeldahl digestion,
186 respectively. Soil pH was measured by the potentiometric method. Nitrate nitrogen (NO₃⁻-
187 N) was determined by ultraviolet spectrophotometry, and ammonium nitrogen (NH₄⁺-N)
188 by the indophenol blue colorimetric method. Total carbon (TC) and total nitrogen (TN)
189 were quantified using an elemental analyzer, from which the carbon-to-nitrogen ratio (C/N)
190 was calculated. Soil organic carbon (SOC) was determined by the potassium dichromate
191 oxidation–spectrophotometric method. Soil temperature (ST) and volumetric water content
192 (VWC) were monitored *in situ* using a portable soil temperature and moisture meter (TDR-
193 315H, Acclima).

194 **2.3 Data Processing**

195 All data analyses and graphical visualizations were performed using R software
196 (version 4.3.0). Two-way analysis of variance (ANOVA) was conducted to assess the
197 significance of differences in GHG fluxes between different sites across months or seasons.
198 Pearson correlation analysis was used to examine the relationships between GHG fluxes
199 and environmental variables. To further elucidate the integrated effects of multiple
200 environmental factors on GHG fluxes, we employed Mantel tests and structural equation
201 modeling (SEM). These approaches allowed us to disentangle the direct and indirect
202 influence pathways between environmental drivers and GHG fluxes, and to identify the
203 key regulatory factors and underlying processes governing flux variations. To
204 cross-validate the robustness of the SEM results in terms of predictive accuracy, we
205 performed random forest (RF) importance ranking for 13 environmental factors and



206 applied SHAP (SHapley Additive exPlanations) based on the XGBoost algorithm. SHAP
207 summary plots were used to visualize the direction and nonlinearity of each factor's effects,
208 thereby achieving complementary validation between causal inference (SEM) and
209 predictive modeling (RF + SHAP).

210

211 **3. Results and Analysis**

212 **3.1 Seasonal Variation Characteristics of GHG Fluxes from Urban Lawns**

213 Figure 2 presents the long-term dynamics of environmental factors and GHG fluxes
214 from lawns beneath urban park-planted BF and CF tree stands. During the observation
215 period, CO₂ fluxes from both lawn types exhibited pronounced seasonal fluctuations, with
216 generally consistent temporal trends (Fig. 2c). Monthly mean CO₂ fluxes followed an
217 n-shaped trajectory, peaking in July and subsequently declining (Fig. S1). For both BF and
218 CF lawns, mean CO₂ fluxes ranked summer > autumn > spring > winter (Table 1); summer
219 fluxes were significantly higher than the other seasons in both lawns. The maximum CO₂
220 flux in the BF lawn ($258.38 \pm 15.68 \text{ mg C m}^{-2} \text{ h}^{-1}$) occurred on 8 July, while the minimum
221 ($-30.02 \pm 8.68 \text{ mg C m}^{-2} \text{ h}^{-1}$) occurred on 21 May. In the CF lawn, the emission peak
222 ($464.84 \pm 22.18 \text{ mg C m}^{-2} \text{ h}^{-1}$) was recorded on 29 July, and the lowest flux (63.81 ± 20.75
223 $\text{mg C m}^{-2} \text{ h}^{-1}$) was observed on 6 December. The period from April to September exhibited
224 the most pronounced intra-annual fluctuations in CO₂ flux, with amplitudes of 288.40 and
225 359.75 $\text{mg C m}^{-2} \text{ h}^{-1}$ for the BF and CF lawns, respectively. Fluctuations during the
226 remaining months were smaller, with amplitudes of 132.37 and 117.77 $\text{mg C m}^{-2} \text{ h}^{-1}$.

227 Seasonal fluctuations in CH₄ fluxes were less pronounced for both lawn types, with
228 both systems acting predominantly as CH₄ sinks throughout the year (Fig. 2d). Mean CH₄
229 uptake in the BF lawn ranked as autumn > winter > summer > spring, with no significant
230 inter-seasonal differences (Table 1). In the CF lawn, uptake ranked as autumn > winter >
231 spring > summer (Table 1), with a significant difference detected only between summer
232 and autumn. The maximum CH₄ uptake peak in the BF lawn occurred on 6 November
233 ($-41.45 \pm 5.00 \text{ ug C m}^{-2} \text{ h}^{-1}$), whereas the CF lawn exhibited its emission peak on 25 July
234 ($5.33 \pm 3.37 \text{ ug C m}^{-2} \text{ h}^{-1}$) and its maximum uptake ($-45.85 \pm 4.32 \text{ ug C m}^{-2} \text{ h}^{-1}$) on 18
235 April. Monthly mean CH₄ fluxes for both lawn types were more negative in November–
236 December than in other months (Fig. S1).



237 N₂O fluxes from both lawn types exhibited pronounced seasonal fluctuations, with
238 higher values in spring–summer and lower values in autumn–winter. Overall, both systems
239 acted as net N₂O sources throughout the observation period (Fig. 2e). Mean N₂O fluxes
240 ranked as summer > spring > autumn > winter for both lawn types (Table 1), with
241 significant differences observed between autumn–winter and summer. In the BF lawn, the
242 maximum N₂O flux ($26.11 \pm 0.14 \text{ ug N m}^{-2} \text{ h}^{-1}$) appeared on 18 May and the minimum
243 ($-0.19 \pm 0.40 \text{ ug N m}^{-2} \text{ h}^{-1}$) on 6 November. For the CF lawn, the emission peak ($33.79 \pm$
244 $4.06 \text{ ug N m}^{-2} \text{ h}^{-1}$) occurred on 14 July and the minimum ($-2.40 \pm 0.39 \text{ ug N m}^{-2} \text{ h}^{-1}$) on 2
245 November. Monthly mean N₂O fluxes for both lawn types began to decline in July (Fig.
246 S1).

247 **3.2 Diurnal Variation Characteristics of GHG Fluxes from Urban Lawns**

248 Figure 3 illustrates the diurnal dynamics of GHG fluxes from lawns beneath urban
249 park-planted BF and CF tree stands. For CO₂ fluxes, the BF lawn exhibited greater
250 variability during daytime than at night (Fig. 3a), whereas the opposite pattern was
251 observed for the CF lawn (Fig. 3b). Both lawns displayed large diurnal amplitudes in spring
252 and summer, with much smaller amplitudes in autumn and winter; the BF lawn fluctuated
253 more than the CF lawn during the latter two seasons. In spring, the BF lawn showed a V-
254 shaped curve, with CO₂ flux declining during the daytime to a minimum at 15:00 ($35.83 \pm$
255 $5.88 \text{ mg C m}^{-2} \text{ h}^{-1}$) and then rising sharply to a peak at 21:00 ($287.23 \pm 47.40 \text{ mg C m}^{-2}$
256 h^{-1}), yielding an amplitude of $251.40 \text{ mg C m}^{-2} \text{ h}^{-1}$. In contrast, the CF lawn exhibited an
257 M-shaped bimodal curve, with peaks at 03:00 ($356.22 \pm 44.08 \text{ mg C m}^{-2} \text{ h}^{-1}$) and 18:00
258 ($318.24 \pm 65.26 \text{ mg C m}^{-2} \text{ h}^{-1}$) and a trough at 09:00 ($217.22 \pm 22.48 \text{ mg C m}^{-2} \text{ h}^{-1}$), giving
259 an amplitude of $196.88 \text{ mg C m}^{-2} \text{ h}^{-1}$. In summer, the two lawn types showed broadly
260 similar trends, both characterized by higher nighttime and lower daytime fluxes. Peaks
261 occurred at 03:00 ($273.18 \pm 20.35 \text{ mg C m}^{-2} \text{ h}^{-1}$ for BF; $387.29 \pm 24.81 \text{ mg C m}^{-2} \text{ h}^{-1}$ for
262 CF), trough at 12:00 for BF ($153.41 \pm 16.62 \text{ mg C m}^{-2} \text{ h}^{-1}$) and at 18:00 for CF ($272.24 \pm$
263 $25.03 \text{ mg C m}^{-2} \text{ h}^{-1}$); the amplitudes were 119.77 and 115.05 $\text{mg C m}^{-2} \text{ h}^{-1}$, respectively.
264 In autumn and winter, the diurnal amplitude diminished substantially for both types. The
265 BF lawn displayed a V-shaped curve with both peak and trough occurring during daytime,
266 whereas the CF lawn exhibited near-constant fluxes ("flat-line" pattern).



267 Diurnal CH₄ flux amplitudes were likewise greater in spring–summer than in autumn–
268 winter for both lawn types. In spring, the CF lawn exhibited a stronger CH₄ uptake capacity
269 than the BF lawn, whereas the opposite pattern was observed in summer; no significant
270 difference was detected between the two types in autumn or winter. In the BF lawn,
271 daytime variability exceeded nighttime variability during spring and summer (Fig. 3c),
272 whereas in the CF lawn, nighttime variability was greater than daytime variability except
273 during summer and autumn (Fig. 3d). In spring, the BF lawn exhibited transient CH₄
274 emission around midday (12:00–15:00), with a maximum emission peak at 12:00 ($6.19 \pm$
275 $6.21 \text{ ug C m}^{-2} \text{ h}^{-1}$) and a maximum uptake peak at 18:00 ($-28.60 \pm 5.10 \text{ ug C m}^{-2} \text{ h}^{-1}$),
276 resulting in a diurnal amplitude of $34.79 \text{ ug C m}^{-2} \text{ h}^{-1}$. The CF lawn reached its maximum
277 uptake at 06:00 ($-45.90 \pm 3.12 \text{ ug C m}^{-2} \text{ h}^{-1}$) and its minimum uptake at 21:00 ($-24.75 \pm$
278 $2.07 \text{ ug C m}^{-2} \text{ h}^{-1}$), with an amplitude of $21.15 \text{ ug C m}^{-2} \text{ h}^{-1}$. In summer, the BF lawn
279 showed an inverted-U curve, with the maximum uptake at 18:00 ($-68.14 \pm 6.07 \text{ ug C m}^{-2}$
280 h^{-1}) and an amplitude of $38.02 \text{ ug C m}^{-2} \text{ h}^{-1}$. In contrast, the CF lawn exhibited an M-
281 shaped bimodal pattern, with peaks at 09:00 ($-17.84 \pm 5.50 \text{ ug C m}^{-2} \text{ h}^{-1}$) and 15:00
282 ($-18.03 \pm 5.23 \text{ ug C m}^{-2} \text{ h}^{-1}$), and an amplitude of $32.71 \text{ ug C m}^{-2} \text{ h}^{-1}$. In autumn and winter,
283 diurnal variations were modest for both lawn types, and seasonal mean daily fluxes did not
284 differ significantly between types.

285 Diurnal N₂O flux variation exhibited the smallest inter-seasonal differences among the
286 three gases. In spring and winter, the N₂O uptake capacity of the BF lawn exceeded that of
287 the CF lawn, whereas the opposite pattern was observed in summer; no significant
288 difference was detected between the two types in autumn. In the BF lawn, daytime
289 variability was greater than nighttime variability in spring, but the reverse was true in
290 summer (Fig. 3e), in contrast to the spring–summer pattern observed in the CF lawn (Fig.
291 3f). In spring, the BF lawn acted primarily as an N₂O sink, with weak emission occurring
292 between 06:00 and 09:00 (maximum emission peak $14.18 \pm 13.62 \text{ ug N m}^{-2} \text{ h}^{-1}$ at 06:00)
293 and an amplitude of $5.36 \text{ ug N m}^{-2} \text{ h}^{-1}$. In contrast, the CF lawn functioned mostly as an
294 N₂O source, reaching its maximum emission at 03:00 ($20.21 \pm 12.75 \text{ ug N m}^{-2} \text{ h}^{-1}$) and a
295 secondary peak at 18:00 ($6.52 \pm 3.96 \text{ ug N m}^{-2} \text{ h}^{-1}$), with an amplitude of $19.98 \text{ ug N m}^{-2}$
296 h^{-1} . In summer, the two lawn types followed broadly similar diurnal trends, both peaking
297 at 03:00. However, the BF lawn exhibited an uptake peak at 00:00, whereas the CF lawn



showed its uptake peak at 18:00, with amplitudes of 49.87 and 21.48 $\mu\text{g N m}^{-2} \text{ h}^{-1}$, respectively. In autumn, both types displayed a relatively stable diurnal emission pattern, with amplitudes close to zero. In winter, the BF lawn exhibited a U-shaped diurnal curve with turning points at 06:00 and 18:00, and an amplitude of 54.86 $\mu\text{g N m}^{-2} \text{ h}^{-1}$, obviously greater than the nearly flat pattern observed for the CF lawn (amplitude 8.28 $\mu\text{g N m}^{-2} \text{ h}^{-1}$).

3.3 Effects of Environmental Factors and Mowing Management on GHG Fluxes

3.3.1 Effects of Environmental Factors on GHG Fluxes

Pearson correlation analyses between GHG fluxes and environmental factors for the BF and CF lawns are shown in Figure 4. For both lawn types, ST and AT were highly significantly positively correlated with CO_2 flux ($r > 0.80$, $p < 0.001$), and VWC was significantly positively correlated with CH_4 flux ($r = 0.50$ and 0.43 , respectively, $p < 0.01$). The environmental correlates of N_2O flux differed most markedly between the two canopy types: in the BF lawn, N_2O flux was primarily associated with AT, ST, and $\text{NH}_4^+\text{-N}$, whereas in the CF lawn, it was associated only with PI and MBN. SEM further quantified the regulatory pathways for each lawn type (Fig. 5). In the BF lawn, CO_2 flux was driven primarily by direct positive effects of AT ($\beta = 0.79$), ST ($\beta = 0.56$), and $\text{NH}_4^+\text{-N}$ ($\beta = 0.42$), while PI, RH, and VWC exerted indirect effects through $\text{NH}_4^+\text{-N}$ ($\beta = -0.12$, 0.19 , and 0.18 , respectively). CH_4 flux was jointly suppressed by direct negative effects of ST ($\beta = -0.98$), TC ($\beta = -0.51$), and MBC ($\beta = -0.60$), with VWC showing a direct positive effect ($\beta = 0.49$). N_2O flux was indirectly and positively driven by ST through SOC and $\text{NH}_4^+\text{-N}$ (total effect = 0.86), whereas SOC ($\beta = -0.72$) and $\text{NH}_4^+\text{-N}$ ($\beta = -0.73$) exerted direct negative effects on N_2O . The CF lawn exhibited a distinctly different pathway structure. CO_2 flux was predominantly driven by a direct negative effect of MBC ($\beta = -0.86$), with ST and PI exerting indirect influences through MBC ($\beta = 0.59$ and 0.33 , respectively). CH_4 flux was jointly regulated by negative direct effects of RH (total effect = -0.72), C/N ($\beta = -0.52$), and SOC ($\beta = -0.70$), while VWC showed a positive effect ($\beta = 0.43$). N_2O flux was positively driven by PI ($\beta = 0.48$) and MBN ($\beta = 0.75$) and negatively regulated by C/N ($\beta = -0.67$).

To cross-validate the robustness of the above causal pathways, we further employed RF modeling (Fig. 6) and SHAP interpretable analysis (Fig. 7) to complement the Mantel test and SEM results. The RF importance ranking was highly consistent with SEM in terms



of key predictors. In the BF lawn, CO₂ flux was predicted primarily by TN, AT, and ST as the top three predictors; CH₄ flux was dominated by TN, followed by C/N and TC; and N₂O flux was overwhelmingly dominated by NH₄⁺-N, far exceeding all other factors. In the CF lawn, CO₂ flux was dominated by MBC, followed by ST and AT; CH₄ flux was primarily governed by VWC, followed by TC and TN; and N₂O flux was dominated by NH₄⁺-N, followed by NO₃⁻-N and MBN. The RF ranking patterns of the two lawn types were distinctly different, with the BF lawn pointing toward temperature (ST/AT) and nitrogen (TN/NH₄⁺-N/C/N), and the CF lawn pointing toward microbial biomass (MBC/MBN) and moisture (VWC). These patterns closely matched the SEM core drivers.

SHAP analysis further validated the direction of environmental factor effects. For CO₂, high values of ST and AT corresponded to positive SHAP values (promoting emission) in both lawn types, whereas high MBC values (CF lawn) corresponded to negative SHAP values, consistent with the SEM results. For CH₄, high values of ST and TC corresponded to negative SHAP values in the BF lawn, while high VWC values corresponded to positive SHAP values in the CF lawn, again matching the SEM pathways. For N₂O, high values of NH₄⁺-N and SOC corresponded to negative SHAP values in the BF lawn, whereas high PI and MBN values corresponded to positive SHAP values in the CF lawn, consistent with their SEM effects. Collectively, the cascading evidence from Mantel tests, SEM, RF, and SHAP analyses demonstrates that GHG fluxes from the BF lawn are governed by a "temperature–inorganic nitrogen dominated" regime, whereas those from the CF lawn follow a "moisture–microbial–carbon pool co-regulation" regime.

3.3.2 Effects of Mowing Management on GHG Fluxes

Figure 8 illustrates the changes in GHG fluxes before and after three mowing events (spring, summer, and autumn) for urban BF and CF lawns. For CO₂ fluxes, the BF lawn exhibited elevated emissions following all three mowing events, with increases of 49.6%, 42.1%, and 9.1% relative to pre-mowing levels, respectively; the increase was statistically significant for the summer event. In the CF lawn, summer mowing led to a significant 72.1% increase, spring mowing caused a non-significant slight increase, and autumn mowing resulted in a significant 22.1% decrease. Overall, summer mowing most strongly stimulated CO₂ emission from both lawn types.



359 For CH₄ fluxes, spring and summer mowing reduced the CH₄ uptake capacity of the
 360 BF lawn by 41.7% and 40.8%, respectively (with the summer difference being statistically
 361 significant), whereas autumn mowing significantly enhanced CH₄ uptake by 50.8%. In the
 362 CF lawn, mowing produced no significant change in CH₄ uptake during any of the three
 363 seasons. Thus, mowing exerted a stronger effect on CH₄ flux in the BF lawn than in the CF
 364 lawn.

365 Regarding N₂O fluxes, the BF lawn showed no significant overall change in response
 366 to mowing; flux decreased slightly in spring but increased by 7.1% and 186.5% in summer
 367 and autumn, respectively. In the CF lawn, spring and summer mowing resulted in non-
 368 significant decreases of 27.8% and 35.1%, respectively, whereas autumn mowing produced
 369 a significant increase of 309.8%. Autumn mowing thus substantially promoted N₂O
 370 emission from both lawn types, with a more pronounced effect in the CF lawn.

371 **4. Discussion**

372 **4.1 Seasonal Variation Differences in GHG Fluxes**

373 The CO₂ fluxes from lawns beneath urban BF and CF exhibited a pronounced "high
 374 in summer, low in winter" seasonal pattern, consistent with observations from temperate
 375 grasslands, semi-arid agro-pastoral transitional zones, and forest–grassland ecotones
 376 (Wang et al., 2023; Wang et al., 2019; Zou et al., 2025). the combination of high
 377 temperature and abundant precipitation creates favorable soil thermal and moisture
 378 conditions. Vigorous plant growth simultaneously enhances root autotrophic respiration
 379 and soil microbial heterotrophic respiration, and elevated temperatures accelerate soil
 380 organic carbon decomposition and microbial metabolic activity, collectively driving CO₂
 381 flux to its annual maximum (Guo et al., 2018). In winter, low temperatures suppress root
 382 and microbial activity, slowing organic matter decomposition (Sang et al., 2021).

383 We found that CO₂ flux from the CF lawn consistently exceeded that from the BF
 384 lawn, a difference likely attributable to disparities in understory vegetation. The herbaceous
 385 plant density in the CF lawn was 1.46 times that of the BF lawn, and both Simpson and
 386 Shannon–Wiener diversity indices were higher overall (Table 2). Coniferous trees have
 387 narrower crowns and relatively lower canopy closure, which improves understory light
 388 conditions and promotes herbaceous plant growth and species diversity (Zheng et al., 2022).
 389 The greater herbaceous plant abundance not only elevates the direct contribution of root



390 autotrophic respiration but also stimulates microbial activity and rhizosphere priming
391 effects through enhanced litter inputs and root exudates (Shahzad et al., 2015). The positive
392 effect of plant diversity on microbial respiration has been widely corroborated by global
393 meta-analyses (Chen et al., 2019). Superimposed on these biotic factors, the higher ST in
394 the CF lawn further promotes microbial metabolism, jointly explaining the persistently
395 higher CO₂ flux observed relative to the BF lawn.

396 CH₄ flux patterns were consistent with studies on urban lawns in Poland and North
397 American grassland ecosystems (Liebig et al., 2010; Yaroslav et al., 2022). However, not
398 all grassland ecosystems act as CH₄ sinks. Global meta-analyses indicate that managed
399 grasslands subjected to intensive grazing, fertilization, and land-use change have
400 transitioned from CH₄ sinks to emission sources (Chang et al., 2021; Li et al., 2025), and
401 analogous sink-to-source reversals have been documented in degraded Amazonian pasture
402 soils (Fonseca et al., 2022). The fundamental explanation for this divergence lies in VWC
403 and soil aeration status: urban lawns and natural grasslands typically maintain relatively
404 low VWC, which favors methanotroph activity and promotes CH₄ uptake. By contrast,
405 when soil becomes water-saturated due to compaction, excessive irrigation, or
406 waterlogging, oxygen diffusion is restricted, methanotrophy is suppressed, and CH₄
407 emission is enhanced (Bai et al., 2026; Wang et al., 2022). Our results confirm that VWC
408 is the key factor regulating CH₄ flux, further supporting this mechanistic framework.

409 The higher CH₄ uptake fluxes observed in autumn and winter, relative to other seasons,
410 likely stem from the slow decomposition of organic matter accumulated during the growing
411 season (e.g., litterfall), which provides a sustained carbon source for microorganisms.
412 Combined with the cold tolerance of certain methanotrophs, this facilitates CH₄ uptake
413 during the colder months (Yang et al., 2024). We further found that CH₄ uptake was
414 stronger in the CF lawn than in the BF lawn, potentially reflecting differences in vegetation,
415 ST, and VWC between the two sites. The BF lawn accumulated more litter in autumn and
416 winter, and litter-derived monoterpenes may be mineralized by soil microorganisms,
417 inhibiting particulate methane monooxygenase (pMMO) and thereby attenuating CH₄
418 oxidation (Maurer et al., 2008). This mechanism likely contributes to the relatively lower
419 CH₄ uptake observed in the BF lawn compared with the CF lawn.



N₂O flux trends were consistent with observations from Australian urban lawns and French temperate urban understory lawns, where N₂O fluxes were higher in summer than in other seasons owing to warm ST and sufficient precipitation (Tom et al., 2023; van Delden et al., 2018). Previous studies have demonstrated that N₂O emission flux is positively correlated with temperature; temperatures above 5 °C generally stimulate nitrifier and denitrifier activity as well as nitrogen-related enzymatic processes, with maximum N₂O emission occurring in the 25–35 °C range (Agehara and Warncke, 2005; Butterbach-Bahl et al., 2013; Peng et al., 2018). Moreover, precipitation elevates VWC and promotes plant growth and litter decomposition, thereby increasing soil nitrogen supply (Su et al., 2023). Under aerobic conditions, nitrifying bacteria oxidize NH₄⁺-N, producing N₂O, while denitrifying bacteria reduce NO₃⁻-N to N₂O or N₂, jointly driving elevated summer N₂O fluxes (Levy-Booth et al., 2014). Although VWC does not decline in winter, low temperatures substantially weaken nitrifier and denitrifier activity, resulting in reduced N₂O fluxes (Hou et al., 2025). Additionally, N₂O flux was greater in the BF lawn than in the CF lawn. This may be attributable to the higher VWC (Fig. 2a) and greater litterfall in the BF lawn, which produce humus that provides nutrients for soil microorganisms (Liang et al., 2025), thereby enhancing nitrification and denitrification relative to the CF lawn and yielding higher N₂O production.

4.2 Diurnal Variation Differences in GHG Fluxes

CO₂ flux peaks for both BF and CF lawns occurred during nighttime or the early morning hours, exhibiting a "daytime low, nighttime high" trend. This pattern deviates markedly from the typical "daytime high, nighttime low" unimodal pattern commonly seen in grassland soil respiration. For example, diurnal CO₂ flux maxima in alpine meadows on the Qinghai–Tibet Plateau consistently occur between 11:00 and 16:00 (Hang et al., 2022). The anomalous pattern observed in the present study is likely linked to daytime water stress induced by high temperatures. Under strong daytime solar radiation, elevated ST intensifies surface soil water evaporation. When VWC drops below a critical threshold, the inhibitory effect of water stress on root and microbial activity outweighs the direct stimulatory effect of temperature (Li et al., 2022), leading to reduced CO₂ emission. During nighttime, radiative cooling causes RH to rebound (Fig. S2), alleviating water limitation and partially restoring root respiration and soil microbial activity, thereby increasing CO₂ emission (Hao



451 et al., 2025). Studies have shown that VWC and ST are key controlling factors for grassland
452 CO₂ flux at the diurnal scale (Ma et al., 2017; Wang et al., 2021). Our results further
453 indicate that, across all four seasons, diurnal CO₂ fluxes were consistently higher from the
454 CF lawn than from the BF lawn. This may reflect that in spring and summer, the large,
455 dense leaves of BF trees enhance daytime CO₂ assimilation by the BF lawn through
456 photosynthesis, whereas at night, the greater herbaceous plant abundance in the CF lawn
457 (Table 2) leads to higher plant respiratory CO₂ release (Tang et al., 2025). In autumn and
458 winter, leaf litterfall in the BF lawn forms a physical barrier that intercepts solar radiation,
459 lowering soil temperature (Fig. 2a), reducing microbial organic matter decomposition rates,
460 and consequently diminishing CO₂ emission.

461 Diurnal CH₄ pattern aligns with the general finding from global meta-analyses that
462 grassland soil CH₄ uptake weakens with increasing soil temperature (Li et al., 2025), as
463 elevated temperatures may enhance methanogen activity and thereby weaken net CH₄
464 uptake during the day (Yang et al., 2021). In autumn and winter, diurnal CH₄ dynamics
465 were consistent between the two lawn types, with no discernible diurnal pattern. This is
466 attributable to the fact that the optimal temperatures for most methanogens and
467 methanotrophs are approximately 30–40 °C and 20–30 °C, respectively (Arikan et al., 2015;
468 Mo et al., 2025), whereas autumn and winter soil temperatures in these lawns generally
469 remain below 20 °C (Fig. S2), suppressing the activity of both groups. In summer, net CH₄
470 flux was higher in the CF lawn than in the BF lawn, whereas the opposite held in spring,
471 with no significant difference in other seasons. During summer, higher ST and VWC in the
472 CF lawn (Fig. S2) created favorable hydrothermal conditions for methanogens, leading to
473 relatively increased CH₄ emission. In spring, the BF lawn had markedly higher VWC,
474 coupled with moss layer coverage, which restricted O₂ diffusion in the soil, inhibiting
475 methanotrophs and stimulating methanogens (Wei et al., 2015), resulting in a higher net
476 CH₄ flux.

477 The N₂O dynamics observed here are broadly consistent with those reported by Harris
478 et al. (2018) for European mountain meadows, where abandoned meadows emitted N₂O
479 during most measurement periods but exhibited occasional uptake. The differences
480 between the two lawn types are likely attributable to variations in vegetation composition,
481 ST, and VWC (Ribeiro et al., 2016; Taylor et al., 2017). We observed that when VWC



482 exhibited erratic fluctuations, N₂O flux likewise displayed irregular variations and even
483 shifted to uptake. For example, N₂O uptake occurred during nighttime in the BF lawn in
484 spring and during daytime in the CF lawn in summer, corresponding to periods when VWC
485 exhibited M-shaped irregular fluctuations (Fig. S2). A global meta-analysis by Wu et al.
486 (2021) demonstrated that approximately 80% of datasets exhibited significant diurnal N₂O
487 patterns, with water-filled pore space (WFPS) identified as a key factor modulating these
488 patterns. Furthermore, during autumn and winter, nighttime temperature differences were
489 small and N₂O fluxes remained stable in both lawn types. During daytime, in contrast,
490 temperature differences were larger; the BF lawn exhibited N₂O uptake, whereas the CF
491 lawn showed no significant change. This divergence may arise from leaf litter coverage in
492 the BF lawn, which creates a physical barrier that intercepts daytime solar radiation,
493 lowering daytime ST and suppressing nitrification and denitrification, thereby attenuating
494 N₂O emission intensity. The regulatory role of the litter layer on soil–atmosphere N₂O flux
495 has been corroborated in studies of subtropical plantations (Wang et al., 2014).

496 **4.3 Effects of Mowing Management on GHG Fluxes**

497 Mowing constitutes one of the most important anthropogenic factors influencing GHG
498 fluxes from urban lawns. The CO₂ pattern likely arises because mowing increases bare soil
499 surface area, allowing greater solar radiation penetration that elevates soil temperature,
500 thereby stimulating root and microbial activity and accelerating organic matter
501 decomposition (Sang et al., 2021). During the non-growing season, soil temperatures are
502 low, and mowing removes vegetation and litter, reducing soil organic matter input and
503 weakening root and microbial metabolic activity (Li et al., 2010; Sun et al., 2013).
504 Consequently, post-mowing CO₂ flux declined in the CF lawn. In the BF lawn, however,
505 abundant autumn leaf litter partially compensated for the reduction in organic matter
506 caused by mowing.

507 CH₄ fluxes from both lawn types were likewise higher after mowing than before
508 during the growing season, with the reverse pattern observed in the non-growing season.
509 Meteorological data indicated that precipitation events occurred both before and after
510 mowing during the growing season, elevating VWC and restricting O₂ diffusion in the soil,
511 thereby inhibiting methanotrophs and reducing CH₄ uptake (Wei et al., 2015). During the
512 non-growing season, pre-mowing conditions were relatively dry, limiting methanotroph



activity through water stress (van den Pol-van Dasselaar et al., 1998); post-mowing precipitation increased soil moisture to levels favorable for methanotrophs, enhancing CH_4 uptake. The absence of a drought-inhibition effect on CH_4 flux in the CF lawn may reflect differences in vegetation type and microbial community structure, though the specific mechanisms warrant further investigation.

N_2O fluxes from the two lawn types exhibited divergent responses to summer mowing but both declined after spring mowing and increased after autumn mowing. The contrasting summer N_2O responses are likely related to differential VWC responses: after summer mowing, VWC increased in the BF lawn but decreased in the CF lawn (Fig. 2a). Spring mowing reduced soil nitrogen supply (Berliner and Kioko, 1999), and combined with low ST, suppressed nitrifier and denitrifier activity, thereby decreasing N_2O emission (Hou et al., 2025). In autumn, abundant leaf litter combined with increased post-mowing precipitation promoted litter decomposition, furnishing abundant nitrogen substrates for nitrifiers and denitrifiers and consequently elevating N_2O emission.

4.4 Integrated Effects of Environmental Factors

The "temperature–inorganic nitrogen dominated" regime of the BF lawn reflects a coupled litter-nitrogen mineralization mechanism. The strong driving effect of ST on CO_2 flux arises from the temperature sensitivity of soil organic matter decomposition, with reaction rates following the Arrhenius equation and Q_{10} typically ranging between 1.5 and 3.0 (Rustad et al., 2001). The substantial litter input further accelerates native organic carbon mineralization through priming effects (Zhuang et al., 2023). $\text{NH}_4^+\text{-N}$ plays a pivotal role in regulating all three GHGs: its direct positive effect on CO_2 reflects the promotion of microbial metabolism by $\text{NH}_4^+\text{-N}$ through provision of nitrification substrates (Liu et al., 2023); its direct negative effect on N_2O arises from stoichiometric coupling of nitrification–denitrification. When $\text{NH}_4^+\text{-N}$ is abundant, ammonia oxidation preferentially produces NO_3^- rather than N_2O , and under sufficient carbon supply, denitrification tends toward complete reduction to N_2 (Butterbach-Bahl et al., 2013). The concurrent negative effect of SOC on N_2O further indicates that carbon substrate availability promotes complete denitrification (Stuchiner and von Fischer, 2022). The direct negative effects of TC and MBC on CH_4 flux suggest that higher soil carbon substrates favor the maintenance of methanotroph activity and community abundance, thereby promoting CH_4 uptake. A global



544 meta-analysis by Petr et al. (2024) identified TOC as the primary driver of pmoA gene
545 abundance. VWC was identified as a significant correlate of CH₄ flux in both Mantel tests
546 and SEM for the BF lawn, yet its predictive contribution in RF ranked fourth, below C/N,
547 ST, and TN, indicating that CH₄ uptake in the BF lawn is not dominated by moisture alone;
548 rather, the combined regulation of carbon–nitrogen substrate supply and temperature is
549 equally important. This pattern contrasted with the VWC-dominated regime observed in
550 the CF lawn.

551 The "moisture–microbial–carbon pool co-regulation" regime of the CF lawn exhibited
552 a fundamentally different mechanism. MBC ranked first in predicting CO₂ flux and showed
553 a direct negative effect in SEM. High MBC may suppress CO₂ emission, as studies have
554 shown that when microbial carbon pools are large, microbial communities tend toward
555 growth accumulation rather than respiratory consumption, with more carbon sequestered
556 as necromass (Schnecker et al., 2023). ST and PI indirectly promoted CO₂ emission by
557 reducing MBC, suggesting that carbon fluxes in the CF lawn are mediated primarily
558 through the microbial carbon pool rather than direct thermal effects. For CH₄, VWC
559 emerged as the core controlling factor, with Mantel tests ($r = 0.43$), SEM ($\beta = 0.43$), and
560 RF (top rank) all independently pointing to VWC as the primary regulator of CH₄ flux in
561 the CF lawn. Our observed linear positive correlation between VWC and CH₄ flux indicates
562 that CH₄ uptake weakens with increasing moisture. Koyama et al. (2024) further reported
563 a parabolic response of CH₄ uptake to WFPS in U.S. Great Plains grasslands, where
564 methanotroph activity is limited at both very low and very high moisture levels due to
565 restricted microbial activity and gas diffusion, respectively, suggesting that VWC
566 regulation of CH₄ may exhibit nonlinearity across a broader moisture gradient. For N₂O,
567 NH₄⁺-N was the top predictor in both lawn types, whereas the contributions of PI and MBN
568 were elevated in the CF lawn, indicating that N₂O emissions in the CF lawn are more
569 dependent on precipitation-driven anaerobic conditions and internal microbial nitrogen
570 transformation. Increased precipitation elevates WFPS, creating anaerobic conditions
571 favorable for denitrification (Krichels et al., 2025), consistent with the direct positive effect
572 of PI on N₂O in SEM. The negative effect of C/N on N₂O further supports that adequate
573 carbon supply promotes complete denitrification (Stuchiner and von Fischer, 2022).



Quantitative comparison of the two canopy types with existing literature helps to contextualize the source–sink positioning of subtropical understory lawns within the global urban green space GHG budget. Tom et al. (2023) found that tree canopy shading reduced soil respiration in temperate understory lawns by 34%, whereas in our study, CO₂ flux from the BF lawn was only 42.9% of that from the CF lawn (~1.7 times lower), reflecting the stronger shading and evapotranspiration-induced mitigation effects of highly closed broad-leaved canopies in the subtropics. Trémeau et al. (2024) observed near-balanced CH₄ exchange in boreal urban lawns in Helsinki, whereas our CH₄ uptake intensity was approximately 3–4 times higher, likely attributable to the maintenance of methanotroph activity by the consistently higher soil temperatures in the subtropics, although differences in soil type and management practices between the two regions cannot be discounted. The annual mean N₂O emissions in our study were comparable to those reported for subtropical urban lawns in Australia (van Delden et al., 2018), suggesting that under long-term low-nitrogen input conditions, nitrogen input, rather than climatic differences, is the primary controlling factor for N₂O emissions across climate zones. The pronounced differences in lawn GHG emissions observed across studies further highlight the critical need for in-depth investigations of urban lawns.

591

5. Conclusion

This study elucidates the canopy-type-dependent differences, spatiotemporal patterns, and key environmental driving mechanisms of CO₂, CH₄, and N₂O fluxes from urban lawns beneath BF and CF. The mean annual CO₂ flux from the CF lawn was significantly higher than that from the BF lawn. Both lawn types exhibited an atypical "daytime low, nighttime high" diurnal CO₂ pattern, contrasting with the canonical "daytime high, nighttime low" pattern commonly observed in grassland soil respiration. Both lawn types acted as net CH₄ sinks throughout the year, with stronger uptake capacity in the CF lawn; VWC was identified as the key common regulatory factor. N₂O fluxes were predominantly emissive across both types, with a "high in spring–summer, low in autumn–winter" seasonal trend, and emissions from the BF lawn exceeded those from the CF lawn. SEM, RF, and SHAP analyses jointly revealed two contrasting driver regimes: "temperature–inorganic nitrogen dominated" for the BF lawn and "moisture–microbial–carbon pool co-regulation" for the



605 CF lawn. The RF importance rankings and SHAP-derived effect directions were highly
606 consistent with the SEM path coefficients, collectively confirming the robustness of the
607 identified regulatory patterns. Mowing effects were season-dependent: summer mowing
608 significantly promoted CO₂ emission, autumn mowing enhanced CH₄ uptake and N₂O
609 emission, whereas spring mowing exerted the smallest impact. These findings suggest that
610 optimizing the timing and intensity of mowing management could contribute to emission
611 mitigation in urban lawns. Overall, this study provides empirical evidence to address the
612 variability in GHG emission dynamics and regulatory mechanisms in urban lawns, offering
613 a scientific basis for carbon sink monitoring, greenhouse gas assessment, and low-carbon
614 urban green space management.

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617 **Data availability**

618 Data will be made available on request.

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621 **Author contributions**

622 Ziyang Wu performed greenhouse gas sampling and the collection and measurement of soil
623 variables including soil temperature and soil volumetric water content, analyzed the dataset
624 and wrote the manuscript. Shaohui Li measured soil pH and carried out data analysis, as
625 well as implementing the random-forest (RF) machine-learning analyses. Cunxin Hu
626 assisted Zihan Fan with the SHAP machine-learning analysis. Fuhao Li and Kunpeng Zang
627 contributed to figure preparation. Shuangxi Fang reviewed and edited the manuscript. Shuo
628 Liu took part in data analysis, writing and manuscript revision.

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630

631 **Competing interests**

632 The contact author has declared that none of the authors has any competing interests.



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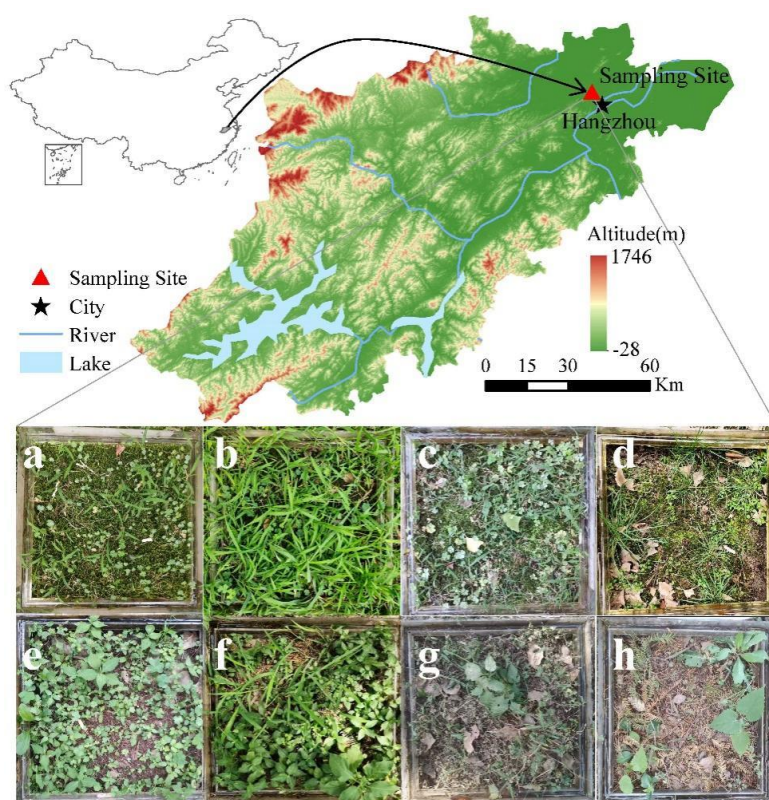
837 **Tables and Figures**

838 **Table 1.** Seasonal variations in CO₂, CH₄, and N₂O fluxes from the BF and CF urban lawns. Different lowercase letters indicate
839 significant differences in GHG fluxes among seasons within each lawn type ($p < 0.05$).

	CO ₂		CH ₄		N ₂ O	
	BF (mg C m ⁻² h ⁻¹)	CF (mg C m ⁻² h ⁻¹)	BF (ug C m ⁻² h ⁻¹)	CF (ug C m ⁻² h ⁻¹)	BF (ug N m ⁻² h ⁻¹)	CF (ug N m ⁻² h ⁻¹)
Spring	64.00±15.40 ^d	162.58±13.03 ^b	-15.37±1.86 ^a	-23.98±2.58 ^{ab}	9.48±1.54 ^{ab}	6.08±1.05 ^{ab}
Summer	152.70±19.59 ^{bc}	355.66±16.08 ^a	-20.51±2.05 ^a	-20.52±2.85 ^a	11.57±1.23 ^a	9.94±2.02 ^a
Autumn	86.73±7.06 ^{cd}	163.01±19.04 ^b	-24.89±2.22 ^{ab}	-32.65±2.43 ^b	4.32±0.72 ^b	4.15±0.89 ^b
Winter	9.63±15.00 ^d	80.70±8.30 ^{cd}	-21.52±4.64 ^{ab}	-24.49±6.24 ^{ab}	3.89±0.69 ^b	3.29±0.53 ^b

841 **Table 2.** Biodiversity indices pre- and post-mowing of lawns beneath the BF and CF urban lawns.

	BF						CF					
	Pre-mowing			Post-mowing			Pre-mowing			Post-mowing		
	March	June	November	March	June	November	March	June	November	March	June	November
Number	115	290	583	0	0	78	909	346	192	828	795	112
Simpson	0.48	0	0.50	0	0	0	0.13	0.41	0.62	0.48	0.36	0.44
Shannon	0.67	0	0.70	0	0	0	0.26	0.60	1.03	0.85	0.73	0.63



843

844 **Figure 1.** Geographical location of the study area and schematic diagram of seasonal
 845 sampling sites for lawns under BF (a: Spring; b: Summer; c: Autumn; d: Winter) and CF
 846 (e: Spring; f: Summer; g: Autumn; h: Winter).

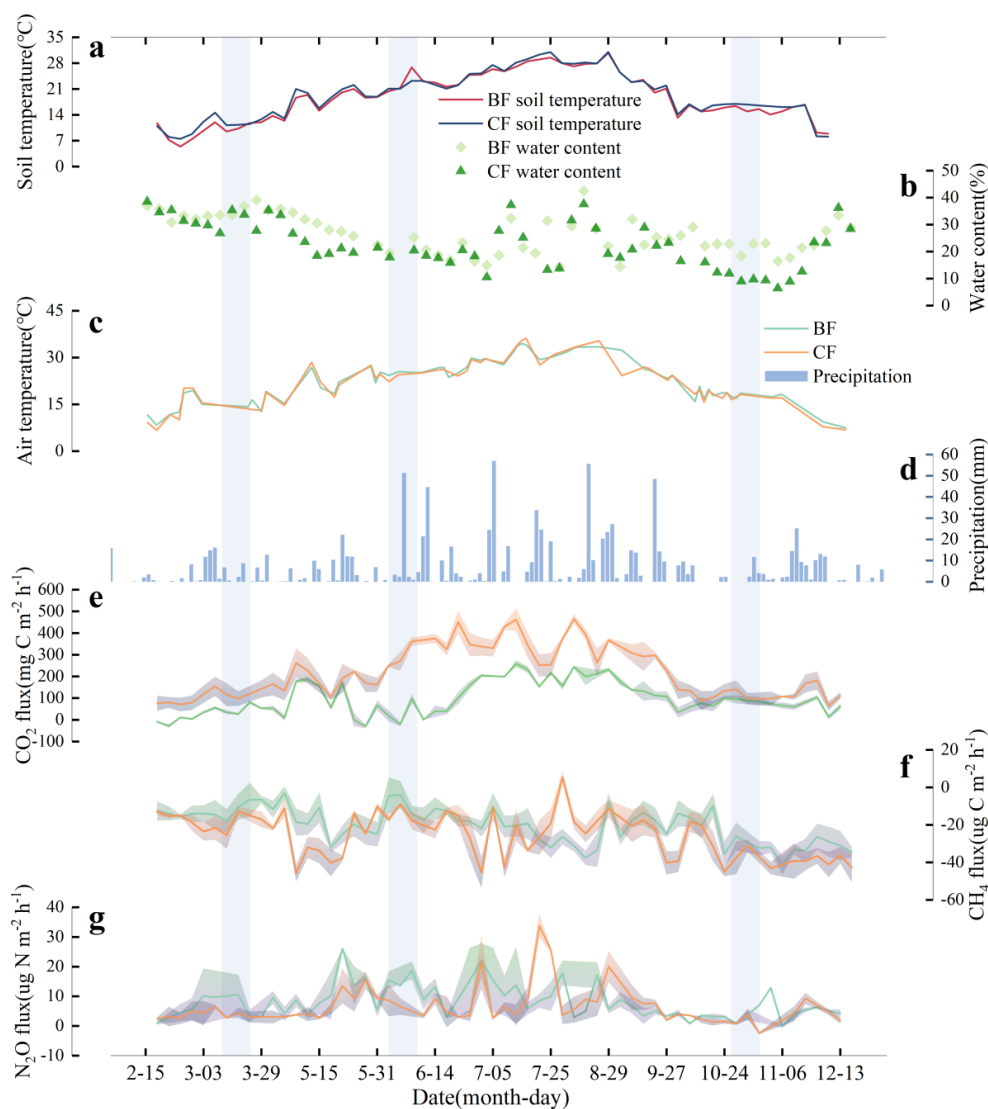


Figure 2. Seasonal dynamics of environmental factors and greenhouse gas fluxes. (a) Soil temperature and soil volumetric water content, (b) air temperature and precipitation, and greenhouse gas fluxes from the BF and CF urban lawns (c: CO₂; d: CH₄; e: N₂O). Dashed lines indicate lawn mowing dates.

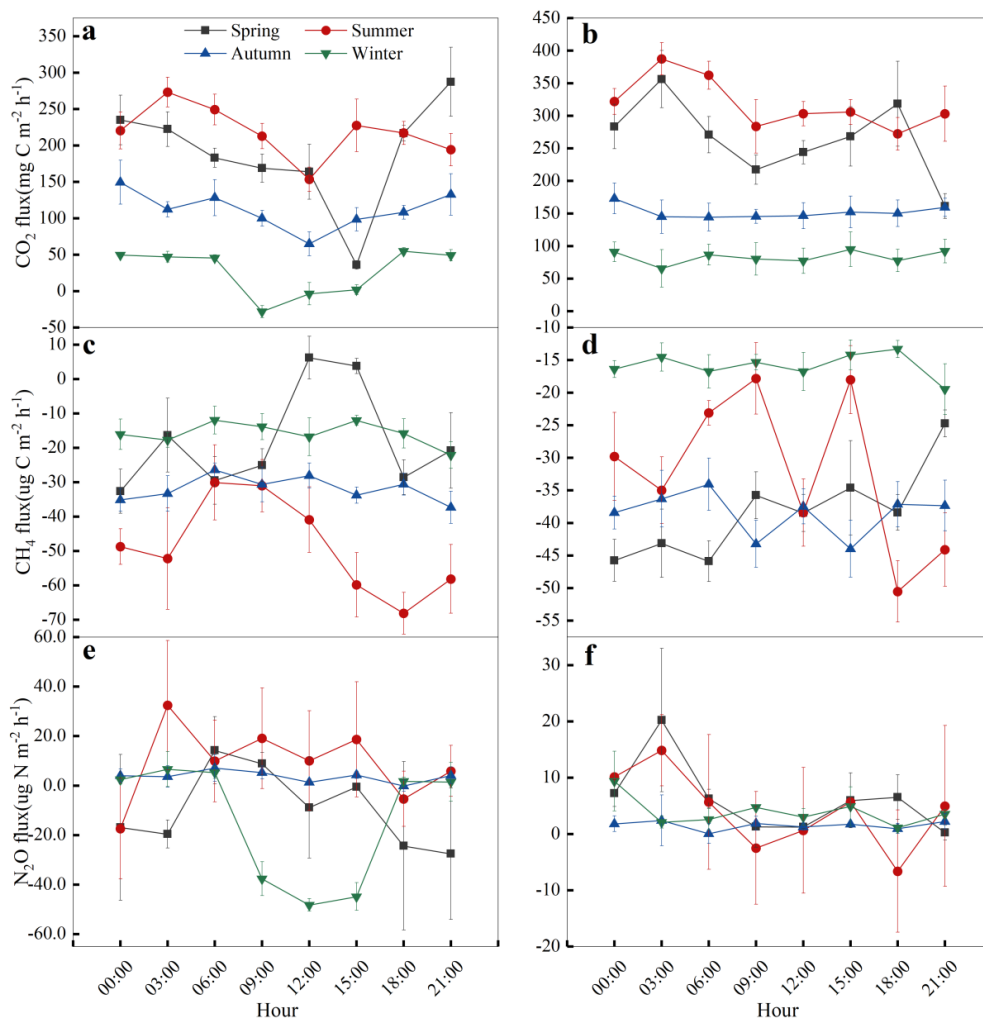
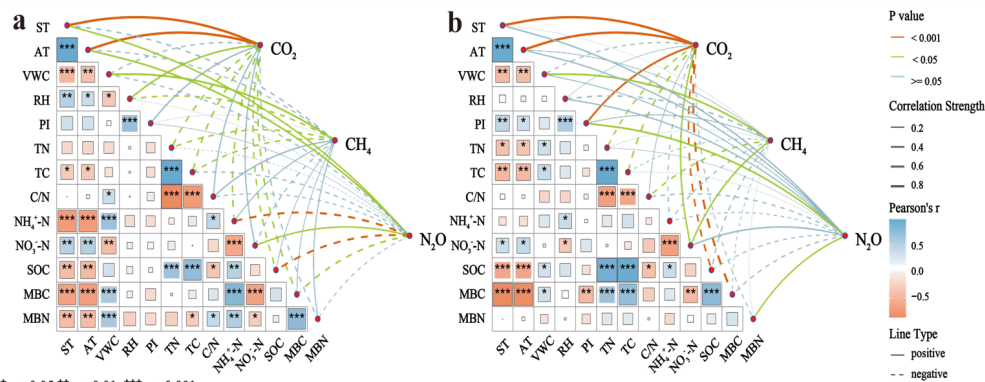
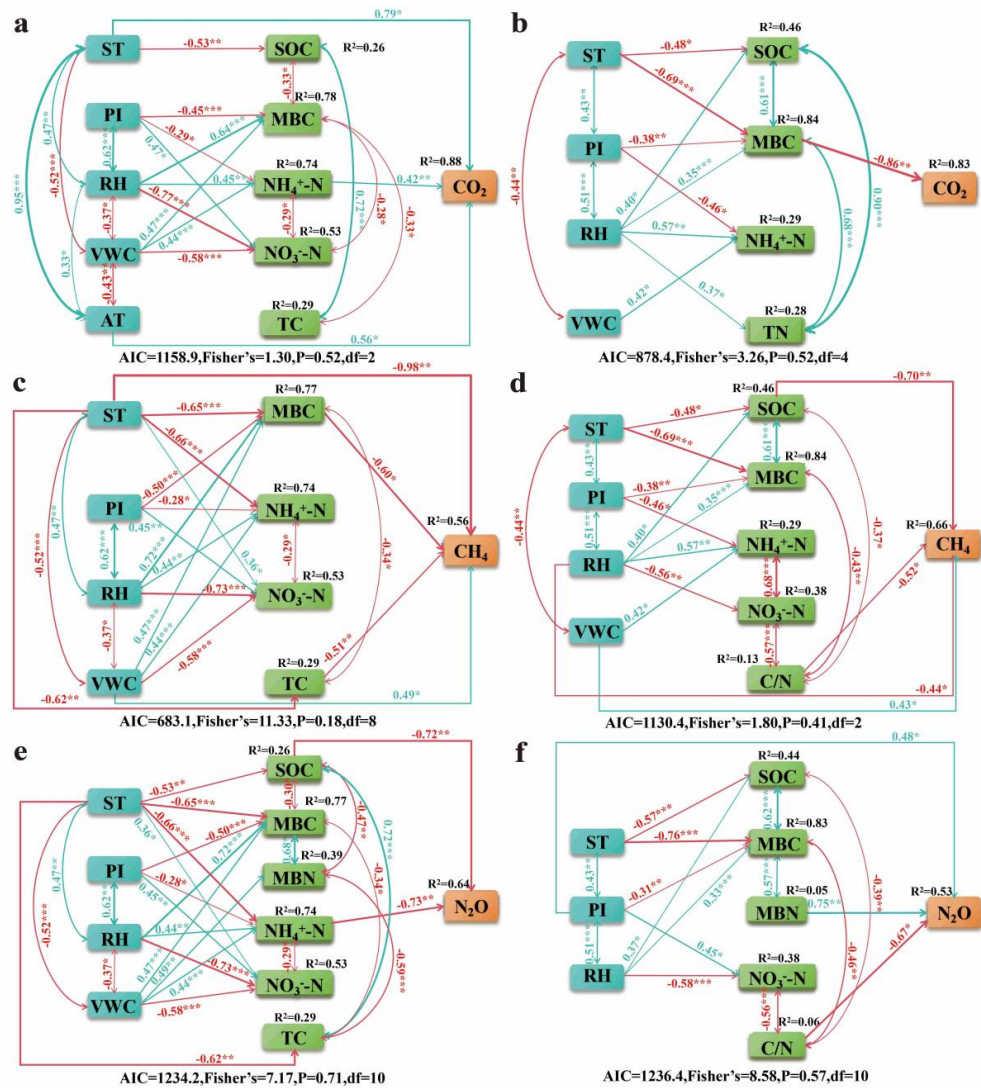


Figure 3. Diurnal dynamics of greenhouse gas fluxes in BF (a, c, e) and CF (b, d, f) urban lawns.

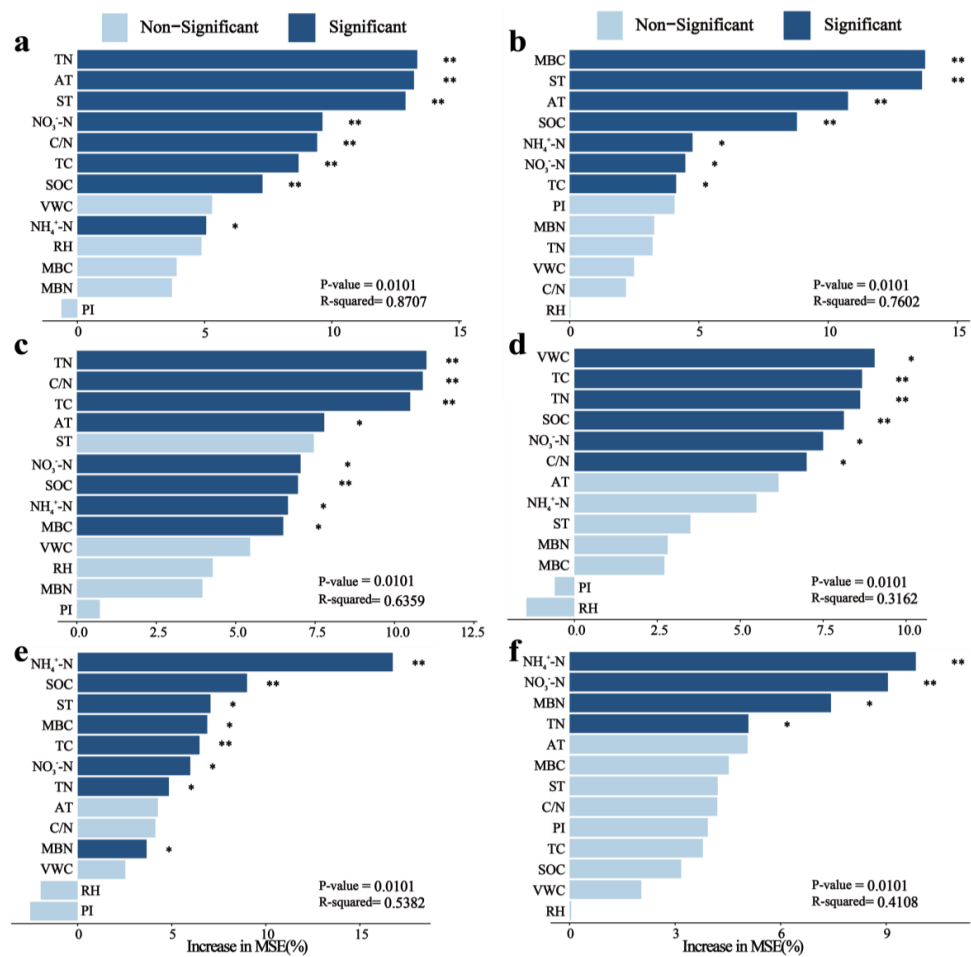


855 * $p < 0.05$ ** $p < 0.01$ *** $p < 0.001$
856 **Figure 4.** Correlation heatmap of environmental factor effects on greenhouse gas fluxes in
857 BF (a) and CF (b) urban lawns. The color gradient of Pearson's r indicates Pearson
858 correlation coefficients among environmental factors; line width of Mantel's r reflects the
859 correlation between GHG fluxes and environmental factors, and line color shows the
860 significance level (Mantel's P).



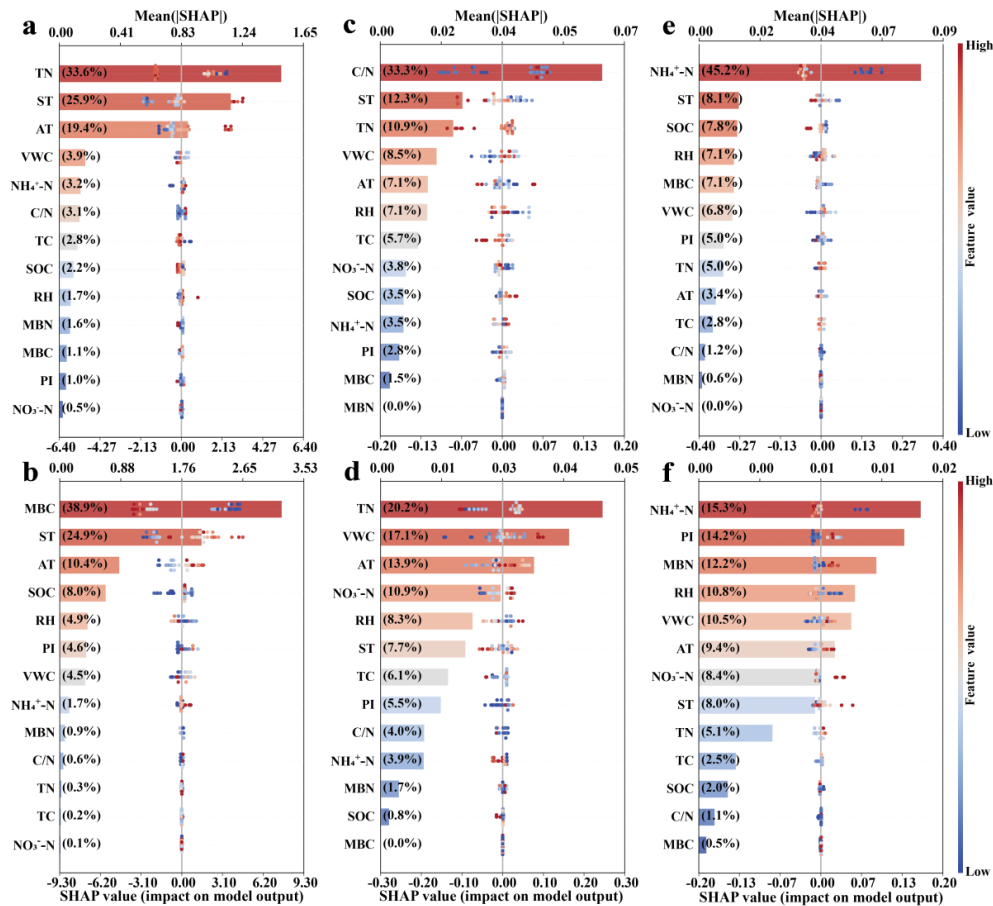
* p<0.05 **p<0.01 ***p<0.001

Figure 5. Structural equation models (SEM) illustrating relationships between environmental factors and greenhouse gas fluxes in BF (a, c, e) and CF (b, d, f) urban lawns. Arrow width reflects standardized path coefficients. Solid blue and red arrows denote positive and negative paths, respectively. Numbers on arrows are standardized path coefficients, and R² indicates the explained variance of each dependent variable.



867

868 **Figure 6.** Variable importance rankings of environmental factors for predicting soil
869 greenhouse gas fluxes in BF (a, c, e) and CF (b, d, f) urban lawns based on Random Forest
870 models (a, b: CO₂; c, d: CH₄; e, f: N₂O). Dark blue bars denote variables with significant
871 predictive effects; light blue bars represent non-significant predictors. Asterisks indicate
872 significance levels: * $p < 0.05$, ** $p < 0.01$.



873

874 **Figure 7.** SHAP summary plots revealing the direction and magnitude of environmental
875 factor effects on soil greenhouse gas fluxes in BF (a, c, e) and CF (b, d, f) urban lawns(a,
876 b: CO₂; c, d: CH₄; e, f: N₂O).Each point represents a single observation; the x-axis position
877 indicates the SHAP value, and the color gradient (blue to red) denotes the feature value
878 from low to high. The bar length represents the mean absolute SHAP value.

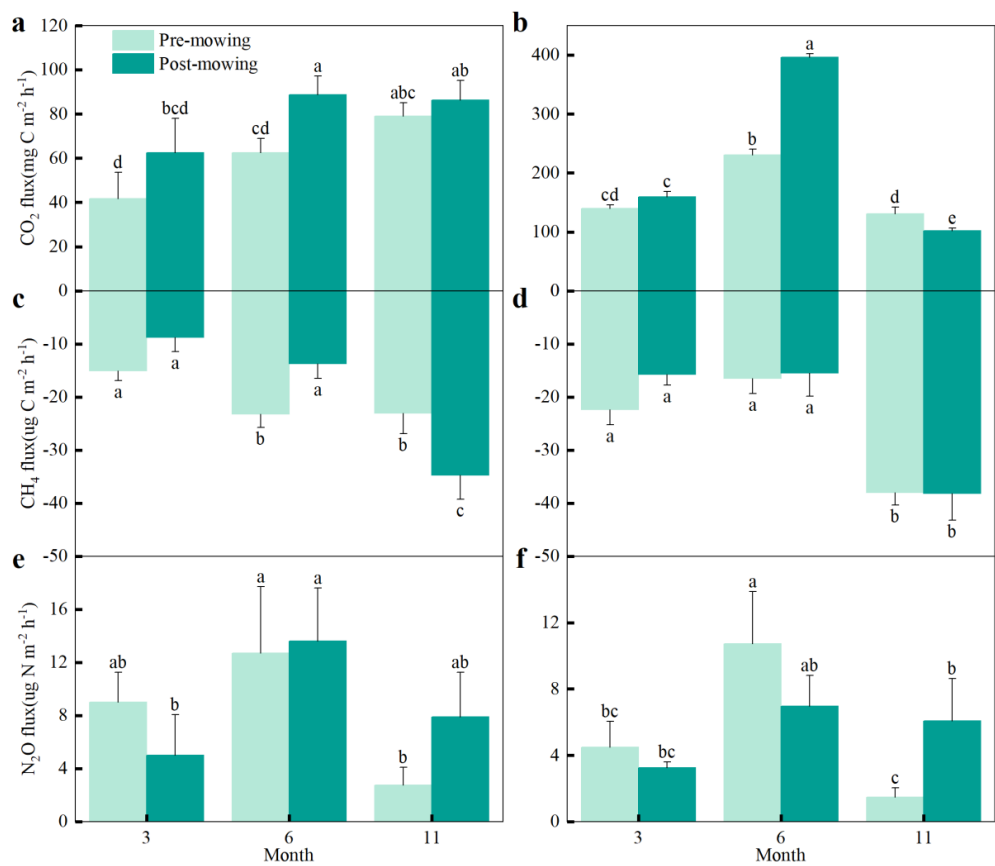


Figure 8. Changes in greenhouse gas fluxes before and after mowing events in BF (a, c, e) and CF (b, d, f) urban lawns. Different lowercase letters denote significant differences in greenhouse gas fluxes of understory lawns between pre-mowing and post-mowing stages.