Grass Research

ARTICLE

Open Access

https://doi.org/10.48130/grares-0026-0014

Grass Research **2026**, 6: e022

Temporal and spatial variations in methane flux across golf course fairways, rough, and restored native grass areas

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Abstract

Methane (CH_4) is an important component of the carbon cycle that is emitted and absorbed by soils. Few studies have quantified CH_4 flux from turfgrass systems with variable management intensity on golf courses. In 2011–2013, a field study was conducted on golf course fairways, rough, and restored native grass (RNG) areas on two holes (Sites A and B) to measure soil-turf-atmospheric methane flux using the vented chamber method. In 2011–2012, methane emissions from Site A averaged 2.39, 1.66, and 1.18 g CH_4-C $ha^{-1}\cdot d^{-1}$, and decreased to 1.18, 1.33, and 0.82 g CH_4-C $ha^{-1}\cdot d^{-1}$ in 2012–2013 from the fairway, rough, and RNG, respectively. At Site B, emissions averaged 0.78, −0.53, and −2.03 g CH_4-C $ha^{-1}\cdot d^{-1}$ in the fairway, rough, and RNG areas, respectively, indicating that the fairway is a source, while the rough and RNG on Site B were sinks for methane. There was a linear relationship between water-filled pore space (WFPS) and CH_4 emissions from fairways (r = 0.37) and rough (r = 0.59). Frequent irrigation likely reduces methane oxidation potential of soil and restricts methane uptake. This study suggests that best management practices that incorporate site-specific irrigation management, reduce compaction, and promote soil aeration as part of a low-input management strategy could reduce CH_4 emissions in urban turfgrass systems.

Keywords: Methane, Trace gas flux, Turfgrass, Water filled pore space, Water management, Fertilization

Citation: Gillette KL, Qian Y, Follett RF. 2026. Temporal and spatial variations in methane flux across golf course fairways, rough, and restored native grass areas. *Grass Research* 6: e022 https://doi.org/10.48130/grares-0026-0014

Introduction

Methane is a potent greenhouse gas (GHG) that has a global warming potential (GWP) approximately 25–30 times greater than that of carbon dioxide[1]. However, methane has a shorter atmospheric lifetime (~12 years) than that of CO_2 (over 100 years), and its total radiative forcing is substantially lower than that of CO_2 due to its lower atmospheric concentration[2]. Despite this, methane plays an important role in near-term climate warming, and reductions in methane emissions can yield rapid climate benefits[2]. Natural ecosystem processes impact atmospheric methane concentration. Depending on management practices, such as irrigation, nitrogen (N) fertilization, and soil properties that affect soil aeration or compaction, grassland soils may act as simultaneous sinks and sources of CH_4[3,4]. Methane oxidation, which creates methane sinks, is a process in which, under aerated soil or well-drained soils, methane from the atmosphere can diffuse into soil, and then specific microbes (e.g., methanotrophic bacteria) in soil use methane as a carbon and energy source, converting it to CO_2. Methanogenesis, which creates methane sources, is a process where methanogenic microbes break down organic matter under anaerobic conditions, producing methane as a byproduct.

It is generally accepted that soil disturbance, fertilization, and irrigation tend to decrease soil CH_4 oxidation capacity[5−7]. Aerobic soils make up the largest biological sink for methane by fostering soil microbes that use methane as an energy and a carbon source[8]. Therefore, natural temperate grasslands in arid and semiarid climates serve as a critical methane oxidation sink that regulates atmospheric CH_4 concentration. However, anthropogenic land use changes, such as urbanization, alter soil methane oxidation potential and modify methanogenesis conditions, causing significant changes in methane flux[9].

Urban growth along with the associated use of turfgrass[10] has converted native and arable land to suburban homes and cities[11]. The urban population is projected to grow, with seven out of ten people living in urban areas by 2050[12]. Due to urbanization, significant areas of urban and suburban landscapes are planted with turfgrass. In the United States alone, there are 16 million ha (Mha) of turfgrass[13], including home lawns, parks, urban green spaces, sports fields, golf courses, and more. Golf courses are prominent landscape features in urban and suburban areas, with approximately 32,000 courses globally that are typically 60–80 ha in size. In the United States, there are about 16,000 golf courses that collectively occupy around 1.0 Mha[14,15].

There is limited research on turf-atmospheric exchanges of methane[16−19]; however, no research has been done to address the impact of management practices unique to golf courses on methane flux. Different sections of a golf course, such as fairways, rough, and restored native grass areas, employ different management practices. Fairways are typically managed more intensively by receiving more irrigation and more frequent and shorter mowing heights than the rough. Rough is typically managed with similar intensities as home lawns. Restored native grass areas receive occasional irrigation and mowing. Therefore, the degree of management typically decreases from fairway, to rough, to restored native grassland. In addition to generating billions (USD) for the economy, golf courses also serve as important urban green spaces and support ecological biodiversity[14]. The extent to which these benefits are diminished by GHG emissions and their associated environmental consequences is uncertain.

The objectives of this study were to: (1) measure methane exchange from a Colorado golf course's fairways, rough, and RNG areas that have varying degrees of management; and (2) determine the spatial and temporal variations of methane flux and their

https://doi.org/10.48130/grares-0026-0014

associations with soil porosity, water-filled pore space, and temperature. We hypothesize that methane flux from golf course turfgrass is affected by the varying degrees of management intensity, and that native grass should absorb more methane than highly managed golf course turfgrass.

Methods and materials

Study site and experimental design

This trace gas emission study was conducted at a golf course (40.5233° N, 105.0436° W, at an elevation of approximately 1,515 m above sea level) near Fort Collins, located on the Rocky Mountain Front Range of Colorado, where population growth is rapid, and land use has dramatically changed from rural to suburban areas. The region has a semi-arid temperate climate, with a 15-year mean air temperature of 10.2 °C and cumulative annual precipitation of ~ 380 mm.

The golf course location was previously used as a 100-year-old sheep/alfalfa farm. Golf course construction began in 2005 and opened to golfers in 2007[20]. In 2006, perennial ryegrass (*Lolium perenne* L.) was seeded on the fairway, and Kentucky bluegrass (*Poa pratensis* L.) was sodded on the rough. Sections of restored native grasses (RNG), consist of a mixed grass community of smooth bromegrass (*Bromus inermis* Leyss), blue grama (*Bouteloua gracilis* [Willd. ex Kunth] Lag. ex Griffiths), and buffalo grass (*Buchloe dactyloides* [Nutt.] Engelm.). Fairways were mowed three times per week to a 1.9 cm height. Rough was mowed one or two times per week to a 6.35 cm height. Irrigation water was applied every day starting in March and ending in October at approximately 100% of the estimated evapotranspiration (ET) of turfgrass on both the fairway and rough sites. The estimated turfgrass (mowed at 6.35 cm) ET from March to October totaled 86, 94, and 80 cm for 2011, 2012, and 2013, respectively. Based on Brown & Kopec[21], we estimated that fairway ET was approximately 85% of the rough ET rate.

The field study was conducted over the course of two growing seasons. Soil methane samples in Year 1 were collected from June 2011 to April 2012, and Year 2 samples were collected from May 2012 through June 2013. Soil characteristics, including bulk density, soil WFPS, and nitrate concentrations, were determined on the first day of the experiment for each study site. Experimental field plots were selected at hole 16 on day of year (DOY) 122 of 2011. Measurements of soil methane were conducted over the 2-year study period at an adjacent fairway (40.53352° N, 104.95018° W), rough (40.53351° N, 104.95017° W), and RNG (40.53354° N, 104.95017° W). From here on, these plots are referred to as Site A fairway, rough, and native (Table 1). Site A was initially selected due to its accessible position on the course and minimal soil disturbance during golf course construction.

The hole 5 site (Site B) was added in 2012 (Table 1). Soil CH_4 measurements were collected from May 2012 (DOY 95) to June 2013 (DOY 514) on Site B plots located on an adjacent fairway (40.53336° N, 104.93359° W), rough (40.53336° N, 104.93359° W), and RNG (40.53336° N, 104.93360° W) areas. From here on, these plots are referred to as Site B fairway, rough, and native. Site B is located on the 'front nine' of the golf course, where a strongly sloping man-made hill was built during course construction.

On each fairway and rough study site, the area was divided into 9 plots with a randomized complete block design to accommodate three fertilizer treatments with three replications. Each plot measured 1.5 m^2 (Site A) or 2.3 m^2 (Site B) and was separated by 0.6 m unfertilized perennial ryegrass borders. Three fertilizer treatments included two different enhanced efficiency fertilizers: urea with nitrification inhibitor (UNI) (Koch Agronomic Services, LLC, Wichita, KS, USA) and polymer-coated urea (PCU) (Agrium, Loveland, CO, USA), and one control (zero fertilizer), in replications of three. Briefly, UNI contains a soil nitrification inhibitor (nitrification dicyandiamide) and a urease inhibitor (N-[n-butyl] thiophosphoric triamide) as slow-release mechanisms, and PCU is a polymer-coated controlled-release urea fertilizer. Each fertilized fairway and rough plot received a total of 150 kg N ha^{-1}·yr^{-1}, applied in three applications of 50 kg N ha^{-1} each. Site A plots were fertilized in June, July, and September (DOY 165, 210, and 259) of 2011 and again the following year in May, July, and October (DOY 139, 205, and 303) of 2012. Site B plots were fertilized in the same months of 2012, though the last two fertilizations were slightly delayed (DOY 153, 216, and 303) to accommodate intensive trace gas sampling following fertilizer applications.

One year prior to the study, golf course maintenance staff fertilized the fairways and rough in May and July of 2010 with Polyon® at total rates of 250 and 175 kg N ha^{-1}, respectively. No fertilizer was applied to the RNG plots prior to or during the experiment. These plots measured 3 m^2 in size. Irrigation water was typically not applied to RNG sites. However, due to early-season drought conditions in 2012, approximately 20 mm of irrigation water was applied during the first 2 weeks of April.

Soil analysis

Soil baseline samples were extracted on DOY 122 of 2011 from the Site A fairway, rough, and native areas at a 0–10 cm depth with three replications. Baseline soil samples were collected from the Site B fairway, rough, and native areas on DOY 95 of 2012 using the same procedures. Soil samples were collected with a metal soil core of known volume. Each collected intact soil sample was immediately placed carefully into plastic bags to minimize soil disturbance and moisture loss. Soil samples were transported to the laboratory facilities, weighed, and stored in refrigerated coolers at 4 °C until further processing.

Table 1. Bulk density, soil water filled pore space (WFPS), soil organic carbon (SOC), nitrate (NO_3-N), and soil texture from baseline soil samples from the fairways, rough, and restored native grass areas (RNG) within the 0–10 cm soil profile.

Site	Bulk density (g·cm^{-3})	Soil WFPS (%)	Soil organic carbon (g·kg^{-1} soil)	NO_3-N (mg·kg^{-1} soil)	Texture
Site A- Fairway	1.43 a$^{\Psi}$	64*	9.5 a$^{\Psi}$	25.6 a$^{\Psi}$	Sandy clay loam/sandy loam
Site A-Rough	1.52 a	62**	8.4 a	37.4 a	Sandy clay loam/sandy loam
Site A-RNG	1.42 a	36**	0.6 b	1.0 c	Sandy clay loam
Site B-Fairway	1.10 b	50	11.3 a	10.5 b	Sandy clay loam/sandy loam
Site B-Rough	1.09 b	40	10.0 a	7.5 b	Sandy clay loam
Site B-RNG	1.14 b	24	0.4 b	1.1 c	Sandy clay loam

$^{\Psi}$ means followed by different letters indicate significant differences at $p < 0.05$. * mean soil water-filled pore space (WFPS) is presented for Site A and Site B baseline soil samples in 2011 and 2012, respectively; *, ** significant difference between Site A and Site B for the respective turf types at $p < 0.05$ and $p < 0.001$, respectively.

To attain soil moisture at field conditions, 10–15 g of soil was subsampled and placed in a 110 °C drying oven overnight to determine gravimetric soil water content. Gravimetric water content values and soil core volumes were used to calculate soil bulk density. Approximately 200 g of soil was hand-picked to remove roots and plant materials (> 2 mm in length), then passed through a 2 mm sieve, air dried, and sent to the Colorado State University Soil Laboratory and analyzed for pH, soil texture, soil organic matter, and soil nitrate (NO_3^-). Soil organic matter was converted to soil organic carbon (SOC) based on the conversion coefficient of 0.58[22].

Soil water-filled pore space

Water-filled pore space (WFPS) was calculated as:

$$WFPS = \frac{VSWC}{1 - \left(\frac{BD}{PD}\right)} \quad (1)$$

where, VSWC is the measured-volumetric soil water content, BD is the soil bulk density at each site (fairway, rough, and native) for Site A and B presented in Table 1, and particle density (PD) is 2.65 g·cm^{-3}.

Trace gas collection and analysis

The measurement of turf-atmosphere CH_4 exchange for this study began on DOY 166 of 2011 and continued through DOY 172 of 2013 using the vented chamber method as described by Parkin & Venterea[23]. Trace gas chambers consisted of a removable chamber top and an anchor installed into the soil. Chamber tops were constructed from polyvinyl chloride (PVC) pipe measuring 12.5 cm in height and 21.2 cm in diameter. Two PVC trace gas anchors were installed into the soil at the center of each plot as duplicate measurements, totaling 18 anchors on each fairway and rough. The restored native section had one anchor per plot, totaling three anchors that were evenly spaced and installed in the center of each plot. In order to interrupt golf course play or turf maintenance as little as possible, collars were installed to a depth such that they were flush with the ground, and the chamber's usual rubber seal was replaced with a fabricated 16-gauge stainless steel band that was bolted to the bottom of the chamber. The steel band overhung the chamber top by 2.5 cm, and a weather filler rod was adhesively applied to the bottom of the PVC pipe, enabling the necessary airtight seal and allowing the chamber to fit securely on top of the anchor during trace gas sample collection.

On measurement days, chambers were placed on anchors, and gas samples from inside the chamber were extracted at three time intervals: 0, 15, and 30 min. Upon installing the chambers, gas samples were extracted using 35 mL polypropylene syringes fitted with nylon stopcocks. Air temperature was measured at the beginning and the end of each 30-min sampling period on all measured sections for methane flux calculation as described below. The temperature inside the chamber was assumed to be equivalent to the ambient air temperature because the chambers were covered with highly reflective foil. On each sampling day, soil temperature and volumetric water content were measured in the top 10 cm of the soil profile at each experimental plot using an EC-Tm probe (Decagon Devices Inc., Pullman, WA, USA).

Upon trace gas collection, samples were transported to the USDA-ARS laboratories in Fort Collins, CO, and 25 ml of gas sample was injected into evacuated tube exetainers that are sealed with butyl rubber septa (Exetainer vial Labco Limited, High Wycombe, Buckinghamshire, UK) for analysis by gas chromatography (GC). The concentration of CH_4 in the injected exetainer sample was determined using a fully automated gas chromatograph instrument (Varian Model Inc., Palo Alto, CA, USA), equipped with a Poropak and Hayesen N packed column and flame ionization detector for CH_4[5]. Gas flux rates were calculated by the linear and non-linear increases in concentration as described by Mosier & DelGado[24]. Flux calculation included adjustments for prevailing air temperature and for an atmospheric pressure of 640 mm mercury (Hg) using the ideal gas law[25].

Methane flux sampling frequency was twice weekly during the turfgrass growing seasons from 2011 to 2013, and the frequency was increased to three times per week the week following N fertilization. Sampling frequency was reduced to once to twice per month during the winter and spring seasons, when turfgrass management activities were minimal. All samples were collected between 0900 h and 1400 h. Overall, methane flux was measured on 77 dates over 2 years at Site A, and on 39 dates over 1 year at Site B.

Statistical analysis

The daily measured flux data were subjected to analysis of variance (ANOVA) using the PROC GLM model of the statistical analysis software (SAS 9.4, SAS Institute Inc., Cary, NC, USA) to test the effects of fertilization treatments on fairways and rough for Sites A and B. The effects of fertilizer treatment (control [zero fertilizer], UNI, and PCU) on CH_4 flux were not significant on fairways and rough for both Sites A and B; therefore, the mean flux across fertilizer treatments was averaged by replication. The three plots without fertilizer treatment on the RNG sites were treated as replications for analysis.

Because of differences in measurement dates and fertilizer treatment times, data from Sites A and B were analyzed separately. An ANOVA was performed to test for the effects of section/turf type (fairway, rough, and RNG) and measurement time (DOY), along with corresponding interactions on daily methane flux for years 2011–2012 and 2012–2013 at Sites A and B using the PROC GLM model. Means for fairway, rough, and RNG sections on each measurement date were separated using protected LSD and presented in Fig. 1 for 2011–2012 and 2012–2013 at Sites A and B, respectively.

Correlation analysis was performed to evaluate the relationship between daily methane flux from individual sections/grass types and soil WFPS (calculated as described above) and measured air temperature using Pearson's correlation in PROC CORR (SAS 9.4, SAS Institute Inc., Cary, NC, USA).

Results

General soil properties

Soil texture was classified as sandy clay loam or sandy loam for both Site A and B study areas (Table 1). Soil organic carbon concentrations in the fairway and rough areas were 14 to 25 times higher than those in RNG areas (Table 1). Similar trends were observed for soil nitrate concentration. Soils in the RNG areas had very low organic carbon and NO_3-N concentrations (Table 1), likely due to the absence of fertilizer application, both before and during the experiment. Before the experiment began, soil NO_3-N concentrations at the fairway and rough areas of Site A were twice as high as those at Site B. Mean soil bulk density at the 0–10 cm depth was significantly greater in the fairway, rough, and RNG areas of Site A compared to those of Site B.

Methane flux

The measured methane fluxes over time are shown in Fig. 1a–c, and the average daily flux over each year for each site is summarized

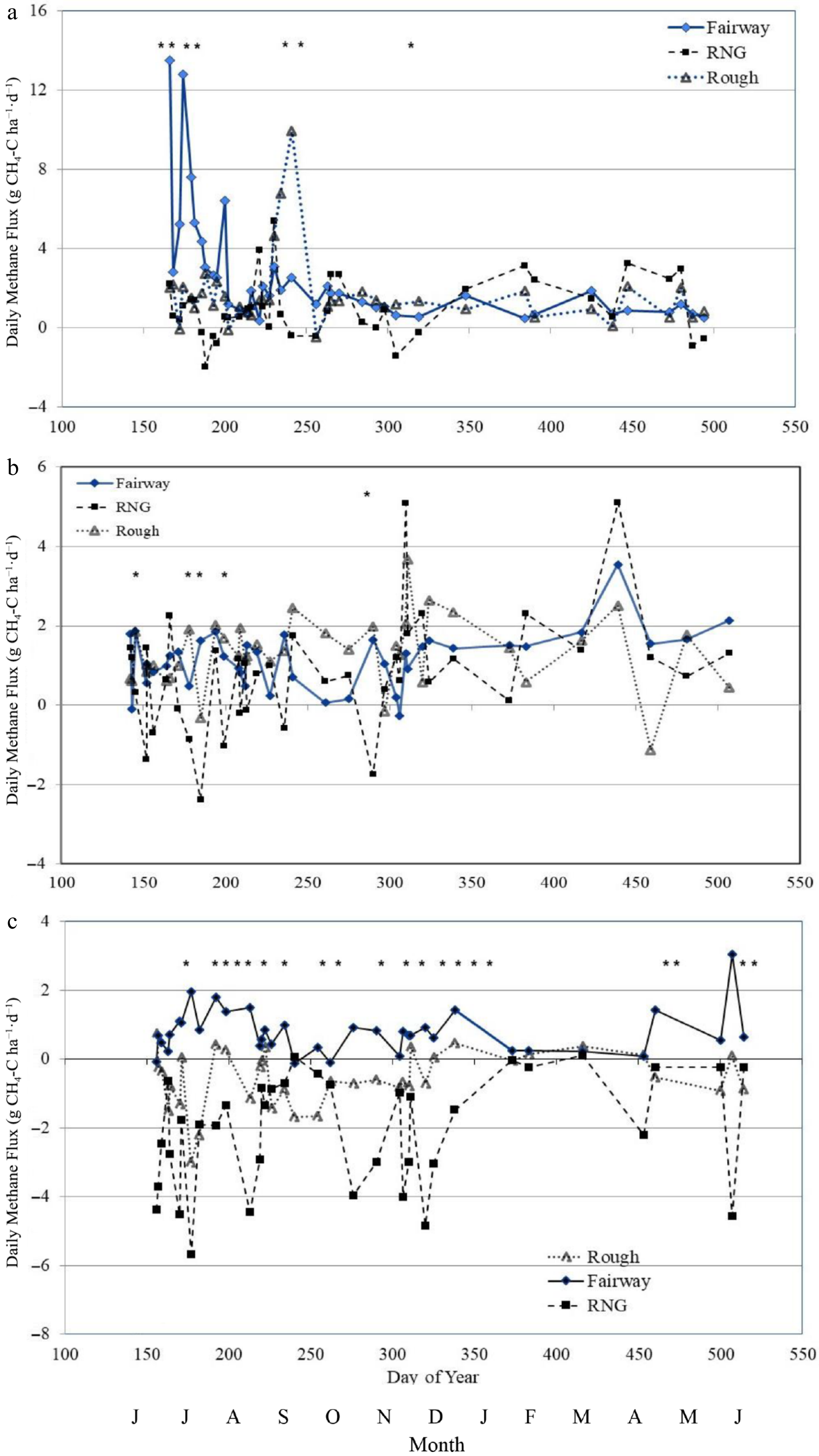

Fig. 1 Daily methane fluxes (g CH_4-C $ha^{-1}\cdot d^{-1}$) from fairway, rough, and restored native grass (RNG) soils during (a) 2011−2012, and (b) 2012−2013 at Site A, and during 2012−2013 at Site B (c). The X-axis shows the day of year (DOY) along with abbreviated months. Each methane emission data point represents the mean of three treatment replications. Asterisks (*) indicate significant differences (p < 0.05) among fairway, rough, and RNG treatments on individual measurement dates.

in Table 2. In both the figures and the table, a positive methane flux value indicates a methane source, whereas a negative value indicates a methane sink. While methane fluxes were not significantly different (p > 0.35) among different fertilizer

Table 2. Average methane flux (g CH_4-C $ha^{-1}\cdot d^{-1}$) from fairways, rough, and restored native grass (RNG) areas in 2011–2012, and 2012–2013.

Section/turf type	Mean methane flux (g CH_4-C $ha^{-1}\cdot d^{-1}$)		
	Site A		Site B
	2011–2012	2012–2013	2012–2013
Fairway	2.39 a A[Ψ]	1.18 ab B	0.78 a C
Rough	1.66 b A	1.33 a A	−0.53 b B
RNG area	1.18 b A	0.82 b A	−2.03 c B

[Ψ] Methane fluxes presented are the means of treatment replications over 38–39 measurement dates: from June 15, 2011 to May 8, 2012, and from May 21, 2012 to May 21, 2013 for Site A, and from June 4, 2012 to May 28, 2013 for Site B. Means followed by different lowercase letters are significantly different among the three turf types (Fairway, Rough, and RNG) within each season. Means followed by different uppercase letters are significantly different across seasons and sites within each turf type (Fairway, Rough, or RNG). A positive methane flux value indicates a methane source, whereas a negative value indicates a methane sink.

treatments, substantial spatial and seasonal variations of methane fluxes were observed, especially differences among turfgrass types/sections and changes over time.

Site A was located at the bottom of a hillside in a relatively flat area. Methane fluxes from the fairway and rough plots at Site A were predominantly positive throughout the study, indicating net methane emissions from the soil (Fig. 1a). High methane emissions were observed on the Site A fairway at the beginning of the experiment in 2011. Between June 15–21, 2011, fluxes from the fairway plots greatly exceeded those from the rough and RNG plots, with a peak flux rate of 13.7 g CH_4-C $ha^{-1}\cdot d^{-1}$ recorded on the first day of measurement. The differences in methane emissions between the fairway, rough, and RNG areas were significant during the first three measurement dates. These episodic emission events contributed significantly to the annual total, even though fluxes declined to below 4 g CH_4-C $ha^{-1}\cdot d^{-1}$ for the remainder of the season. These emission peaks coincided with periods of high soil water content as indicated by the high WFPS, exceeding 60% (Fig. 2a). During this time, WFPS on the fairway was significantly higher (> 60%) than in the rough (< 50%) and RNG areas (< 45%) (Fig. 2a). As the season progressed, methane fluxes from the fairway decreased considerably.

At Site A, rough plots exhibited lower methane emissions than fairway plots in June of the 2011–2012 season. Coincidentally, the water-filled pore space (WFPS) in the rough was also much lower than that of the fairway in June 2011. By August, both methane emissions and WFPS had increased in the rough plots. Soil WFPS and temperature in the 0–10 cm depth were generally similar between the rough and fairway plots from August 2011 to March 2012. However, on DOY 234 and 241, the rough plots showed higher methane emissions than the RNG plots, although differences from the fairway plots were not statistically significant. Throughout the 2011–2012 season, the RNG plots functioned either as a weak methane source or sink, depending on the measurement date. Their daily methane emissions were typically lower than those from the fairway or rough plots or not significantly different (Fig. 1a). On average, daily methane fluxes from the Site A fairway plots were 2.39 g CH_4-C $ha^{-1}\cdot d^{-1}$ during the 2011–2012 season, which was 31% and 51% higher than fluxes from the rough and RNG plots, respectively (Table 2).

During the second measurement season (2012–2013), methane flux from the fairway at Site A decreased compared to the season of 2011–2012 (Fig. 1a, b). However, year-to-year differences were not significant for the rough and RNG areas at Site A. On most dates, daily methane fluxes from the fairway and rough were similar, with both acting as sources of methane (Fig. 1b). The RNG area fluctuated between acting as a weak sink or source at the beginning of the 2012–2013 season but gradually became a source as the season progressed.

Although the year/season effects on methane emissions were not significant for the rough and RNG plots, the average daily methane emission from the Site A fairway declined by 51% from the 2011–2012 to the 2012–2013 season (Table 2). During the 2012–2013 season, the average daily methane flux from Site A fairway plots was 1.18 g CH_4-C $ha^{-1}\cdot d^{-1}$, which was not significantly different from that of either the rough or RNG plots. However, methane emissions from the rough plots were significantly higher

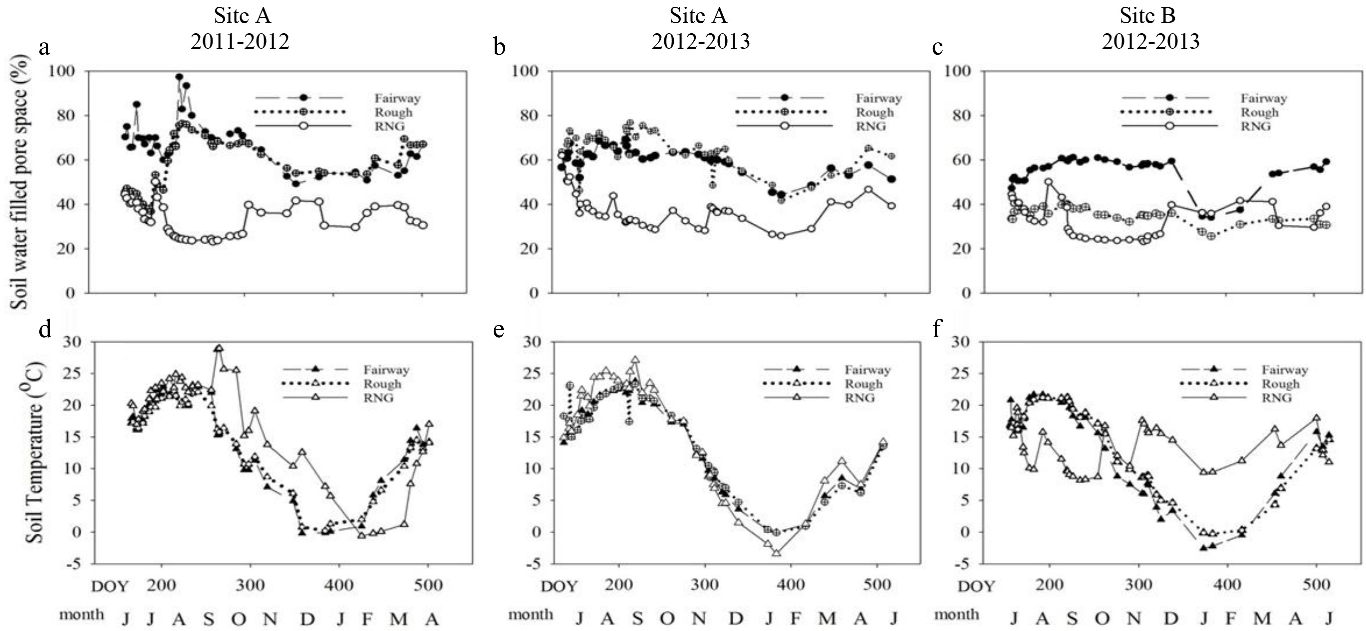

Fig. 2 (a)–(c) Soil water-filled pore space (%), and (d)–(f) soil temperature (celsius) of the top 10 cm of the soil profile are presented for each turfgrass section during the 2011–2012 and 2012–2013 seasons. The X-axis shows day of year (DOY) along with abbreviated months.

than those from the RNG plots during the 2012–2013 season (Table 2).

Site B was located on a man-made hill with a more porous (lower BD) soil medium than Site A (Table 1). During the 2012–2013 measurement period, methane fluxes from the RNG and rough plots at Site B were predominantly negative (Fig. 1c), indicating methane uptake and oxidation by the soil, making these areas general methane sinks. In contrast, methane fluxes from the fairway plots were mostly positive, suggesting they acted as a methane source. On many measurement dates, methane emissions from the fairway plots were higher than those from the RNG plots, while fluxes from the rough plots typically fell between the two on many measurement dates (Fig. 1c). However, during the winter months of January and February, methane fluxes were minimal and no significant differences in methane fluxes were observed among the fairway, rough, and RNG plots, indicating that both methane oxidation and methanogenesis activities were minimal during this period. At Site B, WFPS was generally lower than at Site A.

Therefore, in contrast to Site A, which acted as a methane source during the 2-year measurement period, methane fluxes measured from Site B plots were lower, with the rough and RNG areas functioning as methane sinks. The average daily methane fluxes at Site B during the 2012–2013 season were 0.78, −0.53, and −2.03 g CH_4-C $ha^{-1} \cdot d^{-1}$ for the fairway, rough, and RNG plots, respectively (Table 2).

Soil water-filled pore space, temperature, and nitrate

In addition to the baseline soil bulk density and WFPS measurements, which showed significant differences between Site A and Site B, as well as among turfgrass sections/types (Table 1), soil WFPS was monitored using soil moisture and temperature sensors, revealing both spatial and temporal variability (Fig. 2). Soil WFPS averaged 63.5% and 61.0% at the Site A fairway and rough, respectively (Fig. 2), which was 14% to 42% higher (p < 0.001) than Site B with WFPS averaging 54.6% and 35.2% at Site B fairway and rough, respectively (Fig. 2). The difference in soil WFPS at RNG between Sites A and B was much smaller, with average values of 35.7% and 34.6%, respectively. Soil temperatures for the fairways and rough were typically similar and tended to be lower than the RNG (Fig. 2), presumably due to less water input and evapotranspiration in the RNG plots.

There was also a difference in baseline soil NO_3-N between Sites A and B, with Site B having less than half of the NO_3-N concentration of Site A soil (Table 1). Drier soil conditions, as indicated by a lower WFPS at Site B, likely slowed N turnover rates, thereby lowering soil NO_3-N levels, which indicate the effects of topographic location and the associated increased drainage potential at Site B.

As shown in Fig. 3b, there was a clear distinction in the distribution of rainfall between the 2011 and 2012 growing seasons, with the spring of 2011 having early moisture and late-season drought, while 2012 had drought conditions all season long. The effects of the different precipitation patterns between 2011 and 2012 are somewhat reflected in the measured soil WFPS at Site A plots (Fig. 2), although daily irrigation buffered the effects of drought during 2012.

In 2011, early-season precipitation coupled with irrigation water applications created nearly saturated (> 65% WFPS) soil conditions at the Site A fairway (Fig. 2a), which resulted in spikes in methane emission due to methanogenesis. Soil was drier in the rough than in the fairway during 2011 (WFPS~50%), but WFPS in the rough

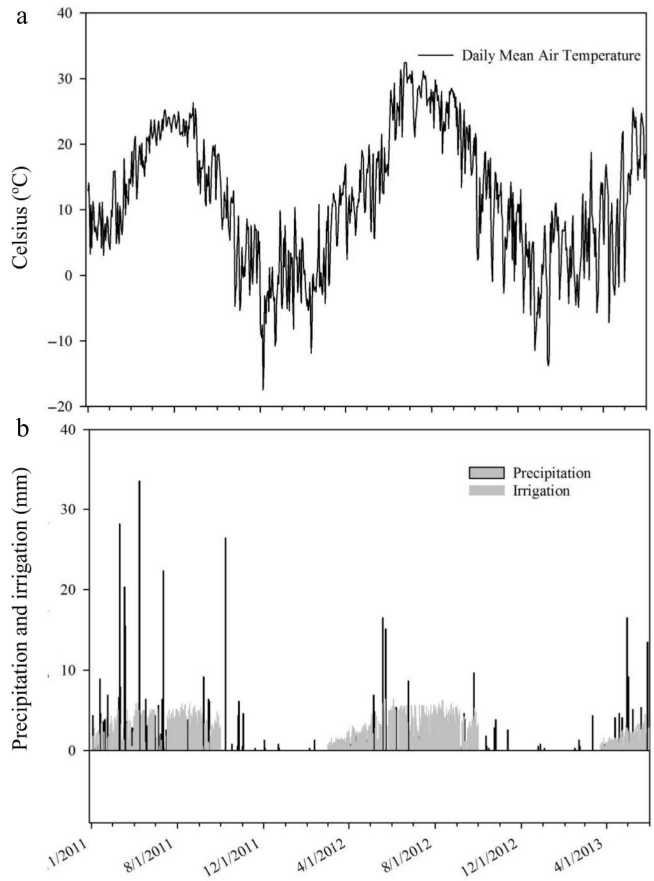

Fig. 3 (a) Daily mean air temperature; and (b) precipitation and irrigation water applications during the 2-year study period, 2011 to 2013.

increased to 65% in mid-July of 2011 (DOY 210), which resulted in a spike of methane emission later on (see Fig. 1a).

Due to the onset of drought conditions, extra irrigation water was supplemented to golf course turf during 2012 (Fig. 3). In 2012, soil WFPS averaged 60.0% and 63.9% at the Site A fairway and rough, respectively (Fig 2). Soil WFPS was significantly higher (p < 0.001) at Site A compared to the Site B fairway and rough (Fig. 2b, c). We assume that due to the higher location and lower bulk density of the well-drained Site B soil[26], high irrigation water applications did not increase soil WFPS as much as measured at Site A.

Correlations between daily methane flux, soil WFPS, and temperature for fairways, rough, and RNG sites are presented in Fig. 4. Significant correlations (p < 0.001) were found between methane flux and soil WFPS at the fairway and rough sites, with correlation coefficients of 0.37 and 0.59, respectively, explaining approximately 14% and 35% of the variation (Fig. 4a, b). A weak but significant correlation (r = 0.18, p < 0.05) was observed between soil temperature and methane flux at the fairway sites. However, correlations between soil temperature and methane flux were not significant for the rough and RNG areas.

Discussion

Comparing methane fluxes from different sections/grass types of the golf course

In this study, we found that methane emissions were highest in the intensively managed turf sections (fairways), intermediate in the

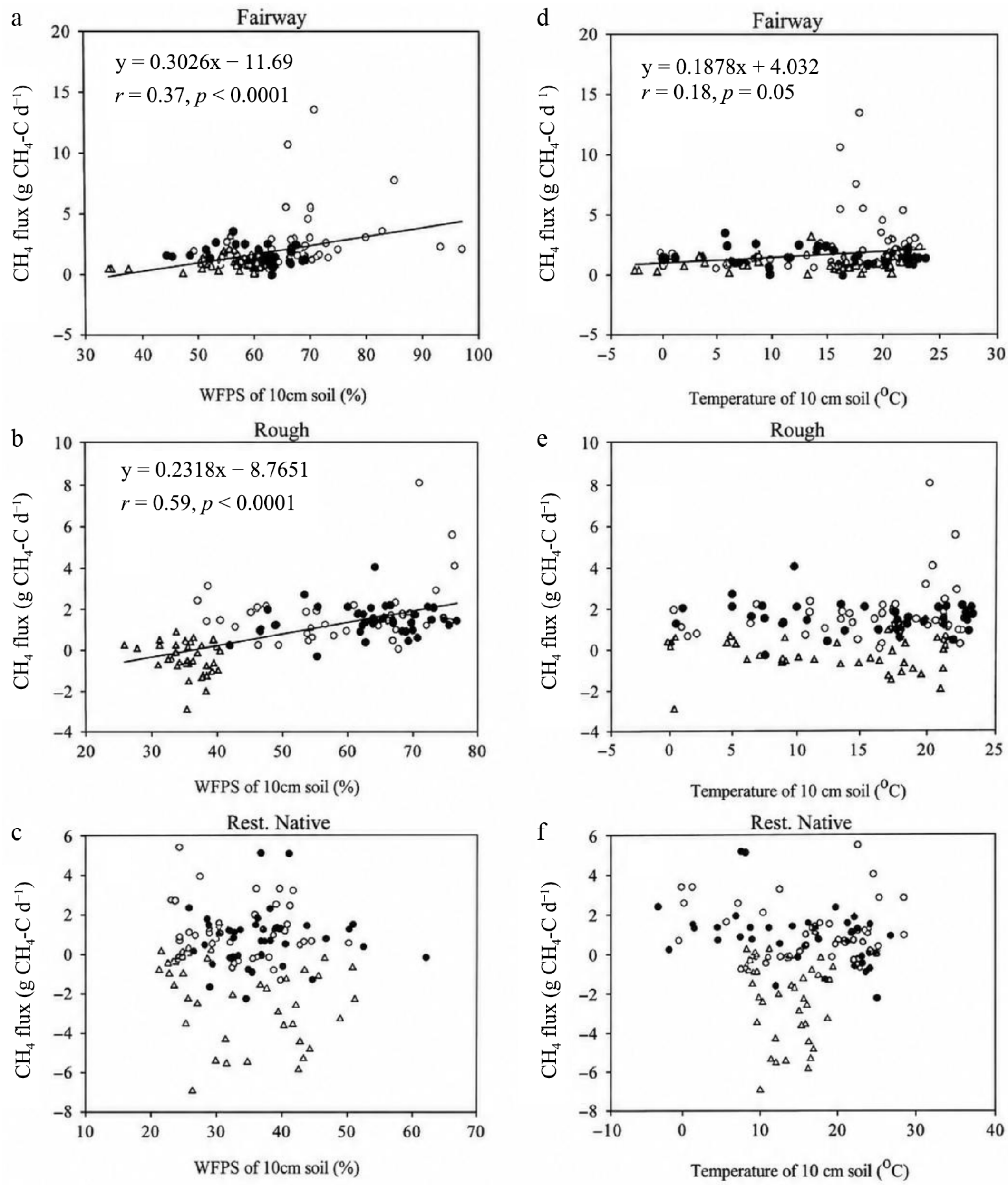

Fig. 4 (a)–(c) Correlations for soil methane flux and water-filled pore space (WFPS), and (d)–(f) temperature from soils at experimental turfgrass sites. *p*-values, *r*-values, and linear correlations are presented for each test location where significance exceeds the $p < 0.05$ level. Open and closed circles are from Site A soil in 2011 and 2012, respectively, and open triangles are from Site B.

rough areas (which were managed similarly to home lawns), and lowest in the RNG areas. Notably, the RNG at Site B acted as a methane sink, with an average daily flux of –2.03 g CH_4-C $ha^{-1}\cdot d^{-1}$. The absence or low levels of fertilizer inputs, low WFPS, and low soil compaction/high porosity, as indicated by low bulk density, may all have contributed to the low methane flux observed in RNG areas compared to fairways and rough. Kaye et al.[17] reported an average daily methane flux of –3.6 g CH_4-C $ha^{-1}\cdot d^{-1}$ from irrigated and fertilized lawns in the same region as this study. Mosier et al.[5] found that undisturbed prairie grasslands in Colorado acted as methane sinks, with fluxes ranging from –3 to –15 g CH_4-C $ha^{-1}\cdot d^{-1}$. Groffman & Pouyat[16] reported high methane uptake in native forests, with fluxes up to −16.8 g CH_4-C $ha^{-1}\cdot d^{-1}$. However, this high uptake capacity was almost entirely lost when native forests were converted to lawns. These findings suggest that methane oxidation is suppressed, and methane emissions are elevated when native grasslands or forests are converted to urban turfgrass, although fluxes can vary depending on turfgrass type and season.

Groffman & Pouyat[16] suggested a 'turfgrass effect' on soil methane oxidation potential, whereby methane uptake is severely inhibited in urban turfgrass systems. Several explanations likely account for the reduced capacity of soils under turfgrass to oxidize or absorb methane. (1) Irrigation and historical nitrogen fertilizer applications have been shown to reduce CH_4 uptake in grasslands[5]. Groffman & Pouyat[16] also speculated that land-use changes associated with urban and suburban development may indirectly influence ecosystem processes by increasing atmospheric nitrogen deposition and CO_2 concentrations. These environmental changes can affect methane uptake in both natural and anthropogenically altered components of urban landscapes[27]. (2) Human activities and traffic associated with urban turfgrass and its maintenance, especially on sports fields such as golf courses, can lead to increased soil compaction. Compacted soils tend to retain more water and reduce gas exchange, which can suppress microbial activity and, in turn, reduce methane uptake and oxidation while increasing methane production. (3) Turfgrass systems contribute to soil quality through increased organic carbon inputs[28−31]. Methanogenic bacteria produce methane through anaerobic decomposition of organic matter[32], and the availability of labile carbon is often the limiting factor in methane production in soils[33]. High organic matter inputs from turfgrass may stimulate methane emissions by providing abundant carbon substrates for methanogens. Lu et al.[34] also found that enrichment of organic carbon in the root zone was positively correlated with methane emissions in submerged rice fields.

Factors contributing to temporal and spatial variations

Temporal variations in methane flux were observed on the golf course (Fig. 1). For instance, on the fairway at Site A, the maximum methane emission measured was 13.7 g CH_4-C $ha^{-1}·d^{-1}$, occurring in June 2011 following significant rainfall (Figs 1 and 3). However, the 2-year average daily flux for the Site A fairway was only 1.78 g CH_4-C $ha^{-1}·d^{-1}$ (Table 2). Law et al.[18] also reported that methane flux rates in turfgrass systems tend to vary more over time than other greenhouse gas fluxes. The measured daily CH_4 flux rate in this study was consistent with previous findings for turfgrass systems[35,36].

Spatial variations in methane flux were evident from the differences among turfgrass types and between Sites A and B. This study found that Site A, located at the base of a hillside in a relatively flat area, exhibited significantly higher soil bulk density, higher WFPS, and greater methane emissions than Site B, which was situated on an elevated area. This suggests that topography had a significant influence on methane flux. At Site B, the rough plots retained some capacity for atmospheric CH_4 uptake and oxidation during several periods of the year-long measurements (Fig. 1c). Microbial consumption may have been enhanced in the well-drained upland rough plots due to lower WFPS. In addition, higher porosity at Site B increased soil gas diffusivity, which is an important control on CH_4 oxidation rates as observed by Del Grosso et al.[37]. Aronson et al.[38] found that long-term drainage differences between the sites likely drove the differences in CH_4 flux. By analyzing methane flux across complex terrain in a Montana forest, Kaiser et al.[39] demonstrated that environmental factors associated with topography can help predict CH_4 flux. They found that locations with volumetric water content (VWC) below 38% acted as methane sinks, with uptake increasing as VWC decreased. Conversely, sites with VWC above 43% showed net CH_4 emissions, while intermediate VWC levels resulted in near-zero flux. The RNG area at Site B maintained a more consistent methane oxidation capacity, with an annual methane uptake rate of −2.03 g CH_4-C $ha^{-1}·d^{-1}$. However, this rate was lower than those reported for native grasslands in Northern Colorado, which ranged from −3 to −15 g CH_4-C $ha^{-1}·d^{-1}$[5]. Undisturbed soils typically absorb more CH_4 than they emit[40]. The lower methane oxidation rates at Site B's RNG area were likely due to soil disturbance during golf course construction, even though the area was later reseeded with native grasses. In general, human-induced soil disturbance is known to significantly reduce methane oxidation capacity[6,37,41].

The significant positive correlations between daily methane flux and WFPS on fairways and rough in this study indicate that irrigation and the resulting high soil water content in these intensively managed areas substantially inhibit methane uptake and oxidation. Soil water content affects methane flux in several ways. First, as described in the investigations of Del Grosso et al.[37] and Aronson et al.[38], methane diffusion into soil is severely inhibited during periods of high soil water content. Soil WFPS at both the fairway and rough at Site A were often between 50% and 70% throughout the 2-year study. Soil water content can be the dominant factor controlling soil methane emissions[42], and is generally regarded as the strongest predictor of methane flux[38]. Secondly, high soil WFPS interacts with compaction to inhibit soil CH_4 oxidation. In this study, measurement of soil bulk density indicated that compaction at the lowland Site A fairway led to significantly greater WFPS ($p < 0.0001$) compared to the upland Site B fairway (Table 1). High WFPS enhances compaction under traffic, and compaction subsequently leads to even higher WFPS by reducing macro-porosity and increasing water holding capacity. Soil compaction and high WFPS reduce soil aeration, thereby significantly limiting the potential for methane oxidation and increasing methanogenesis.

The significant positive correlations between daily methane flux and soil temperature observed at the fairway plots might have resulted from the effects of temperature on microbial activity and the decomposition of labile carbon substrates[43,44]. In France, Künnemann et al.[19] found that lawns with trees were a sink for methane because tree shading reduced soil temperature and, consequently, soil respiration. While methane efflux was higher at the Site A RNG plots compared to those at the Site B location, neither soil WFPS nor soil temperature was related to methane flux at the RNG sites. Due to the minimal irrigation and fertilization inputs, RNG methane flux could be impacted by many variables, and correlations for any one variable may be weak due to the complex interactions of abiotic soil properties that affect greenhouse gas emissions[33,42].

Nitrogen effects

Soil at RNG areas had very low NO_3-N concentrations (Table 1), likely due to the absence of fertilizer application to RNG areas. Soil NO_3-N concentrations prior to the start of the experiment were approximately twice as high at Site A fairway and rough compared to those at Site B. Interestingly, methane emissions of Site A fairway, rough, and native plots were higher than those from the respective plots at Site B (Fig. 1 and Table 2). Despite soil NO_3-N concentration likely influencing soil CH_4 flux potentials[5], we found a lack of fertilizer treatment effect between the fertilized plots and the control plots (zero fertilizer), with control plots having similar methane emissions as fertilized plots that received 150 kg N $ha^{-1}·yr^{-1}$ in fairway and rough plots. Previous research demonstrated that nitrogen fertilizer is the most important nutrient addition for maintaining turf quality[45], and fertilizer additions strongly inhibit soil CH_4 oxidation capacity[5,46−48], with oxidation being negatively correlated with soil mineral N concentrations[3]. This has been related to the complex interactions and competition of nitrifying and oxidizing soil microbes[46,47]. Mosier et al.[9] found that cultivation and fertilization of cropping soils inhibited up to 90% of CH_4 consumption compared to native grasslands. However, in this study, we found no fertilizer treatment effect on either fairway or rough plots. This trend was persistent over the 2-year study period at both Sites A and B, and in both fairway and rough plots. This result is similar to the findings of Riches et al.[49] for sports fields, where methane flux did not change before and after nitrogen applications. Livesley et al.[35] found that there was no difference between fertilized and non-fertilized treatments in a home lawn research setting. Previous research indicated that historically high N fertilization and its inhibition on methane oxidation may prevail long after the soil N concentrations have declined to levels typical of unfertilized soils[5], and may cause permanent shifts in soil microbial populations[50]. In this study, fertilizations that occurred since the establishment of the golf course, but prior to the start of this study, may have inhibited the methane oxidation capacity of both fairway and rough in the plots that received no fertilizer inputs during the study years.

This study demonstrated that methane flux varied spatially and temporally across golf course turf sites, which acted simultaneously as methane sinks and sources, depending on WFPS, soil compaction and aeration, and other environmental variables. Management practices can greatly affect soil water content, soil compaction/aeration, and N availability. The best management practices that implement site-specific irrigation management, reduce compaction, promote soil aeration, avoid excessively high WFPS, and minimize water input could reduce CH_4 emissions in urban turfgrass systems. Consistent with these approaches, survey data indicate that water use in the golf course industry has declined, with many superintendents adopting deficit irrigation management (< 100% evapotranspiration replacement)[51].

Conclusions

The study supported our hypothesis that methane fluxes across different turfgrass types of a golf course are affected by varying levels of management intensity. Factors influencing gas diffusivity, especially WFPS and porosity, influence methane production and absorption. However, there was no significant effect of N fertilizer treatments, although fertilizations that occurred since the establishment of the golf course but prior to the start of this study may have inhibited the methane oxidation capacity of both fairway and rough. In contrast, N_2O emissions measured at Site A were strongly influenced by nitrogen fertilization and fertilizer type[20]. This study demonstrated that turf management intensity affected methane emissions: fairway plots were a source of methane, rough plots were either sources or weak sinks of methane. Restored native grass areas could serve as a sink or weak source depending on environmental factors. Soil WFPS appeared to be the dominant soil factor affecting methane oxidation potential in golf course turfgrass, with low WFPS/low compaction soils having lower emissions and increased soil oxidation capacity. Our study concludes that intensely managed urban grasslands, such as golf course fairways, may have greater methane emissions than previously considered, whereas restored native areas may serve as a methane sink. This study suggests that best management practices that implement site-specific irrigation management, reduce compaction, and promote soil aeration could reduce CH_4 emissions in urban turfgrass systems.

Author contributions

The authors confirm contributions to the paper as follows: study conception and design, methodology: Qian Y, Follet RF; data collection: Gillette KL, Follet RF, Qian Y; analysis and interpretation of results, draft manuscript preparation: Gillette KL, Qian Y. All authors reviewed the results and approved the final version of the manuscript.

Data availability

The data generated or analyzed during this study are included in this published article and are available from the corresponding author on reasonable request.

Acknowledgments

The authors gratefully acknowledge the important contributions of Dr. Stephen Del Grosso, Soil Scientist/Ecologist at USDA-ARS and Research Soil Scientist in the Natural Resource Ecology Laboratory at Colorado State University for his constructive review and valuable input. We would like to thank Sarah Wilhelm and Mary Smith for technical support.

Conflict of interest

The authors declare that they have no conflict of interest.

Dates

Received 26 January 2026; Revised 12 May 2026; Accepted 19 May 2026; Published online 5 August 2026

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