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Dynamics of soil-derived greenhouse gas emissions from shelterbelts under elevated soil moisture conditions in a semi-arid prairie environment

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Abstract Soil moisture is known to be a major control of greenhouse gas (GHG) emissions from agricultural soils. However, there is little data regarding GHG exchange from the organic matter-rich soils characteristic of shelterbelts—especially under elevated soil moisture conditions. In the present study, we quantified CO_2, CH_4 and N_2O fluxes from shelterbelts under elevated soil moisture (irrigated) and semi-arid (rainfed) conditions. Studies were carried out at the Canada-Saskatchewan Irrigation Diversification Centre (CSIDC) near Outlook, Saskatchewan. Non-steady state vented chambers were used to monitor soil GHG fluxes from three shelterbelts in 2013 and 2014. The shelterbelts consisted of a single row of caragana with a north–south orientation and a single row of Scots pine with either a north–south or east–west orientation. Each shelterbelt was divided into two areas based on whether or not it received irrigation. During the 2-year study period, N_2O emissions from the irrigated shelterbelts (IR-SB) (0.93 kg N_2O-N ha^{-1}) were significantly greater than those from the rainfed shelterbelts (RF-SB) (0.49 kg N_2O-N ha^{-1}). Soil CH_4 oxidation was significantly lower in the IR-SB compared to the RF-SB (−0.85 and −1.20 kg CH_4-C ha^{-1}, respectively). Irrigation activities stimulated CO_2 production/emission in 2014, but had no effect on CO_2 emissions during the much drier 2013 season. Correlation analyses indicate a strong dependence of CO_2 and CH_4 fluxes on soil moisture in both IR-SB and RF-SB sites. There was a significant relationship between N_2O emissions and soil moisture for the IR-SB sites in 2013; however, no such relationship was observed in either the IR-SB or RF-SB sites in 2014. Our study suggests that changes in precipitation patterns and soil moisture regime due to climate change could affect soil-atmosphere exchange of GHGs in shelterbelts; however, elevated soil moisture effect on GHG emissions will depend on the availability of N and C in the shelterbelts.

Keywords Shelterbelt · Agroforestry · Soil moisture · Greenhouse gas · N-fixation · Irrigation · Nitrous oxide · Methane · Carbon dioxide

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Introduction

Afforested marginal soils have the potential to exchange significant amounts of the greenhouse gases (GHG) CO_2, CH_4, and N_2O (Peichl et al. 2010), which are produced and/or consumed via microbial processes in the soil. However, the amount of soil-atmosphere gas exchange depends largely on soil physical factors—with soil moisture and temperature being important drivers of both the production/consumption and atmospheric exchange of GHGs (Smith et al.

2003). Indeed, soil water content and temperature have a strong influence on the activity of both the soil microbial community and plant roots. Gas diffusivity, which varies inversely with soil water content affects soil aeration and the movement of gases in the soil, and thus indirectly controls the capacity of the soil to produce or consume CO_2, N_2O and CH_4 (Smith et al. 2003). Whereas CO_2 fluxes from afforested soils are usually much larger than the fluxes of CH_4 and N_2O, the latter two GHGs have global warming potentials (GWP) that over a 100-year timeframe are 25- and 298-times greater respectively, than that of CO_2 (IPCC 2007), thus magnifying their potential impact on global radiative forcing.

Shelterbelts, consisting of one or more rows of trees and/or shrubs planted to provide protection from the wind, have been used for centuries to regulate environmental conditions in agricultural landscapes and provide a variety of economic, social, and environmental benefits that are valued by landowners and society (Mize et al. 2008). More recently, shelterbelts have been recognized for their potential to offset increasing concentrations of atmospheric CO_2 by storing photosynthetically fixed carbon (C) in woody biomass and in the soil (Kort and Turnock 1999; De Brauw 2006; Sauer et al. 2007). Shelterbelts also have been reported to mitigate N_2O emissions and enhance CH_4 uptake relative to adjacent agricultural fields (Amadi et al. 2016a). The effect of shelterbelts in C sequestration and the mitigation of agricultural GHGs is due mainly to the high rates of C accrual in biomass and soil through litter fall, entrapment of windblown sediments, and modification of the local microclimate and root activity (Amadi et al. 2016a).

The role of soil water as a major control on GHG emissions from soils in the Canadian Prairies has been well documented (Hao et al. 2001; Liebig et al. 2005; Ellert and Janzen 2008; Sainju et al. 2012); however, relatively little research has focused on understanding the dynamics of GHG emission from the organic matter-rich shelterbelt soils under elevated soil moisture conditions. In the agricultural landscape, it is not uncommon to encounter soil and/or management factors that promote elevated soil moisture conditions in the soil. For example, the short-term flooding of depressional areas in landscapes with variable topography (Wang and Bettany 1997) or the application of irrigation water in shelterbelts established along the borders of cropped fields. Elevated soil moisture coupled with warm temperatures and substrate availability favour soil microbial activity, which in turn, may alter the dynamics of soil GHG production/consumption/emission (Dobbie et al. 1999; Smith et al. 2003).

Nitrous oxide is produced in soils as a result of naturally occurring microbial processes; namely, nitrification and denitrification. Nitrifying bacteria are active under aerobic conditions and produce N_2O during the oxidation of ammonium (NH_4^+) to nitrate (NO_3^-). Denitrification, on the other hand, produces N_2O as an intermediate during the reduction of NO_3^- under anaerobic conditions. As a result, N_2O emission are positively correlated with factors that influence microbial activity, including nitrogen availability (Bouwman 1996; Dobbie et al. 1999), soil water content (Corre et al. 1996; Dobbie et al. 1999), aeration status (Linn and Doran 1984), and soil temperature (Dobbie and Smith 2003; Yates et al. 2007). These factors are both spatially and temporally variable; consequently, soil-derived N_2O emissions also are inherently variable in both space and time (Yates et al. 2006, 2007). Indeed, soil N_2O emission patterns are often characterized by small areas ('hot-spots') and brief periods ('hot moments') that account for a high percentage of the total emissions (Groffman et al. 2009; Braker and Conrad 2011; Butterbach-Bahl et al. 2013).

The foregoing discussion suggests that the magnitude of N_2O emissions from shelterbelt soils under elevated soil moisture conditions may rely mainly on soil concentrations of available N (i.e., NO_3^- and NH_4^+). For example, shelterbelts composed mainly of N_2-fixing trees (e.g., *Caragana arborescence*) may contain more available soil N relative to shelterbelts composed of non-N_2 fixing trees (e.g., *Pinus sylvestris* L.) (Vlassak et al. 1973; Albrecht and Kandji 2003; Peichl et al. 2010; Moukoumi et al. 2013). Consequently, under elevated soil moisture conditions, soil-derived N_2O emissions under N_2-fixing trees may be greater than those from soils with a more limited supply of available N (Malhi et al. 1990; Peichl et al. 2010).

Upland soils are natural sinks for atmospheric CH_4 due to oxidation processes facilitated by methanotrophic microbes under aerobic soil conditions (Suwanwaree and Robertson 2005; Fowler et al. 2009). However, short-term increases in soil water content may increase soil CH_4 production, thereby reducing the soils' capacity for CH_4 uptake (Liu et al. 2006; Sainju et al. 2012). For example, in a study quantifying methane emissions from Canadian prairie and forest soils under short term flooding conditions, Wang and Bettany (1997) found increased CH_4 emission rates shortly after snowmelt in the spring and from low slope positions after rainfall in the summer. Reductions in CH_4 uptake also have been linked to increases in soil N availability (Bronson and Mosier 1994; Sainju et al. 2012) due to competition between ammonia and methane oxidizing microbial communities (Hütsch et al. 1993). Consequently, shelterbelts that are characterized by increased available soil N (as is the case with N_2-fixing tree species) may have a lower capacity for CH_4 uptake, a condition that may be exacerbated under elevated soil moisture conditions.

Soil water content, particularly water-filled pore space (WFPS), also can influence rates of CO_2 emission. In dry soils, soil respiration may be limited by the slow diffusion of soluble C substrates in thin water films (Davidson et al. 2006). On the other hand, the addition of water to shelterbelt soils (either as precipitation or irrigation) can elicit substantial increases in total respiration—reflecting enhanced decomposition of the LFH layer and an increase in substrate availability (Davidson et al. 2006; Cisneros-Dozal et al. 2007). However, under saturated conditions, a large proportion of the pores are filled with water, thus restricting soil aeration and respiration, and slowing the diffusion of CO_2 to the soil-atmosphere interface. Consequently, CO_2 fluxes generally decrease under saturated conditions, though not necessarily by as much as when lack of water is the limiting factor (Smith et al. 2003).

There exists more than 60,000 km of shelterbelts in Saskatchewan alone (Amichev et al. 2014), and many more throughout the remainder of the Canadian agricultural landscape. Natural and anthropogenic modifications of soil hydrology in shelterbelt ecosystems may significantly alter the rates of production and consumption of GHGs. Thus, it is relevant to investigate the effect of changes in soil moisture status on the dynamics of GHG emissions from soils occupied by shelterbelts. Furthermore, climate models have predicted that most areas in Temperate North America will probably experience the greatest alterations in precipitation under changing climate scenarios (IPCC 2007); therefore, an accurate assessment of the impact of elevated soil moisture on GHG exchange in agroecosystems—including shelterbelts—is imperative. In the present study, we monitored GHG emissions from shelterbelts that received irrigation water and compared them against non-irrigated sections of the same shelterbelts in the Prairie ecozone of Saskatchewan, Canada. The objective of the study was to quantify and compare fluxes of CO_2, CH_4 and N_2O from shelterbelts under irrigated conditions to those from non-irrigated (rainfed) shelterbelts.

Materials and methods

Study site

This study was carried out at the Canada-Saskatchewan Irrigation Diversification Centre (CSIDC), in Outlook, SK. The CSIDC is located in the moist mixed grassland Ecoregion of Saskatchewan, Canada (51°29′N, 107°03′W), with an average annual air temperature of 12.5 °C and cumulative annual precipitation of 278 mm during the April to October sampling season (based on 1981–2010 climate norms; Environment Canada, 2015). Average annual air temperatures during 2013 and 2014 seasons were 16.1 and 14.0 °C, respectively; cumulative annual precipitation during 2013 and 2014 sampling seasons were 180 and 326 mm, respectively. Soils at the site are classified as Orthic Dark Brown Chernozems, a mix of Asquith and Bradwell Association, with moderately sandy loam textures. They consist of well drained soils formed mainly in wind deposited sands and loamy lacustrine materials on a slightly undulating topography (Soil Classification Working Group 1998).

The study included three shelterbelts: a single row of caragana (C. *arborescence*) running north to south along the western border of a field planted to wheat in 2013 and canola and soybean in 2014 (C_N); a single row of Scots pine (*P. sylvestris* L.) running north to south along the western border of a field planted to soybean in 2013 and wheat in 2014 (SP_N) and a single row of Scots pine running east to west along the southern border of the same field (SP_E). Each shelterbelt consisted of an irrigated (IR) section and a rainfed (RF) section. Details of shelterbelt design and the characteristics of each shelterbelt are summarized in Table 1.

Table 1 Characteristics of shelterbelts at the three study sites at CSIDC Outlook, Saskatchewan, Canada

Site characteristic	Tree species		
	Caragana	Scots pine	Scots pine
Shelterbelt orientation	North–south	North–south	East–south
Age at start of this study (years)	36	18	21
Number of rows	1	1	1
Mean tree height (m)	6	11	12.5
Mean DBH (cm)[a]	6.8	27.7	30.5
Tree spacing (m)	1	2.5	2.5
Total length of shelterbelt (m)	750	200	435
Soil C (0–30 cm) (Mg ha^{-1}/ %)	93.0/2.60	57.1/1.55	60.5/1.66
Soil N (0–30 cm) (Mg ha^{-1}/ %)	7.60/0.21	4.56/0.12	4.64/0.13
Soil NH_4-N (μg N g $soil^{-1}$)[c]	11.0 (1.70)[b]	7.0 (1.50)	7.10 (1.80)
Soil NO_3-N (μg N g $soil^{-1}$)	6.50 (1.40)	2.90 (1.0)	3.20 (1.10)

[a] DBH represents diameter at breast height

[b] Numbers in parenthesis represent standard deviation

[c] Average of values of available N measured on three separate dates (i.e. July 2013, June 2014 and October 2014) during the study period

The irrigated sections of the Scots pine shelterbelts (SP_N-IR and SP_E-IR) received a total of 62.5 and 37.5 mm of irrigation water in 2013 and 2014, respectively; while the irrigated section of the caragana shelterbelt (C_N-IR) received a total of 50 and 75 mm of additional water in 2013 and 2014, respectively. The rainfed sections of the shelterbelts (C_N-RF, SP_N-RF and SP_E-RF) served as a reference, representing the status of GHG exchange under normal precipitation regimes.

Greenhouse gas sampling and analysis

During the fall of 2012, the bases of four rectangular (22 cm wide × 45.5 cm long × 15 cm tall) gas chambers were installed in both the irrigated and rainfed sections of each of the shelterbelts. The chamber bases, made of 0.6-cm thick poly methyl methacrylate (PMMA) were installed to a depth of 5 cm along a transect at the center of each shelterbelt at a spacing of ca. 20-m between chambers. Gas samples were collected by attaching a flux chamber to the base and withdrawing 20-mL gas samples as soon as the chamber was in place at time zero (t_0) and again 20 min later (t_{20}) (see Czóbel et al. 2010; Raut et al. 2014; Amadi et al. 2016b). In addition, ambient air samples (n = 12) were taken on each sampling day and the concentration of GHGs in the ambient air were used as a check on the t_0 samples.

The gas chambers were made of 0.6 cm thick PMMA wrapped with a reflective insolation and had a headspace volume of 10 L and a surface area of 1000 cm^2. Upon deployment, close-cell polyolefin foam gaskets (1 cm thick × 1.2 cm wide) were secured to the underside of the chamber lids to seal against the top edge of chamber bases. Gas chambers were vented with a clear flexible vinyl tube (4.8 mm i.d.) attached through an elbow fitting to the cover (Hutchinson and Livingston 2001; Rochette and Bertrand 2008). The sampling port consisted of a silicone septum (9.5 mm o.d.) secured by a nylon bolt with a lengthwise opening, which served as a syringe guide (Amadi et al. 2016a).

All green vegetation within the chambers was removed prior to gas sampling. Gas samples were collected using a 20-mL polypropylene syringe (Monoject™, Luer lock fitting) fitted with a 25-gauge needle; injected into pre-evacuated 12-mL Exetainer® vials (LabCo Inc., High Wycombe, UK) fitted with butyl rubber stoppers (Rochette and Bertrand 2003); and returned to the Department of Soil Science at the University of Saskatchewan in Saskatoon for analysis.

Gas sampling was initiated in spring 2013, at the start of the spring thaw, and continued through to soil freeze-up in the late fall in both 2013 (May 10th–October 9th) and 2014 (April 22nd–October 15th). Disturbance of the soil and litter deposits underneath the shelterbelts was minimized during chamber installation and all chamber bases remained in place throughout the 2-year sampling period. Gas sampling occurred twice per week during the spring snowmelt period, after which sampling intensity was reduced to once per week throughout the summer and then to once every two weeks during the fall. During gas sampling, soil moisture and temperature measurements were taken directly beside the gas chambers and at a depth of 10 cm. Soil temperature was measured using a stem-style digital thermometer (Reed PS100, Brampton, ON) whereas soil moisture measurements were taken using a digital soil moisture meter (HydroSense, Campbell Scientific, Inc., Logan, UT).

At the end of each sampling day, gas samples were transported to the laboratory for analysis using a gas chromatograph (Bruker 450 GC, Bruker Biosciences Corporation, USA) equipped with a thermal conductivity detector (TCD), flame ionization detector (FID) and electron capture detector (ECD) for CO_2, CH_4 and N_2O measurements, respectively (Farrell and Elliot 2008). Gas samples were introduced into the chromatograph using a CombiPAL auto-sampler (CTC Analytics AG, Switzerland) and data processing was completed using the Varian Star Chromatography Workstation (ver. 6.2) software. Daily gas fluxes (F_{GHG}) were calculated using Eq. (1):

$$F_{GHG} = \Delta C \frac{V \cdot k_t}{A} \tag{1}$$

where F_{GHG} is the gas flux at t_0 (g m^{-2} d^{-1}); ΔC = change in concentration (g L^{-1} min^{-1}) measured during the 20-min chamber deployment; V = volume of the flux chamber (L); A = surface area enclosed by the chamber (m^2); and k_t = time-constant 1440 min d^{-1}).

Soil sampling and analysis

During June 2014, four soil cores (3.2 cm i.d.) were collected from an area immediately adjacent to each gas chamber (0–30 cm bgs) in the irrigated and rainfed sections of the shelterbelts. The soil samples were air dried, crushed and ground with a rolling pin to break aggregates; all visible roots were removed and a subsample of soil (~150 g) was passed through a 2-mm sieve. A 20-g subsample of the air dried soil was placed on a ball grinder for 5 min to create a fine powder (<250 µm) for Total N (TN) and soil organic carbon (SOC) analyses. Bulk density samples were collected using a hand-held core sampler (i.d. = 5.4 cm, height = 3 cm), which were then weighed and dried at 105 °C for 24 h. Soil samples (0–30-cm) collected in July 2013, June 2014, and October 2014 were used to monitor soil nitrate N (NO_3-N) and ammonium N (NH_4-N) concentrations, and were treated as described above prior to analysis.

Soil organic C was determined using a LECO C632 Carbon analyzer (Wang and Anderson 1998), following a 12 M HCl pretreatment to remove all inorganic C. Total N was determined by dry combustion using LECO TruMac CNS analyzer (Figueiredo 2008). Total inorganic N (NO_3^--N + NH_4^+-N) were determined using 2.0 M KCl extraction (Maynard et al. 2008) and analyzed colorimetrically (Technicon Autoanalyzer; Technicon Industrial Syatems, Tarrytown, NY, USA). Soil particle size was determined using a modified pipette method (Indorante 1990). Measurement of soil pH in water (1:1 paste; Hendershot et al. 2008) was completed using a Beckman 50 pH Meter (Beckman Coulter, Fullerton, CA, USA).

Statistical analysis

Differences in gas exchange and soil properties in the irrigated and rainfed sections of the shelterbelts were analyzed using the PROC MIXED procedure in SAS (ver. 9.4) for an RCBD with treatments (irrigation vs. rainfed) as a fixed effect and block (shelterbelts) as a random effect (SAS Institute Inc. 2013). Degrees of freedom were approximated by the method of Kenward-Roger (ddfm = kr). Treatments were compared using the Fisher Protected Least Significance Difference (LSD) method. Significance was declared at $P < 0.05$.

Results

Soil and environmental conditions

Average annual air temperatures at the CSIDC site were warmer than the long-term norms in both 2013 and 2014 (+3.6 and +1.5 °C, respectively). At the same time, cumulative annual precipitation at the site was about 35% lower (−98 mm) than normal in 2013, and 17% greater (+48 mm) than normal in 2014. The addition of irrigation water, increased the total water input in the caragana site (C_N-IR; Fig. 1a) by 28% in 2013 and by 23% in 2014. Likewise, irrigation in the Scots pine shelterbelts (SP_N-IR and SP_E-IR; Figs. 2a,

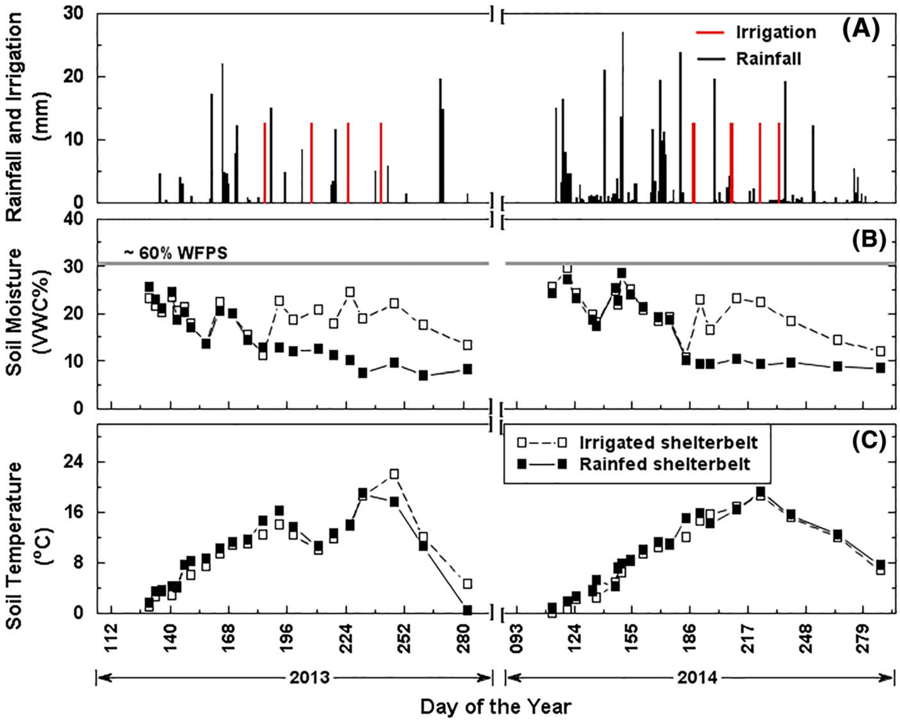
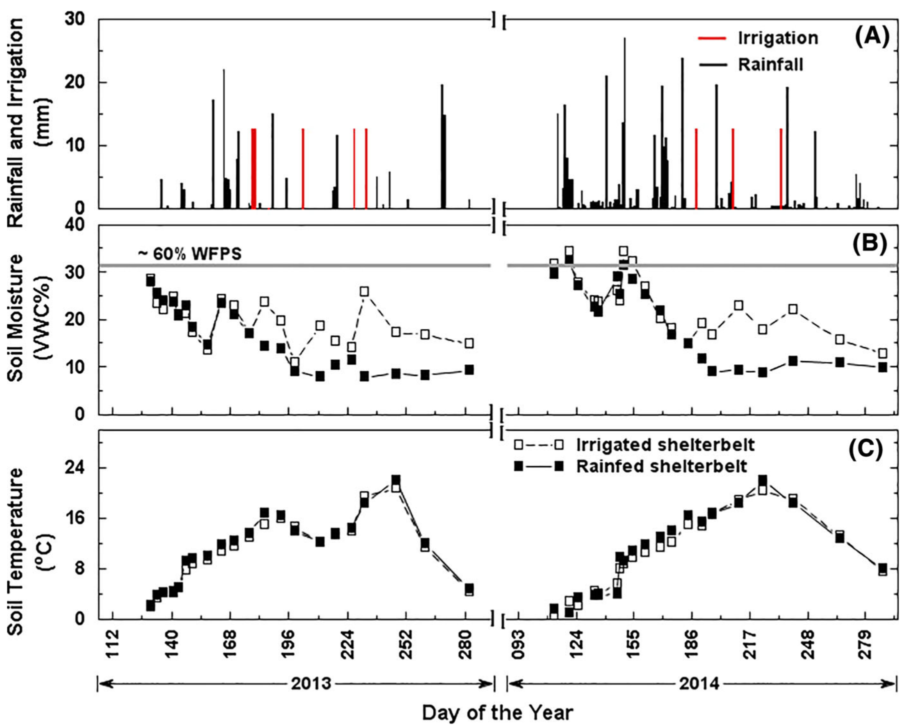


Fig. 1 Rainfall plus irrigation (**a**), soil water content (**b**) and soil temperature (**c**) measured at the Caragana (N–S) shelterbelt during 2013 and 2014. Volumetric Water Content (VWC%; averaged across the 0–10 cm depth) and soil temperature (°C; measured at a depth of 10 cm) were measured in both irrigated (*open squares*) and rainfed (*filled squares*) sites of the shelterbelt. The *grey bar* on Panel **b** represents the approximate range where water filled pore space (WFPS) is at 60%

Fig. 2 Rainfall plus irrigation (**a**), soil water content (**b**) and soil temperature (**c**) measured at the Scots pine (N–S) shelterbelt during 2013 and 2014 sampling seasons. Volumetric Water Content (VWC%; averaged across the 0–10 cm depth) and soil temperature (°C; measured at a depth of 10 cm) were measured in both irrigated (*open squares*) and rainfed (*filled squares*) sites of the shelterbelt. The *grey bar* on Panel **b** represents the approximate range where water filled pore space (WFPS) is at 60%



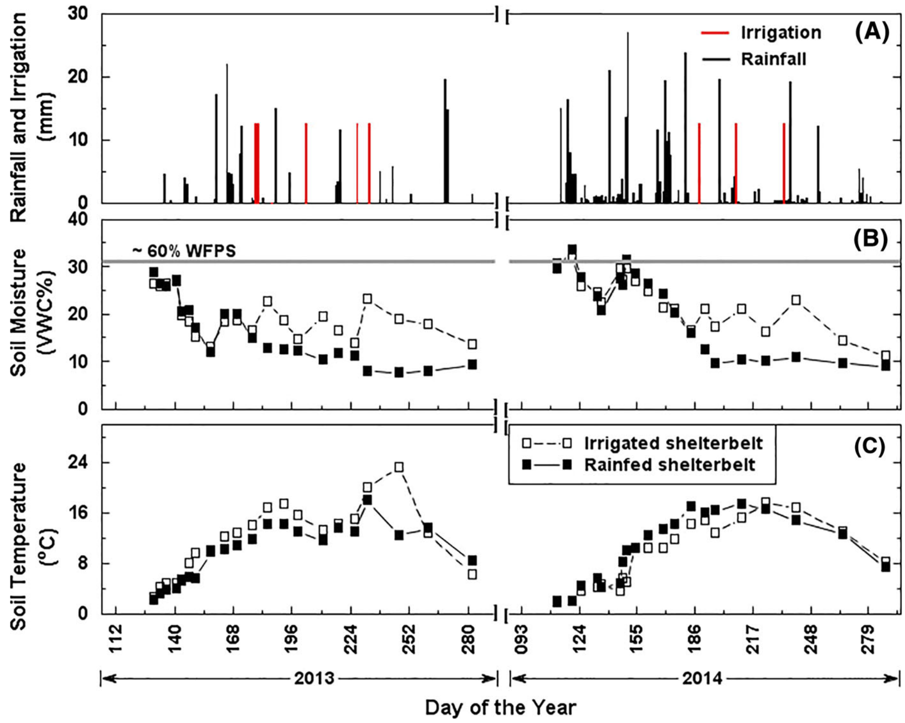


Fig. 3 Rainfall plus irrigation (**a**), soil water content (**b**) and soil temperature (**c**) measured at the Scots pine (E–W) shelterbelt during 2013 and 2014 sampling seasons. Volumetric Water Content (VWC%; averaged across the 0–10 cm depth) and soil temperature (°C; measured at a depth of 10 cm) were measured in both irrigated (*open squares*) and rainfed (*filled squares*) sites of the shelterbelt. The *grey bar* on Panel **b** represents the approximate range where water filled pore space (WFPS) is at 60%

3a, respectively) increased the total water inputs by 35 and 12% in 2013 and 2014, respectively.

Prior to the addition of irrigation water (i.e., from April to June) there were no significant differences in soil moisture between the irrigated and rainfed sections of any of the shelterbelts in 2013 ($P = 0.368$) or 2014 ($P = 0.765$). However, the addition of irrigation water resulted in varying levels of soil water in all three shelterbelts (Figs. 1b, 3b). For example, between the months of July and October 2013, volumetric soil water content in the irrigated sections of the shelterbelts was 43–45% greater than in the rainfed sections of the respective shelterbelts. Likewise, in 2014, soil water content was 38–47% greater in the irrigated sections of the shelterbelts than in the rainfed sections. Indeed, the rainfed sections of the shelterbelts were characterized by severe drying (<15% VWC) from late-July to mid-October in both 2013 and 2014. Conversely, soil water contents in the IR-SB remained above 15% VWC (with VWCs as great as 26%—equivalent to 50% WFPS) throughout the growing season and early fall.

In general, soil temperature in the IR-SBs followed seasonal trends similar to those in the RF-SBs (Figs. 1c, 3c) and did not respond to irrigation-induced changes in soil moisture. Indeed, across sites, there was no significant difference in mean soil temperature between the IR-SB and RF-SB sites in either 2013 ($P = 0.746$) or 2014 ($P = 0.886$).

Soil organic C content in the upper soil horizons (0–30 cm) tended to be greater ($P = 0.077$) in the IR-SB than in the RF-SB (Table 2). However, there were no significant differences ($P = 0.203$, 0.495, 0.612, and 0.613 for TN, bulk density, soil pH and C:N ratio, respectively) between the IR-SB and RF-SB sites. Similarly, there were no significant differences for soil NH_4-N ($P = 0.533$, 0.831, 0.709) and NO_3-N ($P = 0.436$, 0.951, 0.642) concentrations measured in July 2013, June 2014 and October 2014, respectively between the IR-SB and RF-SB sites (Table 3).

Soil N_2O, CH_4 and CO_2 exchange

During the 2-year study period, daily soil N_2O fluxes from the IR-SB and RF-SB sites ranged from -1.3 to 6.6 g N_2O-N ha^{-1} d^{-1}, with negative values indicating uptake (Fig. 4a). Average daily N_2O emissions in the RF-SB sites were highly variable, but followed the general event based/background pattern described by Yates et al. (2006), where the largest fluxes were

Table 2 Soil chemical and physical properties (0–30 cm soil layer) in irrigated and rainfed shelterbelts and across tree species at CSIDC Outlook, Saskatchewan, Canada

Soil property	IR-SB[b]	RF-SB[b]	*P* value
SOC (Mg ha^{-1})	74.4 (21.0)[a]	66.1 (18.6)	0.077
TN (Mg ha^{-1})	5.80 (1.56)	5.40 (1.69)	0.203
Bulk density (Mg m^{-3})	1.21 (0.08)	1.23 (0.05)	0.495
Soil pH	7.07 (0.18)	7.04 (0.18)	0.612
C:N ratio	12.85 (1.52)	12.45 (2.11)	0.613

[a] Numbers in parenthesis represent standard deviation

[b] Site names: *IR-SB* irrigated shelterbelts, *RF-SB* rainfed shelterbelts

Table 3 Available soil ammonium nitrogen (NH_4-N) and nitrate nitrogen (NO_3-N) within 0–30 cm soil depth measured in July 2013, June 2014 and October 2014 across irrigated and rainfed shelterbelt sites in CSIDC Outlook, Saskatchewan, Canada

Soil property	IR-SB[b]	RF-SB[b]	P-value
July-2013			
NH_4-N (µg N g $soil^{-1}$)	8.3 (3.8)[a]	8.9 (2.8)	0.533
NO_3-N (µg N g $soil^{-1}$)	4.4 (2.5)	3.9 (1.9)	0.436
June-2014			
NH_4-N (µg N g $soil^{-1}$)	9.9 (2.7)	9.7 (2.2)	0.831
NO_3-N (µg N g $soil^{-1}$)	5.4 (2.3)	5.4 (2.9)	0.951
October-2014			
NH_4-N (µg N g $soil^{-1}$)	6.8 (3.0)	6.5 (2.6)	0.709
NO_3-N (µg N g $soil^{-1}$)	3.0 (1.8)	3.1 (1.7)	0.642

[a] Numbers in parenthesis represent standard deviation

[b] Site names: *IR-SB* irrigated shelterbelts, *RF-SB* rainfed shelterbelts

associated with emission events activated by snow melt or precipitation, and with low fluxes occurring on most days during the summer and fall. Daily N_2O emissions in the IR-SB sites followed a similar trend, but were generally greater than those from the RF-SB sites—especially during the months when irrigation water was applied. Across the 2-year study period, cumulative N_2O emissions were significantly ($P = 0.016$) greater in the IR-SB than in the RF-SB sites (Table 4). The difference largely reflects the greater ($P = 0.008$) cumulative N_2O emissions in the IR-SB sites during 2013, as there was no difference in cumulative N_2O emissions in 2014.

Methane emission and consumption were observed in both the IR-SB and RF-SB sites during the study period, with rates ranging from −7.8 to 31.4 g CH_4-C ha^{-1} d^{-1} (Fig. 4b), with negative values indicating uptake. Daily CH_4 fluxes did not show any clear seasonal pattern but did appear to vary with changes in soil water content. That is, the soil was a small source of CH_4 during brief periods when the soil water content was high (e.g., during snow melt and following early season high precipitation events), but reverted to a CH_4 sink during the summer and fall months when soil moisture was low.

Large, transient CH_4 emission events were observed in 2013, and in both IR-SB and RF-SB sites. For example, during August 2013 a single large CH_4 emission event at IR-SB sites contributed 28% of the total annual CH_4 emission at these sites (Fig. 4b). Regardless of these emission events, however, soils under all the shelterbelts were small net sinks for CH_4 (Table 4). Indeed, summed across the entire 2-year study period, the sink potential (i.e., cumulative CH_4 uptake) of the IR-SB was significantly lower ($P = 0.034$) than that of the RF-SB sites (Table 4). This largely reflects the significant ($P = 0.007$) difference in cumulative CH_4 oxidation between the RF-SB and IR-SB in 2013 (Table 4).

Average daily CO_2 fluxes in both the IR-SB and RF-SB sites ranged from 2.7 to 45.4 kg CO_2-C ha^{-1} d^{-1} (Fig. 4c). Moreover, daily CO_2 fluxes followed similar seasonal trends as soil temperature; i.e., the highest fluxes occurred during periods of high soil temperature (June–August) and the lowest fluxes occurred during periods of low soil temperature (typically during the early spring and late fall). Summed across both study years, cumulative CO_2 emissions in the IR-SB tended to be greater ($P = 0.063$) than those in the RF-SB (Table 4).

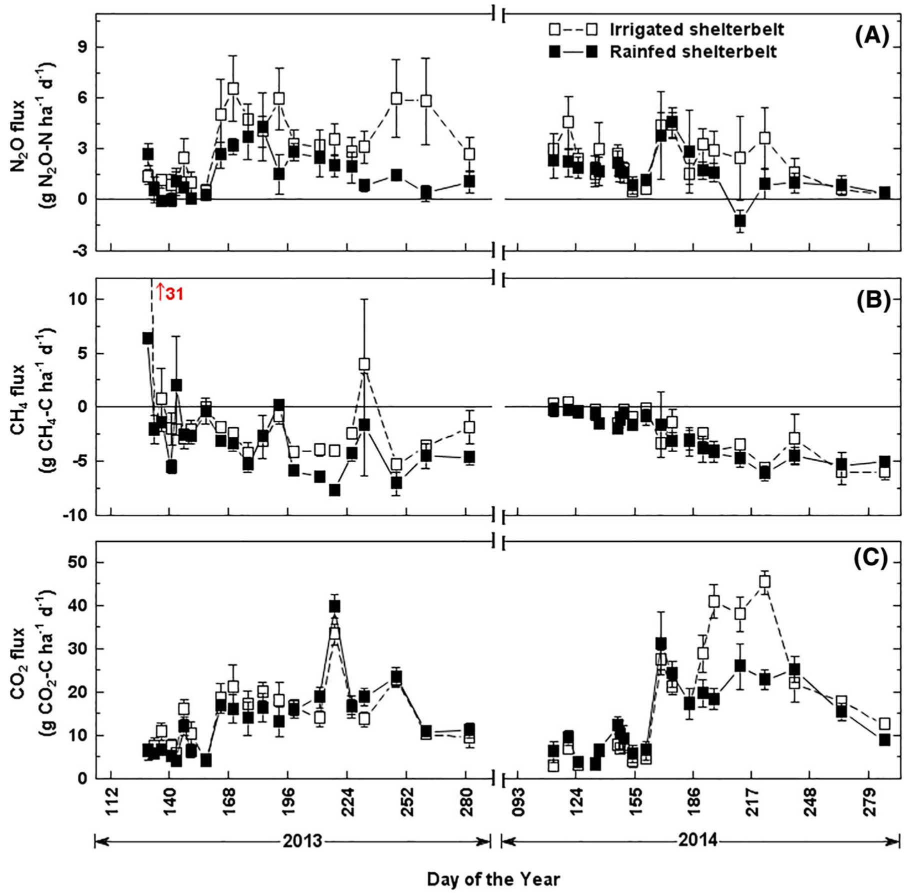


Fig. 4 Daily soil N_2O fluxes (**a**), CH_4 fluxes (**b**) and CO_2 fluxes (**c**) (g flux ha^{-1} d^{-1}) in the irrigated (*open squares*) and rainfed (*filled squares*) shelterbelts during 2013 and 2014 study period. *Error bars* represent standard deviation (n = 3)

Table 4 Cumulative fluxes of N_2O (kg N_2O-N ha^{-1}), CH_4 (kg CH_4-C ha^{-1}) and CO_2 (Mg CO_2-C ha^{-1}) in irrigated and rainfed shelterbelts during 2013 and 2014 study periods, and for the sum of both years

Gas flux	IR-SB[b]	RF-SB	P-value
N_2O flux (kg N_2O-N ha^{-1})			
2013	0.57 (0.64)[a]	0.25 (0.19)	0.008
2014	0.36 (0.39)	0.24 (0.32)	0.204
Total flux	0.93 (1.00)	0.49 (0.47)	0.016
CH_4 flux (kg CH_4-C ha^{-1})			
2013	−0.30 (0.24)	−0.59 (0.28)	0.007
2014	−0.55 (0.15)	−0.62 (0.26)	0.462
Total flux	−0.85 (0.29)	−1.20 (0.51)	0.034
CO_2 flux (Mg CO_2-C ha^{-1})			
2013	2.32 (0.72)	2.30 (0.73)	0.949
2014	3.52 (0.54)	2.88 (1.14)	0.016
Total flux	5.83 (1.10)	5.18 (1.80)	0.063

[a] Numbers in parenthesis represent standard deviation

[b] Site names: *IR-SB* irrigated shelterbelts, *RF-SB* rainfed shelterbelts

However, this was due mainly to a greater response of CO_2 emissions to elevated soil moisture observed in 2014. During 2013, daily CO_2 fluxes in the irrigated sections of all three shelterbelts followed seasonal patterns similar to those observed in the rainfed sections; and irrigation did not appear to stimulate CO_2 emissions in the irrigated sections. Consequently, there was no significant ($P = 0.949$) difference in cumulative CO_2 emissions between the IR-SB and RF-SB sites in 2013 (Table 4). Conversely, irrigation had a significant effect on the daily CO_2 emissions in 2014, with emissions generally being greater in the IR-SB sites (Fig. 4c). As a result, cumulative CO_2 emissions were significantly greater in the IR-SB than in the RF-SB sites in 2014 (Table 4).

Relationships of soil temperature and moisture with soil gas exchange

There was a positive relationship between daily CO_2 flux and soil temperature in both the IR-SB ($r = 0.59$, $P < 0.001$) and RF-SB ($r = 0.53$ $P < 0.001$) sites. Significant correlations also were observed between the daily CO_2 flux and soil water content in both the IR-SB ($r = 0.30$, $P < 0.001$) and RF-SB (0.44 $P < 0.001$) sites. Whereas soil temperature was negatively correlated with daily CH_4 flux (IR-SB: $r = -0.35$, $P < 0.001$; RF-SB: -0.30 $P < 0.001$), the daily CH_4 flux was positively correlated with soil moisture ($r = 0.28$, $P = 0.002$ and $r = 0.46$, $P < 0.001$ for the IR-SB and RF-SB sites, respectively). A significant positive relationship ($r = 0.52$, $P < 0.016$) also was found between N_2O emissions and soil moisture for the IR-SB sites in 2013, but not in 2014. There was no significant correlation between daily N_2O flux and either soil temperature or soil moisture in the RF-SBs in either 2013 or 2014.

Discussion

Given its large global warming potential, even small emissions of N_2O can have a significant impact in terms of radiative forcing. Indeed, emissions totaling only 3.35 kg N_2O-N are enough to offset the sequestration of 1 Mg of CO_2-C. The production and emission of soil-derived N_2O are primarily regulated by soil water content, substrate availability and temperature (Dobbie et al. 1999)—each of which influences the microbial processes by which N_2O is produced; i.e., nitrification and denitrification (Mosier et al. 2006). The range of daily N_2O fluxes measured in this study (−1.3 to 6.6 g N_2O-N ha^{-1} d^{-1}) is similar to that reported in forested soils (−4–7 g N_2O-N ha^{-1} d^{-1}) in the Mid-Boreal Upland Ecoregion of Central Saskatchewan (Matson et al. 2009), but are well below the average emission rate of 13.9 g N_2O-N ha^{-1} d^{-1} reported for cropped fields at the CSIDC (Amadi et al. 2016a). This suggests that regardless of their soil moisture status, shelterbelts planted on agricultural landscapes can play an important role in reducing N_2O emissions from crop production systems. The greater N_2O emission in the cropped fields was attributed mainly to an increase in available N (i.e. NO_3^- and NH_4^+) as a result of the application of fertilizer N (Amadi et al. 2016a).

The application of irrigation water to the IR-SB sites resulted in an increase in soil water content, which in turn tended to induce greater N_2O emissions compared to the RF-SB sites across the 2 years of study (Table 4). The year 2013 was particularly dry, with total precipitation 35% lower than long-term (30-year) average, which may have affected microbial N_2O production processes in the soil. Thus the addition of irrigation water in 2013 appeared to have stimulated microbial processes of N_2O production, resulting in the greater N_2O emissions observed in the IR-SB sites. However, in 2014 where total precipitation was 17% greater than long-term average, water additions through irrigation did not stimulate as much N_2O emissions as in 2013. This suggests that N_2O emissions in 2014 were limited less by soil moisture than by other factors—the most likely being low soil available N in the shelterbelts (Table 4). Aside from a few days in late-April and early-June 2014, soil water content rarely exceeded 60% WFPS (i.e., ~32% VWC)—even during periods of active irrigation (Figs. 1b, 3b). This suggests that nitrification was the predominant driver of N_2O production in the shelterbelts. Other studies have reported nitrification as the major driver for N_2O production, especially in dry, well drained soils such as those at the CSIDC site (Rosenkranz et al. 2006; Chapuis-Lardy et al. 2007).

Correlation analyses showed a significant positive relationship between N_2O emissions and soil moisture for the IR-SB sites, but not in the RF-SB sites, in 2013. As well, there was no significant relationship between soil moisture and N_2O fluxes for either the IR-SB or RF-SB sites in 2014. Although several studies have shown a strong relationship between N_2O emissions and soil water content (Smith et al. 2003; Ball et al. 2007), we suspect that low concentrations of soil available N in the shelterbelts may have obscured any relationship between soil moisture and N_2O emissions. Similar findings were reported by Peichl et al. (2010).

It was observed that in response to elevated soil moisture, N_2O emissions were more intense in the caragana shelterbelts than in the Scots pine shelterbelts, especially in 2013. This was reflected in the high standard deviation values associated with the cumulative N_2O data for the IR-SB sites in 2013 (Table 4). We suspect that the increased N_2O emission response to the addition of irrigation water in the caragana shelterbelts was primarily related to the N-fixing feature of the caragana and perhaps increased soil available N concentrations. Indeed, our data show that compared to soils under the Scots pine shelterbelts, soil under the caragana shelterbelt had significantly greater concentrations of both total (7.6 vs. 4.6 Mg N ha^{-1}; $P = 0.001$) and available N (31.4 vs. 18.6 kg N ha^{-1}; $P = 0.001$) throughout the study period. Other studies have shown that caragana can fix 75–85% of its N from the atmosphere (Moukoumi et al. 2013)—returning about 20–60 kg N ha^{-1} to the soil in the litter (Issah et al. 2014). Thus, it is likely that inputs to the soil of N derived from N_2-fixation exceeded plant N requirements and contributed to the observed enhancement of N_2O emissions.

It should be noted that the caragana shelterbelt is about 17-years older than the Scots pine shelterbelts, which presumably explains, at least in part, the greater concentrations of SOC and total soil N under the caragana (see Table 1). In turn, this likely contributed to the increased N_2O emission response to irrigation in the caragana shelterbelt. Although the current study was not designed to quantify soil N_2O responses of various shelterbelt tree species to elevated soil moisture, our data certainly suggest that N-fixing trees can modify the dynamics of N_2O emissions under shelterbelts—especially under elevated soil moisture conditions. Indeed, our data demonstrate that it is the confluence of N, C and water inputs in the irrigated shelterbelts that drives changes in GHG dynamics in the shelterbelt system. Clearly, future studies will need to investigate the size of the effect of N-fixing shelterbelt trees on the dynamics of GHG emissions in agricultural landscapes.

Daily CH_4 fluxes measured in this study were in the range that Matson et al. (2009) reported for forested soils in the Mid-Boreal Upland Ecoregion of Central Saskatchewan (i.e., −39 to +25 g CH_4-C ha^{-1} d^{-1}). Whereas upland cultivated soils can serve as small sinks or small sources of CH_4 depending on soil water conditions (Mosier et al. 2006), planted shelterbelts have been reported to strengthen the CH_4 sink potential in cultivated soils (Amadi et al. 2016a). Nevertheless, our data show that elevated soil moisture reduced the sink size of CH_4 in the shelterbelts across the entire study period (Table 4). The correlation analysis showed a positive relationship between soil water and CH_4 flux, which is in agreement with studies that have reported decreasing CH_4 uptake (i.e., oxidation) with increasing soil moisture content in forested soils (Rosenkranz et al. 2006).

Methane production involves the microbial breakdown of organic compounds in strictly anaerobic conditions and at very low redox potential. Methane emissions occurred during the snowmelt period and CH_4 uptake was lowest following rainfall and irrigation events. Given the well-drained sandy loam soils of our study sites, in addition to water uptake from the vast network of tree roots, soil moisture in the 0–30 cm soil layer was increased only for a limited period following rainfall or irrigation events. Thus, while the temporal scale and duration of CH_4 emission following elevated soil moisture in shelterbelts remains uncertain, we believe that CH_4 production occurred in short-lived intervals, lasting only as long as anaerobic conditions existed in the soil. In addition, we observed that the LFH-layer remained moist throughout most of the day following rainfall or irrigation events. A similar observation was made by Rosenkranz et al. (2006) who reported a negative relationship between water content in the organic layer and CH_4 uptake. Thus, in addition to CH_4 production in the mineral soil, anaerobic microsites in the moist LFH layer may have contributed to the lower CH_4 uptake observed in the IR-SB sites. This is in agreement with Peichl et al. (2010) who reported that anaerobic microsites in the LFH-layer can be sources of CH_4 production and that these anaerobic microsites may at times reach magnitudes that surpassed limited microbial CH_4 consumption in the dry mineral soil.

Daily CO_2 fluxes measured in this study were similar to those reported by Peichl et al. (2010) for forested soils in southern Ontario (i.e., 2–50 kg CO_2-C ha^{-1} d^{-1}). In general, CO_2 fluxes were driven mainly by trends in soil temperature, which would have exerted an effect on both soil microbial respiration and root respiration (Robertson et al. 2000). This trend was supported by the correlation analysis—both in the present study and the study by Peichl et al. (2010), which indicated a strong link between the daily CO_2

flux and soil temperature. In addition to this overarching trend, increased soil moisture resulting from above-normal precipitation and the application of irrigation water also appeared to stimulate CO_2 production/emission in 2014. However, irrigation had no significant effect on CO_2 emissions during the much drier 2013 season. Nevertheless, when examined across the entire study period, we found a strong correlation between CO_2 flux and soil water content. This agrees with previous studies that have demonstrated an increase in aerobic soil respiration in response to an increase in soil moisture and a positive linear relationship between soil moisture and CO_2 emissions (Ellert and Janzen 2008; Jabro et al. 2008; Trost et al. 2013).

Conclusion

Shelterbelts are valuable tree features in agricultural landscapes that while intended mainly to protect soils and crops from wind damage, can also play a role in C storage and the mitigation of soil-derived GHG emissions. Elevated soil moisture had a significant impact on soil-derived GHG fluxes in the shelterbelts, though the magnitude of the impact was greatly influenced by weather conditions in each year. The 2 years of the study provided contrasting weather conditions—with 2013 being a dry year (i.e., 35% lower precipitation) and 2014 being slightly wetter (i.e., 17% greater precipitation) compared to long-term (30-year) averages.

In 2013, with limited soil moisture, the addition of irrigation water to the shelterbelts triggered a significant increase in N_2O emissions, and at the same time, a decrease in CH_4 oxidation. In 2014 where there was more precipitation, the addition of irrigation water to the shelterbelts did not have the same magnitude of effect on either N_2O emissions or CH_4 oxidation as compared to 2013. This suggests that the impact of elevated soil moisture conditions on N_2O emissions from shelterbelts depends on the status of other factors such as substrate availability (especially available N). In terms of soil-derived CO_2 emissions, shelterbelts have the potential to emit more CO_2 in response to irrigation, though this effect appeared to be dependent on the total amount of water received as both precipitation and irrigation. That is, a positive response to irrigation appears to have depended on the ability of the trees to access water from outside the canopy and a subsequent effect on root and/or microbial respiration.

Nitrogen-fixing trees, such as caragana can fix considerable amounts of atmospheric N_2 in their leaves and this N is then returned to the soil as litterfall, which is then incorporated into the various soil-N pools. This includes the available N pool, which is subject to microbial transformations (nitrification and denitrification) that can yield N_2O. Future studies should monitor the dynamics of soil-atmosphere GHG exchange in N-fixing trees—especially under elevated soil moisture conditions—to improve the effectiveness of shelterbelts as a strategy to mitigate agricultural GHG emissions.

Overall, our study suggests that changes in precipitation patterns and soil moisture regime due to climate change may affect soil-atmosphere exchange of GHGs in shelterbelts. However, rather than just the elevated soil moisture effect, it is the confluence of N, C and water inputs that drive the changes in GHG dynamics in the shelterbelts.

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