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GREENHOUSE GAS DYNAMICS IN A TREE-BASED INTERCROPPING SYSTEM COMPARED TO AN ORGANIC CONVENTIONAL SYSTEM

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ABSTRACT

Tree-based intercropping (TBI) system has been suggested as an alternative to conventional monocropping (CM) for decreasing overall GHG emissions. However, little is known about the advantages of TBI compared with organic CM system with low fertilizer inputs. This project compared CO₂ and N₂O dynamics in replicated TBI and CM in Saint-Paulin (Québec, Canada), established about 12 years prior to this study under organic production, with horse manure and pruning residues as soil amendments. An experimental field, using a complete randomized block design, was instrumented in both systems. A five-point sampling transect was established between tree rows in each of the six plots. Surface CO₂ and N₂O fluxes, CO₂ and N₂O soil concentrations and other physico-chemical properties were analyzed at 0.125 and 0.25 m depths over two growing seasons. CO₂ and N₂O concentrations in TBI soil were similar to or lower than in CM soil. Lower CO₂ and N₂O emissions suggest TBI would decrease overall GHG mitigation, even when compared with organic production with low fertilizer inputs, in addition to its potential for carbon sequestration in wood and other advantages, which are not discussed in this study.

Keywords: Agroforestry, Carbon dioxide, Nitrous oxide, Soil physics, Trees, Emission fluxes

INTRODUCTION

Agroforestry has received renewed attention brought by continuing global change, considering the inclusion of trees could potentially decrease carbon (C) loss in agricultural ecosystems, both through increased carbon sequestration and through decreased C emissions (Quinkenstein et al., 2009; Nair 2012; Winans et al., 2014). Tree growth influences soil organic carbon (SOC) content and CO₂

dynamics (Dixon et al., 1994; Schoeneberger et al., 2012). Trees act as natural long-term sinks for atmospheric CO₂ by capturing and storing carbon in their woody tissues, and by transferring fixed carbon to various soil pools through their annual litter fall as well as through fine root decomposition and turnover (Oelbermann et al., 2004; Quinkenstein et al., 2009). Carbon sequestration resulting from the implementation of agroforestry practices could potentially restore SOC levels in agricultural fields within a period of 20 to 50 years (Montagnini and Nair 2004; Lorenz and Lal 2014). Under agroforestry systems, carbon storage may reach 9 Mg C ha⁻¹ in semiarid, 21 Mg C ha⁻¹ in sub-humid, 50 Mg C ha⁻¹ in humid, and 63 Mg C ha⁻¹ in temperate ecoregions (Schroeder, 1993). Trees can influence soil temperature and moisture regimens under adjacent crops through the wind-buffering, shading and transpiration effects they exert on water and heat dynamics (Davis and Norman, 1998). Thus, trees can have a strong local influence on carbon cycling in the agricultural landscape, which includes mitigating rates of CO₂ emissions and, potentially, rates of other GHGs (i.e., CH₄ and N₂O).

Among the various types of agroforestry systems, tree-based intercropping (TBI) has been shown to increase soil C sequestration (Quinkenstein et al., 2009; Bambrick et al., 2010; Udawatta and Jose, 2012) and to decrease N₂O emissions (Beaudette et al. 2010), when compared with conventional monocropping (CM) systems. In southern Canada, TBI systems can reduce soil N₂O emissions by 1.2 kg ha⁻¹ y⁻¹ compared with CM, through tree uptake of the excess N (nitrate) fertilizer applied to row crops (Evers et al., 2010). Moreover, this nitrogen eventually returns to the soil in the form of slowly decomposing leaf litter and green manure (tree pruning) inputs for the next crops. TBI systems may also indirectly mitigate N₂O emissions, whereby the presence of trees in crop fields favours greater soil moisture extraction, which would reduce local temporary anoxic conditions leading to denitrification. Agricultural N₂O emissions are frequently associated with inefficient inorganic-N applications, and are produced through coupled microbial nitrification-denitrification processes in these soils (Bouwman et

al., 2002; Desjardins et al., 2010). Application of TBI to cropping systems should result in a decrease of synthetic fertilizer demand and in lower N₂O emissions (Thevathasan et al., 2012).

We expected gas dynamics in agroforestry systems would differ from those of CM agricultural practices due to the influence of trees. The effects of trees are shaped by land, which in turn influence a variety of factors, including crop and tree species, climate and time-since-installation, together with management practices. For example, tree species used in temperate TBI systems can differ in their litter production and chemistry, conditions sufficient to cause divergence in soil C and N dynamics (Mungai and Motavalli, 2006). Yet, direct and indirect interactions between trees and soil parameters on GHG dynamics (CO₂, N₂O) and on soil C in agroforestry systems are not well documented in Canada. Even less is known about CO₂ and N₂O dynamics under organic production, regardless of whether they are TBI or CM systems.

The aforementioned studies have included GHG responses of treatment plots that were amended with mineral fertilizers – a routine practice in industrially cropped land areas across Canada. Over a 25-year period, total nitrogen applications (including manure N) per hectare of farmland in Canada rose from an average 24 kg N ha⁻¹ in 1981 to 36 kg N ha⁻¹ in 2006 (Yang et al., 2011). Synthetic inorganic fertilizers have come to dominate total N inputs, i.e., from 41.3 % (1981) to 56.9 % (2006) over the same period, and with no sign of decreasing trend.

Crop production in agroforestry can be managed under a wide range of agricultural practices, but producers in agroforestry systems often employ low-impact techniques, such as organic production using manure inputs, either to protect their trees or because they believe in environmentally sustainable farming (Jordan, 2004). Given that mineral fertilizers are a major source of soil GHGs, it remains unclear whether tree-based intercropping would provide additional environmental benefits compared with CM in organically fertilized soils or organically managed systems. The objective of the study was

to compare in situ N_2O and CO_2 emissions and soil gas concentrations over two growing seasons under TBI and CM crop production, which employed organic fertilizer (horse manure) applications. We made the hypothesis that under organic production with few external inputs, trees will have a small but significant effect on carbon and nitrogen dynamics in soil.

MATERIALS AND METHODS

Site description

The study was conducted over two growing seasons (2012-2013) in a TBI system (established in 2004) near St-Paulin, Québec (46°27'N, 72°59'W). The site was originally installed to test the Ecosys Modelling program of a comprehensive mathematical model of diverse natural and managed ecosystems (Wang et al., 2011). This site serves to collect input data among different sites chosen for agroforestry systems. It was also used to test the effect of trees on water dynamics in agroforestry systems (Rousseau et al., 2012).

The land is relatively flat (1% slope). The soil was classified as a Dystric Brunisol (Lamontagne and Nolin, 1997; FAO equivalent, Dystric Cambisol), with a sandy loam texture (82 % sand, 12 % silt, 6 % clay) and a pH_{water} of 5.5. pH varied little across the field and increased by 0.5 units after liming. Annual temperature average is 4°C in the region which receives 1113 mm y^{-1} of precipitation (Environment Canada 2013). During the 2012 growing season, average monthly air temperatures were 12.1°C (May), 18.7°C (June), 21.0°C (July), 20.3°C (August), 14.3°C (September), and 9.1°C (October). For the same months in 2013, average air temperatures were respectively 14.7, 16.5, 20.7, 18.9, 13.7 and 10.1 °C.

The experimental layout was a complete randomized block design with two agricultural systems (TBI vs CM) replicated in four blocks, three of which were used for studying GHG dynamics (Fig. 1). Rows of hybrid poplars (*Populus deltoides* W. Bartram ex Humphry Marshall $\times$ *P. nigra* L.) alternated

with rows of high-value hardwood (HVH) species (*Quercus rubra* L., *Prunus serotina* Ehrhart). Tree rows were 12 m apart and planted in spring 2004 to form the TBI system. Poplar cuttings were planted at 2 m intervals (238 stems ha⁻¹); hardwood seedlings were planted every 3 m (134 stems ha⁻¹). A 2 m-wide uncultivated strip was maintained within each tree row, including a 1.2 m-wide band of black polyethylene mulch. Each plot (60 m × 24 m) was bounded on either side by a row of hybrid poplars (HP), with two alleys of crops in the middle, which were separated by one row of hardwood species (Fig. 1).

Crop rotations in the alleys consisted of oat (*Avena sativa* L.)-buckwheat (*Fagopyrum esculentum* Moench) in 2004-2005, buckwheat-canola (*Brassica napus* L.) in 2006, and buckwheat in 2007. The annual crops were followed by 4 years (2008 to 2012) of forage production (*Trifolium pratense* L. and *Phleum pratense* L. mixture), and by buckwheat in 2013. CM plots (30 m × 24 m; no trees, Fig. 1) used the same crop rotation as the TBI plots. Annual tree maintenance included pruning and thinning; two out of every three poplars were cut in 2012. The soils were tilled from 2004 to 2009. Tillage using a disk harrow in the spring incorporated manure and unharvested residues into the soil (0-8 cm), within ~1-2 m of the tree stems. Moldboard plowing (to 15-cm depth) occurred in fall after the annual crop harvest, but no tillage was performed after seeding forages. Horse manure (solid manure mixed with bedding; C/N ratio of 25:1) was broadcasted on the surface in spring 2004 and 2007 at a rate of 30 Mg ha⁻¹ with a rear-discharge manure spreader prior to tillage. In November 2007 and 2012, lime (CaCO₃, 5 Mg ha⁻¹) was incorporated into the soil by plowing. No other fertilizers were applied. The site was managed under organic production practices; only horse manure and pruning residues were applied annually as soil amendments. Deep soil tillage (up to 40 cm depth) was completed only once, in fall 2012.

In situ instrumentation and measurements

Thirty sampling points were established along transects (6 transects x 5 sampling points) in each block, perpendicular to the edge of the field (tree rows in TBI plots). One point was located on the HVH tree line in TBI plots (0 m), two points at 4 m on either side of HVH (-4 vs 4 m) and 2 points at 10 m on either side of HVH (-10 vs 10 m). The latter points were 2 m from the poplar lines (Fig. 1). In the CM plots, sampling points were located at the same distances from the plot edge. Transects were aligned from West to East going from one hybrid poplar row (-10 m point), then across the high-value hardwoods (0 m), and finally to the second hybrid poplar row (10 m).

Soil gas concentrations ($\text{CO}_{2\text{-soil}}$, $\text{N}_2\text{O}_{\text{-soil}}$) were measured using sealed 50 cm-long access tubes made of PVC tubing (0.5 cm ID) and vertically inserted to 0.125 m and 0.25 m depths. Soil temperature (T) was monitored using type-T (copper-constantan) thermocouples at the same depths, while volumetric water content (WC) was measured using 0.15 metre-long TDR (Time-Domain Reflectometry) probes at the same depths. WC was calculated with Topp's equation (Topp et al., 1980). Soil parameters were measured 6 times from May to October 2012 and 9 times from April to November 2013. Temperatures ranged from 4.5 to 34 °C, with means of 14 °C in 2012 and 17 °C in 2013 respectively. Still in 2013, WC averaged 0.20 $\text{m}^3 \text{m}^{-3}$ at 0.125 m and 0.16 $\text{m}^3 \text{m}^{-3}$ at 0.25 m depths. Based on soil water retention curves (using tension tables), the lowest WC values indicated that the soil reached the temporary wilting point for short periods for both years.

Intact cores (0.06 m dia. x 0.06 m height) were taken at every sampling point along transects at the same depths as for temperature and moisture measurements to determine soil bulk density (BD, Mg m^{-3} ; Blake and Hartge, 1986) and saturated hydraulic conductivity (K_{sat}). K_{sat} was measured using constant head permeametry (Klute and Dirksen, 1986). Bulk density varied from 1.3 to 1.5 Mg m^{-3} , with a mean value of 1.4 Mg m^{-3} , which indicated relatively high soil densities. K_{sat} values ranged from

3.6 to 30 m d⁻¹, which meant that some water would form puddles on the soil surface after intense rainfall events. The soils were otherwise relatively well drained.

Additional soil was sampled at 0.125 m and 0.25 m depths at every sampling point to measure total organic carbon content (C_{Total}), which was determined using dry combustion (Skjemstad and Baldock, 2007), and soluble organic carbon concentrations (C_{soluble}), which were quantified colorimetrically using a coupled Mn reduction-organic carbon oxidation reaction (Bartlett and Ross, 1988). Inorganic nitrogen pools (NO_3^- and NO_2^- ; NH_4^+ not detectable) in additional soil aliquots were quantified colorimetrically (Griess-Ilosvay reaction) following KCl extraction (Keeney and Nelson, 1982). C_{Total} , C_{soluble} , NO_3^- and NO_2^- were only measured in samples taken on July 5, 2013.

***In situ* soil gas concentrations and flux measurements**

Gas access tubes were flushed through the septa using gas-tight hypodermic syringes (10 mL, Becton-Dickinson 309643, Franklin Lakes, NJ) about 2 h prior to sampling. Gas samples (10 mL) were drawn using the same syringes and immediately inserted into gas-tight vials (10 mL model 5182-0838, Agilent Technologies, Wilmington, DE) that had been previously purged and capped with aluminum seals (20 mm, 224178-01, Wheaton, Milleville, NJ) and rubber butyl septa (Wheaton 224 100-202), which were specifically selected for CO_2 and N_2O measurements (Lange et al. 2008).

Surface gas emissions ($\text{CO}_2\text{-flux}$, $\text{N}_2\text{O-flux}$) were measured using dynamic closed chambers (Rochette and McGinn, 2005) directly installed on the soil surface. They were made of PVC (0.15 m radius x 0.22 m height). Ten chambers were installed simultaneously to minimize measurement duration. They were rotated to complete the 30 sampling points. Gas samples were taken at 0, 2.5, 5, 7.5, 10, 15, 20, and 30 min. These samples (10 mL) were drawn using the same syringes, septa and vials used for sampling the soil atmosphere. Soil gas concentrations ($\text{CO}_2\text{-soil}$, $\text{N}_2\text{O-soil}$) and fluxes ($\text{CO}_2\text{-soil}$, $\text{N}_2\text{O-soil}$)

were measured 6 times over the 2012 growing season (24 May to 11 October) and 9 times during the 2013 growing season (6 May to 23 September).

CO₂ and N₂O concentrations were quantified using a gas chromatograph (model 6890N, Agilent Technologies) equipped with a 30 m HP-PLOT-Q column (19095P-QO4PT), and a TCD detector in 2012, and TCD and ECD detectors in 2013. The carrier gas was helium (UH-T 5.0, Praxair, Danbury, CT) in 2012 and Ar-CH₄ (95:5 P-5, Praxair) in 2013. CO_{2-flux} and N₂O-flux (μg m⁻² s⁻¹) were calculated from linear portions of the chamber concentration vs time curves (Rochette and McGinn, 2005).

Statistical analyses

Descriptive statistics of all variables were calculated using PROC UNIVARIATE (v. 9.3, SAS Institute 2012). Prior to analysis, data was subjected to Box-Cox transformation (Box and Cox 1964) when required, to improve the normality of their distributions.

Treatment responses were analyzed with three-way mixed-effects RMANOVA (MIXED procedure, with repeated measures). Variance-covariance structures of the data were determined by model selection using the Akaike information criterion (Akaike, 1974). Fixed effects were treatment (2 levels), distance (3 levels), depth (3 levels) and time (3 levels), while the random effect was block. Correlations among CO_{2-soil}, CO_{2-flux}, N₂O-soil, N₂O-flux, and other soil properties were calculated using PROC CORR with Bonferroni corrections. Stepwise regressions (PROC STEPWISE, SAS Institute 2013) were performed using bidirectional elimination following pre-selection with main component analyses.

RESULTS AND DISCUSSION

Carbon dynamics

Total soil carbon (C_{Total}) averaged about 30 g kg^{-1} across the site, varying from 9.4 g kg^{-1} to 50.9 g kg^{-1} near the soil surface, and from 12.6 to 93.9 g kg^{-1} at depth (Table 1). Respective coefficients of variation (CV) were $\geq 28\%$. C_{Total} and its variability was consistent with values reported for agricultural soils in the region (Bambrick et al., 2010; Peichl et al., 2006; Rivest et al., 2013). System, depth and distance from trees did not significantly influence C_{Total} . Soil heterogeneity, redistribution of leaves by the wind, and burial of branches likely explained the relatively high degree of spatial variability across the plots. This high variability may explain the lack of significant effects.

In 2013, soluble carbon (C_{soluble}) concentrations averaged 266 mg kg^{-1} at 0.125 m and 241 mg kg^{-1} at 0.250 m (Table 1), with a CV (29 %) in the same range as for C_{Total} . Neither depth nor distance from the trees significantly influenced C_{soluble} , but the treatment system did exert an effect (TBI vs CM, $P = 0.001$). The range of soluble C values (115 to 357 mg kg^{-1}) in CM was wider and higher than that of TBI (128 to 273 mg kg^{-1}), although mean soil C_{soluble} was higher in TBI compared to CM. Given that the same crop was planted in both systems, this difference could have been attributed to faster release of C_{soluble} from leaf litter.

Soil CO_2 concentrations ($\text{CO}_{2\text{-soil}}$) ranged from 636 to $47\,149 \text{ } \mu\text{g L}^{-1}$, with CVs of about 47 % (Table 1). These values were comparable to those estimated in conventional agricultural studies for similar climates (Allaire et al., 2012, 2015), and in agroforestry systems (Peichl et al., 2010). $\text{CO}_{2\text{-soil}}$ was consistently ($P < 0.0001$) lower $+10 \text{ m}$ (East) than -10 m (West) from the tree row (Fig. 2), even though the same hybrid poplars were planted on either side of the plots. This difference may be related to natural soil gradients across the site (among them slope, large gradient in soil texture, horizonation and pH).

Despite potential field gradients, the presence of trees (TBI vs CM, $P = 0.001$), the distance from them ($P < 0.0001$), the soil depth ($P < 0.0001$) and seasonal variation ($P < 0.0001$) strongly affected $\text{CO}_{2\text{-soil}}$ (Fig. 2). Early in 2012, $\text{CO}_{2\text{-soil}}$ was 10 % higher under CM than under TBI, an effect that disappeared toward the end of the growing season. The difference may be attributed to the trees, which affected soil temperature and moisture content (correlated with $\text{CO}_{2\text{-soil}}$ at $P < 0.0001$), that control CO_2 production in the soil (Allaire et al., 2012, 2015). Microbial processes are more efficient in using soil C (lower CO_2 emissions) in TBI than in CM systems, possibly because the microbial communities are more diverse (Lacombe et al., 2009; Rivest et al., 2013). In contrast, the agricultural system did not influence $\text{CO}_{2\text{-soil}}$ over the 2013 growing season ($P = 0.22$), probably because deep tillage severed tree roots and redistributed organic residues from pruning, increasing spatial variability and temporarily offsetting the effects that trees would exert on soil properties.

Within TBI, $\text{CO}_{2\text{-soil}}$ rose by 20 % as the distance from the trees increased (Fig. 2, +4 and -4 m) with the highest concentration under the annual crop located in the middle of the alley. This trend was consistent from season to season, and likely related to intensive soil microbial and root activity of the annual crop (forage and grass). Changes in soil temperature and moisture also likely drove this trend. Both properties varied systematically with the distance from the trees ($P < 0.0001$) and were correlated with $\text{CO}_{2\text{-soil}}$ ($P < 0.0001$), since higher soil temperature was favoured under forage and grass than around trees. Soil water content within the topmost 30 cm was higher under crops than under trees.

As expected, $\text{CO}_{2\text{-soil}}$ varied over the growing season ($P = 0.001$), but was only influenced by the agricultural system in May 2012 (Fig. 2). Trees influenced $\text{CO}_{2\text{-soil}}$, which changes as the distance from them increases, but the average over the whole site did not. Less snow accumulates around the trees during winter, and soil temperature remained higher beneath and around trees prior to leaf out. In dryer years, little snow accumulated over the site, as was the case in 2013, which diminished the aforementioned springtime difference.

Since the trees influenced $\text{CO}_{2\text{-soil}}$, litter fall deposit, and the soil water regimen at different depths ($P = 0.0001$), we expected that they would affect $\text{CO}_{2\text{-soil}}$ differently at various soil depths. Neither agricultural system nor distance from the trees interacted with depth on $\text{CO}_{2\text{-soil}}$, probably because the pronounced gradient of increasing CO_2 concentrations with depth masked the effects of all other treatments. Consistent with previous observations (Allaire et al., 2012), $\text{CO}_{2\text{-soil}}$ was typically higher at 0.25 m depth (mean = $14339 \mu\text{g L}^{-1}$) compared with 0.125 m depth (mean = $9565 \mu\text{g L}^{-1}$, Table 1). The 40 % concentration difference was maintained between depths throughout 2012. In 2013, the difference was smaller, decreasing toward the end of the growing season (time x depth interaction: $P = 0.008$). The soil was drier in 2013, which induced greater CO_2 exchange with the atmosphere, resulting in lower concentrations at depth than in 2012. No significant interaction of depth with trees could be attributed to root systems that were distributed more deeply than the measurements (0-0.3 m depth).

Average CO_2 flux ($\text{CO}_{2\text{-flux}}$) across the site was $110 \mu\text{g m}^{-2} \text{s}^{-1}$ in 2012 and $101 \mu\text{g m}^{-2} \text{s}^{-1}$ in 2013 (Table 1). Seasonal and inter-annual $\text{CO}_{2\text{-flux}}$ ranged from over two orders of magnitude ($7.9\text{--}661 \mu\text{g m}^{-2} \text{s}^{-1}$), so that CVs were as high as 94% (Table 1). The fluxes were comparable to estimates ($5.1\text{--}312 \mu\text{g m}^{-2} \text{s}^{-1}$) from other conventional agricultural cropping systems in similar climates (Fang et al. 1998; Allaire et al., 2008, 2012), but variations in CM fluxes were slightly higher (i.e., CVs > 100 %). Other TBI systems incorporating hybrid poplars have reported a narrower range of CO_2 fluxes ($83\text{--}222 \mu\text{g m}^{-2} \text{s}^{-1}$; Peichl et al., 2006).

The high degree of variability in surface CO_2 emissions depended, in part, upon agricultural system (TBI vs CM, $P = 0.09$), but weakly so. In contrast, surface emission variability strongly influenced by the distance from the trees ($P = 0.0003$). Indeed, $\text{CO}_{2\text{-flux}}$ was up to 20 % higher under CM than under TBI in 2012, given that carbon turnover is typically higher under monoculture. On the St-Paulin site, soil temperature and $\text{CO}_{2\text{-soil}}$ were higher under CM in 2013. The reason for the former response is not

clear, but it may be due to the deep tillage that had redistributed soil C, resulting in greater food availability for soil organisms all across the field and increase in variability.

Distance from the trees significantly influenced $\text{CO}_2\text{-flux}$ but only in 2013 ($P = 0.0003$). Variation within the same treatment was greater in 2012 than in 2013, even though the overall variability across the site during summer 2013 was higher (Table 1, Fig. 3). Although the response was not significant in 2012, more CO_2 was emitted at a distance of 4 m from the central tree row in 2013 (0 m), i.e., within the crop alley where trees exerted the least influence (Fig. 3). The tree row-alley difference was more noticeable in May and June, which would explain the significant time $\times$ treatment interaction ($P = 0.002$).

As for $\text{CO}_2\text{-soil}$, $\text{CO}_2\text{-flux}$ varied from day to day ($P < 0.0001$). There was no interaction with the agricultural system in 2012, since the trend for higher fluxes under CM was maintained over the entire summer. However, the agricultural system interacted with time in 2013. TBI tended to emit more CO_2 than CM during the hottest periods and less during the coldest periods. When moisture was insufficient, such as in 2013, a reversal of the effects that trees had exerted on CO_2 emissions began when drawn-down soil moisture exceeded evapotranspirative demand, allowing soil temperature to rise, and gas diffusion and CO_2 emissions to increase. Higher emissions occurred in May and June (Fig. 3), nearly one month earlier than usually observed in Quebec (Allaire et al., 2008, 2012, 2015), most likely because of earlier spring onset, a thinner snow cover in 2012 (Fig. 2) and new available C released by deep tillage. Negative fluxes were sporadically measured. These responses mainly occurred under cold soil conditions during fall, as observed in other studies under cold conditions (Allaire et al., 2008, 2015).

Trees not only directly affected carbon emissions and soil C pools, but also exerted indirect effects through soil physico-chemical properties. For example, $\text{CO}_2\text{-soil}$ concentrations at 0.125 and 0.250 m depths were strongly inter-correlated ($r = 0.85$), while they were weakly correlated with the

resulting surface $\text{CO}_2\text{-flux}$ ($r = 0.25$). C_{Total} and C_{soluble} pools accounted for 1-2% of the variation in $\text{CO}_2\text{-soil}$, while soil temperature and moisture accounted for at least 40%. These two physical parameters drive soil microbial activities that result in CO_2 production (Fang and Moncrieff, 2001). Microbial and root activity increases as temperature rises, as does CO_2 production, when temperature, moisture and oxygen levels remain within physiological tolerable limits (Smith et al., 2003; Rivest et al., 2013). Increases in soil θ_v and temperature near the surface tended to increase $\text{CO}_2\text{-soil}$. Bulk density and K_{sat} influenced $\text{CO}_2\text{-soil}$, since these physical properties affect temperature and gas diffusion through the control over the moisture content. Chemical properties, such as nitrate concentration and pH, were also closely correlated with $\text{CO}_2\text{-soil}$ through their effects on microbial activities; carbon and nitrogen are necessary to meet microbial metabolic demands, while pH influences C and N availability (Bezdicsek et al., 2003). Consequently, $\text{CO}_2\text{-soil}$ was explained by $\text{N}_2\text{O-soil}$ and nitrate concentrations. Overall, physico-chemical soil properties accounted for up to 78 % of the total variation in $\text{CO}_2\text{-soil}$.

About 34% of the variation in $\text{CO}_2\text{-flux}$ could be explained by soil temperature, sand content, and $\text{CO}_2\text{-soil}$. As described earlier and also explained in several gas flux studies (Allaire et al., 2008, 2012, 2015; Oishi et al., 2013), $\text{CO}_2\text{-flux}$ is inherently correlated with $\text{CO}_2\text{-soil}$, as a function of Fick's law. Since N cycling influences microbial activity (Thornton et al., 2007), $\text{N}_2\text{O-soil}$ was negatively correlated with $\text{CO}_2\text{-flux}$ ($r = -0.376$, 0.125 m; $r = -0.397$, 0.25 m). These properties were all significantly influenced by the agricultural system and the presence of trees ($P < 0.01$).

TBI tended to have C_{Total} equal to that of CM, higher soil C_{soluble} , lower $\text{CO}_2\text{-soil}$ and $\text{CO}_2\text{-flux}$ than CM. These results highlight the environmental benefits of TBI on GHG emissions, even under organic production, together with the carbon stored in wood, which is discussed in another study (Alam et al., 2014; Kinans et al., 2014, 2015).

Nitrogen dynamics

Soil NO_3^- ranged from 1 to 47 mg kg^{-1} (Table 1), while NO_2^- was routinely below detection limits. Their combined concentrations were lower than those typically reported for conventional agricultural systems (Pennock et al., 1992; Clemens et al., 1999); and could be attributed to very low (organic) fertilization rates of the system compared to conventional applications of inorganic-N fertilizer. CVs of both anions ranged from about 60 to 75%, which were similar to those estimated under TBI production in other studies (Beaudette et al., 2010; Evers et al., 2010).

NO_3^- and NO_2^- were not influenced by agricultural system (TBI vs CM) in this study, but they were influenced by the distance from trees. The presence of leaf litter may explain the effect of trees on NO_3^- . Leaf litter is a source of complex N, which in turn may affect bacterial and fungi activity. Higher nitrate may be due to tree effects on soil moisture, particularly during springtime (tree distance, $P = 0.08$). Yet, the distance from the trees did not influence NO_2^- , when concentrations were detectable. Nitrite is an ephemeral compound and can be lost from soils through chemo-denitrification (as NO) and several other processes (Conrad, 1994). Correlations involving the two N species were harder to establish, as the amount of data was limited because they were measured only once during the entire experiment.

As generally observed in other GHG studies (Skinner et al., 2014), $\text{N}_2\text{O}_{\text{-soil}}$ was three orders of magnitude lower than $\text{CO}_2_{\text{-soil}}$ (Table 1). Mean $\text{N}_2\text{O}_{\text{-soil}}$ was about 2.4 $\mu\text{g L}^{-1}$ (minimum, 0.32 $\mu\text{g L}^{-1}$; maximum, 25.2 $\mu\text{g L}^{-1}$; Table 1), which corresponds to lower values encountered for wheat–maize cultivar rotations (5 to 600 $\mu\text{g L}^{-1}$, Wang et al., 2013). Under similar climate conditions, soil N_2O ranges from 285 to 563 $\mu\text{g L}^{-1}$ in forestry systems (Lavoie et al., 2013). Low concentrations and emissions of nitrous oxide were associated with low fertilizer and N inputs in the system. N_2O was always detected at both soil depths, but its values were highly variable (CVs up to 100%). This level of variability is frequently observed for N_2O in agricultural fields (Clemens et al., 1999; Desjardins et al., 2010; Skinner et al., 2014).

Agricultural system (system $\times$ time interaction, $P = 0.008$), together with distance from trees ($P = 0.002$), influenced N_2O_{soil} . TBI formed slightly more soil N_2O than the CM system early in the season (Fig. 4). Otherwise, similar seasonal, spatial, and depth trends were observed for N_2O_{soil} and N_2O_{Flux} . Higher emissions during spring were likely associated with leaf litter decay and release of organic and inorganic nitrogen, which is transported into the profile with snowmelt; this N input was not available in plots without trees. Otherwise, N inputs were very limited and similar among plots, resulting in similar N_2O dynamics between systems. Most N_2O that is formed within the uppermost 5 cm of soil (Tenuta and Sparling, 2011) is rapidly consumed by other microorganisms within the same layer (Wagner-Riddle et al., 2008); consequently, surface fluxes that we measured were low or frequently below detection limits (data not shown). Further, N_2O could only be consistently detected and measured at 0.15 m and 0.25 m depths. Differences in measurement depths and the low number of useful measurement events were among the variety of factors that affect N_2O formation and consumption and which made analysis and interpretation of treatment responses difficult.

Greater N_2O_{soil} concentrations were observed at the beginning and end of the growing season (Fig. 4), a seasonal trend that was opposite to that of CO_2_{soil} , and which was consistent with those of other studies (Desjardins et al., 2010). One main reason for higher N_2O concentrations during springtime was likely due to the activities of specific groups of bacterial N_2O producers. These bacteria tend to be more active under cold soil conditions (Wagner-Riddle et al., 2008). The CV that was associated with mean N_2O_{flux} reached a value of 1693% during the growing season (Table 1). This strong variability and range of values has also been observed in most studies (Clemens et al., 1999), and likely results from the complex nature of soil N_2O formation and emissions.

Although highly variable, temperature and moisture explained more than 55% of variability in N_2O_{soil} , consistent with other studies (Lellei-Kovács et al., 2011). Nitrate, BD, and pH explained an additional 6% of the variation. N_2O_{soil} increased with NO_3^- concentrations, since the latter is the

essential precursor for the denitrification process. Since the soil is relatively acidic, increased pH also favoured higher $N_2O_{\text{-soil}}$, probably because more nutrients became available and soil conditions were more conducive to nitrification (Bouwman et al., 2002). $N_2O_{\text{-soil}}$ decreased with increasing BD -- probably because of limited diffusion and oxygen import in the soil; greater anoxia taking available NO_3^- to N_2 . Soil WC, C_{Total} and clay content explained another 7% of the variation in $N_2O_{\text{-flux}}$, which agrees with Clemens et al. (1999). Clay content influenced N_2O formation by affecting the soil moisture regime, while soil carbon (C_{Total}) may interfere with N_2O formation (Paré and Bedard-Haughn, 2012). All of these parameters, except for clay content, were significantly influenced by the presence of trees.

CONCLUSION

Under organic production with few external inputs, we expected a small but significant effect of trees on carbon and nitrogen dynamics in soil. Although trees influenced most soil physico-chemical properties, their presence result in very small changes in C_{Total} , $N_2O_{\text{-soil}}$ and $N_2O_{\text{-flux}}$. Trees had stronger effects on $CO_{2\text{-soil}}$, while exerting a small effect on $CO_{2\text{-flux}}$ and C_{soluble} . Considering the financial savings relative to applying fertilizers, working the soil, preparing the seed bed, tree harvesting for short and long rotation and so on, considering that TBI tends to reduce emissions of these gases relative to CM and other environmental advantages, such as C sequestration in wood, not discussed here, therefore, installation of TBI systems with low input in this climate and under current land management practices could be a favourable means of applying a GHG mitigation strategy.

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Table 1. Descriptive statistics of C- and N-related parameters.

Property	Units	Year	Depth (m)	n	Mean ^b	Min	Max	CV (%)
^a C _{Total}	(g g ⁻¹)*100	2013	0.125	29	2.86	0.94	5.09	28.7
		2013	0.250	30	3.42	1.26	9.39	47.1
^a C _{Soluble}	mg kg ⁻¹	2013	0.125	30	266	152	466	30.0
		2013	0.250	29	241	115	382	29.4
CO ₂ -soil	µg L ⁻¹ soil air	2012	0.125	179	9.6 E3	636	3.0E4	49.6
		2012	0.250	180	1.4E4	1.9E3	3.4E4	46.0
		2013	0.125	265	9.0E3	2.0E3	4.0E4	62.3
		2013	0.250	266	1.1E5	1019	4.7E4	56.0
CO ₂ -flux	µg m ⁻² s ⁻¹	2012	0.0	143	110	7.93	297	53.7
		2013	0.0	262	101	-27.0	661	93.9
^a NO ₃ ⁻	mg kg ⁻¹	2013	0.125	30	14.2	1.06	47.4	76.4
		2013	0.250	29	9.96	1.14	30.2	65.2
^a NO ₂ ⁻	mg kg ⁻¹	2013	0.125	30	0.037	ND	0.12	72.6
		2013	0.250	29	0.037	ND	0.10	62.0
N ₂ O-soil	µg L ⁻¹ soil air	2013	0.125	234	2.30	0.32	19.6	86.1
		2013	0.250	233	2.48	0.41	25.2	98.4
N ₂ O-flux	µg m ⁻² s ⁻¹	2013	0.0	233	2.9E-3	-3.2E-1	4.1E-1	1693

^aC_{Total}, C_{soluble}, NO₃⁻ and NO₂⁻ were only measured in 2013. ND: not detected. CV: coefficient of variation (%). ^B The values are for TBI and CM plots combined.

Figure captions

Fig. 1. (a) Aerial view of the study site with locations of transects (coloured lines) and blocks (red rectangles), and (b) a schematic representation of a transect with its 5 sampling points.

Fig. 2. Average CO₂ soil concentrations (CO_{2-soil}) as influenced by (a) agricultural system (tree-based intercropping, TBI; monoculture, CM,) with the distance from the tree row averaged over the entire growing season; (b) agricultural system with time averaged over all treatments, and (c) the depth (0.125 vs 0.25 m) over two growing seasons (2012 and 2013) (Error bars: LMSE).

Fig. 3. Average CO₂ emission fluxes (CO_{2-flux}) as influenced by (a) agricultural system (tree-based intercropping, TBI; monocropping, CM), (b) average annual CO₂ fluxes (CO_{2-Flux}) influenced by the distance from the tree rows and agricultural system over two growing seasons (2012 and 2013), and (c) distance from tree rows. (Error bars: LMSE).

Fig. 4. Temporal variation during 2013 of (a) N₂O soil concentrations (N₂O_{-soil}) as influenced by the agricultural system (tree-based intercropping, TBI; monocropping, CM) averaged over distance from the tree row and replicates; and (b) N₂O emission fluxes (N₂O-Flux) during the 2013 season averaged over all treatments, distances from tree rows, and replicates. (Error bars: LMSE).

2012

2013