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Carbon sequestration in agricultural soils in the Cerrado region of the Brazilian Amazon

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ABSTRACT

The introduction of crop management practices after conversion of Amazon Cerrado into cropland influences soil C stocks and has direct and indirect consequences on greenhouse gases (GHG) emissions. The aim of this study was to quantify soil C sequestration, through the evaluation of the changes in C stocks, as well as the GHG fluxes (N_2O and CH_4) during the process of conversion of Cerrado into agricultural land in the southwestern Amazon region, comparing no-tillage (NT) and conventional tillage (CT) systems. We collected samples from soils and made gas flux measurements in July 2004 (the dry season) and in January 2005 (the wet season) at six areas: Cerrado, CT cultivated with rice for 1 year (1CT) and 2 years (2CT), and NT cultivated with soybean for 1 year (1NT), 2 years (2NT) and 3 years (3NT), in each case after a 2-year period of rice under CT. Soil samples were analyzed in both seasons for total organic C and bulk density. The soil C stocks, corrected for a mass of soil equivalent to the 0–30-cm layer under Cerrado, indicated that soils under NT had generally higher C storage compared to native Cerrado and CT soils. The annual C accumulation rate in the conversion of rice under CT into soybean under NT was 0.38 Mg ha^{-1} $year^{-1}$. Although CO_2 emissions were not used in the C sequestration estimates to avoid double counting, we did include the fluxes of this gas in our discussion. In the wet season, CO_2 emissions were twice as high as in the dry season and the highest N_2O emissions occurred under the NT system. There were no CH_4 emissions to the atmosphere (negative fluxes) and there were no significant seasonal variations. When N_2O and CH_4 emissions in C-equivalent were subtracted (assuming that the measurements made on 4 days were representative of the whole year), the soil C sequestration rate of the conversion of rice under CT into soybean under NT was 0.23 Mg ha^{-1} $year^{-1}$. Although there were positive soil C sequestration rates, our results do not present data regarding the full C balance in soil management changes in the Amazon Cerrado.

1. Introduction

Global warming and an increase in the concentration of greenhouse gases (CO_2, CH_4 and N_2O) in the atmosphere enhance the importance of the role of soils as a carbon (C) source or sink (Houghton, 2003; Smith et al., 2008).

Excluding carbonated rocks, soils constitute the largest surface C pool, approximately 1500 Pg, equivalent to almost three times the quantity stored in terrestrial biomass and twice the amount stored in the atmosphere (Bernoux et al., 2006). The soil C stock in the top 30 cm of soil is approximately 800 Pg C, which is almost the same amount as the C stored in the atmosphere (Cerri et al., 2006). Therefore, changes in soil use and management can lead to changes in C stocks (Lal, 1997; Six et al., 2002) and to increases in greenhouse gas emissions to the atmosphere (Bernoux et al., 2001; IPCC, 2007).

Management systems which increase soil C stocks and reduce soil C losses are essential for sustainable development. The use of conventional tillage (CT) systems, involving intensive tillage and disking, cause high soil organic matter (SOM) losses and increases the fluxes of CO_2 to the atmosphere (Resck et al., 2000). The adoption of conservative cultivation systems, such as no-tillage (NT), contributes to the maintenance of the structural integrity of soil aggregates, decreasing SOM oxidation, reducing soil degradation (Castro Filho et al., 2002), and thus reducing greenhouse gases (GHG) emissions to the atmosphere.

NT systems have the potential to mitigate CO_2 emissions, as crop rotations and the maintenance of crop residues on the soil

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surface lead to a slow and gradual decomposition of organic material, which associated to the soil mineral fraction, causes the accumulation of C in the soil (Bayer et al., 2000a; Amado et al., 2001). Several studies report a significant increase in SOM in NT systems compared to conventional tillage systems (Bayer and Mielniczuk, 1997a,b, 1999; Resck et al., 2000; Bayer et al., 2000b; Sá et al., 2001; Bayer et al., 2006). On the other hand, in a recent revision Baker et al. (2007) pointed out that in studies where soil was sampled deeper than 30 cm, conservation tillage has shown no consistent accrual of soil C, instead showing a difference in the distribution of soil C stocks, with higher concentration near the surface in conservation tillage and higher concentrations in deeper layers under CT.

Nevertheless, some studies indicate that NT can promote an increase in C stocks, an increase in soil N content and higher soil aggregate stability (Groffman, 1985). However, higher N_2O emissions (from denitrification) in soils cultivated under NT as compared to CT have been presented in the literature (Six et al., 2004). As with N_2O, soil CH_4 fluxes are influenced by N cycling. Nitrogen fertilization increases soil N cycling and soil N_2O emissions to the atmosphere (Keller et al., 1988), also resulting in a decrease in atmospheric CH_4 consumption by the soil (Keller et al., 1990; Mosier et al., 1991; Smith et al., 2008).

Moreover, Six et al. (2004) suggested that the potential to mitigate global warming with NT is only realized when practiced in the long term. However, most of the studies cited in the literature were performed in soils in subtropical and temperate climates. There are few evaluations of soil C sequestration the compare NT and CT in tropical climatic regions, particularly under the Cerrado ecosystem.

The aim of this study was to quantify soil C sequestration, through the evaluation of changes in C stocks, as well as greenhouse gas fluxes (N_2O and CH_4) during the conversion of Cerrado into agricultural land in the southwestern Amazon, while also comparing NT and CT systems. The concept applied in the present research is based on the definition given by Bernoux et al. (2005), in which "soil C sequestration" for a specific agro-ecosystem in comparison with a reference ecosystem, should be considered as the result (for a given period of time and portion of space) of the net balance of all greenhouse gases, expressed in C-CO_2 equivalents or C-equivalents, computing all emission sources at the soil–plant–atmosphere interface.

2. Materials and methods

2.1. Description of the study area

Soil C sequestration estimates were carried out at the União Farm (12°29′S, 60°00′W), a conventional working farm with an area of 3700 ha in the city of Vilhena in Rondonia State, Brazil (Fig. 1).

The native vegetation of the region is classified as Cerrado, subgroup Cerradão of the dense vegetation type (Ribeiro and Walter, 1998). A specific sequence and management of crops that represents common practices in the area after Cerrado clearing was studied. The experimental area is located within a regular farm and therefore was not set up for scientific purposes.

According to Köppen (1900) the climate is classified as Aw (humid tropical) with mean temperature of 23.1 °C and a minimum temperature of 18.0 °C during the coldest month. The region has a well-defined dry season (May–September) with a monthly rainfall below 10 mm, while the mean annual rainfall is 2170 mm. The mean altitude of the region is 600 m with undulating relief. The soil was classified as a Typic Hapludox (Dystrophic Red Yellow Oxisol) with very clayey texture (730 g clay kg^{-1} of soil).

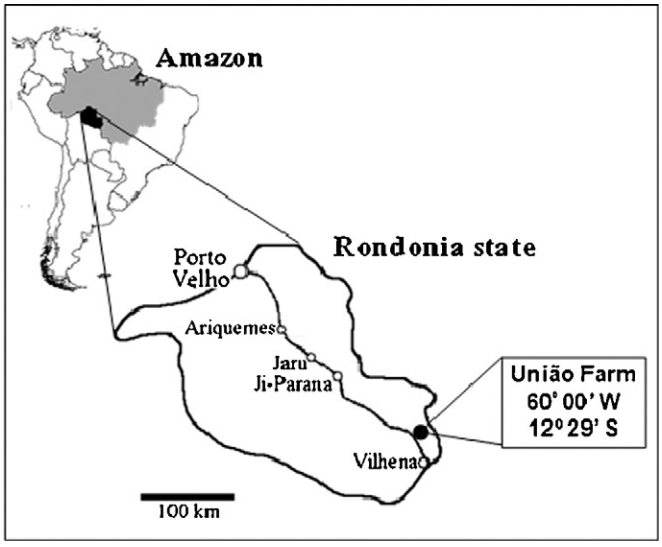

Fig. 1. Map of location of the study area in the União Farm, Rondônia State, Amazon, Brazil.

2.2. Conversion of native vegetation to agriculture and management strategies

From 1999 to 2004, areas of about 500 ha of the farm were cleared yearly for cultivation. The clearing was done by tractor and blade at the end of the wet season (May/June). After a drying period of 20 days, aboveground biomass was burnt. Mechanical windrowing followed this operation and areas were subsequently cleaned by burning stumps and root residues and removing remaining material. For further soil preparation, a disc harrow was used to incorporate dolomite lime, which was applied to achieve 50% base saturation (V) in the 0–20 cm soil layer. Next, a leveling harrow was used. These initial preparation steps had been applied to all sampled areas, except for the native Cerrado (used as control).

Every newly established area was cultivated with rice under CT. After 2 years of rice under CT and associated lime incorporation, leveling and cleaning of the soil surface, a NT system with soybean was introduced for 1–3 years. A chronosequence of six different sites was considered in this study: native Cerrado vegetation (CE), used as a reference area, a CT system cultivated with rice for 1 year (1CT) and 2 years (2CT), and a NT system cultivated with soybean for 1 (1NT), 2 (2NT) and 3 years (3NT), always preceded by a 2-year period of rice under CT alternating either with other crops or fallow land in the winter season (Table 1). This table also shows the crop cycles, annual application rates of lime, pH $CaCl_2$, available P and V in the 0–30 cm soil layer. Nitrogen fertilization rates and other nutrient additions in the study area are described in detail in Carvalho et al. (2007).

The areas were located in close proximity (less than 2 km apart from each other), with similar topography, soil and climate conditions, differing only in the time since clearing and the setting up of the sites. It is important to stress that those areas represent a chronosequence formed by farmer's fields. Therefore, it is expected that some spatial variability of soil attributes occur in those sites and would influence the results presented here.

2.3. Soil sampling and analysis

Soil sampling was carried out in July 2004 (dry season) and in January 2005 (wet season) in six areas of approximately 1 ha (100 m × 100 m) based on a completely randomized sampling design with five pseudo-replicates in each area. We are considering those as pseudo-replicates, since they came from the same evaluated areas.

Table 1
Cultivation history of the main crops (rice, soybean) and land use in the winter season in the corresponding cultivation periods under different land use practices annual lime application rates, pH $CaCl_2$, available P and base saturation in the 0–30-cm soil layer. Adapted from Carvalho et al. (2007).

Land use	Cultivation period	Main crop	Wintercrop	Lime (Mg ha^{-1})	Soil Density (g cm^{-3})	pH $CaCl_2$	Available P (mg dm^{-3})	V (%)
Continuous Cerrado				–	0.77	3.8	4.7	3.6
1CT	2003–2004	Rice (CT)	Fallow land	6	1.00	4.5	6.5	24.0
2CT	2002–2003	Rice (CT)	Fallow land	2	0.93	4.5	5.7	20.1
	2003–2004	Rice (CT)	Fallow land	4				
1NT	2001–2002	Rice (CT)	Fallow land	2	1.11	4.7	9.4	32.5
	2002–2003	Rice (CT)	Fallow land	2				
	2003–2004	Soybean (NT)	Maize	2				
2NT	2000–2001	Rice (CT)	Fallow land	2	0.98	5.0	15.4	39.1
	2001–2002	Rice (CT)	Fallow land	2				
	2002–2003	Soybean (NT)	Sorghum	1				
	2003–2004	Soybean (NT)	Millet	1				
3NT	1999–2000	Rice (CT)	Fallow land	1	1.14	5.4	29.7	58.9
	2000–2001	Rice (CT)	Fallow land	2				
	2001–2002	Soybean (NT)	Fallow land	1				
	2002–2003	Soybean (NT)	Maize	0				
	2003–2004	Soybean (NT)	Maize	2				

1CT and 2CT mean 1 and 2 years of rice under conventional tillage; 1NT, 2NT and 3NT mean 1, 2 and 3 years of soybean under no-tillage after a 2-year period of rice under conventional tillage; V means base saturation.

Soil samples were taken from five profiles at 0–5, 5–10, 10–20 and 20–30 cm depths in each site. For determination of soil bulk density (BD), undisturbed samples were collected using a steel cylinder (5 cm × 5 cm), according to Embrapa (2006). In the 10–20 and 20–30 cm soil layers the samples were taken in the middle part of the corresponding layer.

After air-drying, the samples were sieved at 2 mm. From each sample, 10 g were ground and sieved at 0.25 mm for determination of total organic carbon (TOC). The determination was carried according to Nelson and Sommers (1982) using a Carbon Analyzer, LECO® CN-2000.

2.4. Determination of equivalent soil layers and carbon stock calculation

Since the samples were collected in fixed layers, the stock calculation needs to be corrected for variations in soil BD after land use changes. Therefore, we used the methodology described in Ellert and Bettany (1996) and Moraes et al. (1996) to correct soil C stocks to an equivalent mass depth, i.e. the depth of the cultivated areas soil that contains the same mass of soil as the corresponding layer (0–30 cm) in CE (the reference area).

The calculation of the equivalent soil layer is described briefly as follows: (i) the weighted mean of BD in the respective soil layers in CE (M_{ce}) is calculated; (ii) the weighted mean of BD in the respective soil layers in each area (1CT, 2CT, 1NT, 2NT and 3NT) (M_{area}) is calculated and (iii) the equivalent layer for the area is determined according to Eq. (1), in which the value 30 is related to the 0–30 cm soil depth under CE.

$$\text{equivalent soil layer (cm)} = \frac{M_{ce}}{M_{area}} \times 30 \quad (1)$$

For each soil layer (0–5, 5–10, 10–20 and 20–30 cm) the C stocks (Mg ha^{-1}) were calculated by multiplying the element concentration (%) by the soil BD (g cm^{-3}) and to the equivalent soil layer thickness (cm).

2.5. Trace gas fluxes

Measurements of emissions of CO_2, N_2O and CH_4 from soils were made in July 2004 and January 2005 in the six areas included in the chronosequence. Fluxes were measured over two consecutive days at each sampling time. This short period of gas flux measurements represent a limitation for our study. However, given the logistics to obtain the data from a remote region, we had to assume that the measurements made on 4 days were representative of the whole year.

Gas emission measurements were made using a two-piece static polyvinyl chloride plastic chamber (Steudler et al., 1989). Three chamber-bases separated from each other by approximately six meters were inserted 2 cm into the soil 1 day before the first measurements, at each of the six sites. Initial headspace gas samples were collected using 20-ml nylon syringes at the beginning of the incubation and at 5, 10 and 20 min thereafter (Feigl et al., 1995). Syringes were flushed three times with chamber air to mix the chamber headspace atmosphere and after sampling, and were compressed to 15 ml to improve storage times (Steudler et al., 1996).

Ambient air (at 1 m from soil surface), chamber air and soil temperature at 2.5, 5, and 10 cm depths were measured at one representative chamber for all areas. Barometric pressure was measured at the beginning of each incubation. Gas samples were sent to the laboratory and analyzed immediately.

The concentrations of the CO_2 and N_2O were determined using gas chromatography with a ^{63}Ni electron capture detector operated at 230 °C (Steudler et al., 1991; Feigl et al., 1995). The CH_4 concentrations were determined using gas chromatography with a flame ionization detector (Steudler et al., 1989). The fluxes of each gas were calculated by measuring alteration in concentration as a function of incubation time.

2.6. Conversion of N_2O and CH_4 fluxes into C-equivalent

Conversion to C-equivalents is necessary since each GHG has a different global warming potential (GWP). N_2O fluxes are estimated at a rate of μg m^{-2} h^{-1}, but have a GWP 296 times larger than CO_2, while CH_4 fluxes are in a similar order of magnitude as CO_2 (mg m^{-2} h^{-1}), but with a GWP 23 times larger than CO_2 (IPCC, 2003). The sequence of calculations in the conversion of N_2O and CH_4 fluxes into C-equivalent is presented in Eq. (2).

The C-equivalents for N_2O and CH_4 are calculated as follows:

$$\mathrm{Ceq}(N_2O) = N_2O \times \left(\frac{44}{28}\right) \times \left(\frac{12}{44}\right) \times 296, \qquad \mathrm{Ceq}(CH_4) = CH_4 \times \left(\frac{16}{12}\right) \times \left(\frac{12}{44}\right) \times 23 \quad (2)$$

where $N_2O = N_2O$ flux in the area, $CH_4 = CH_4$ flux in the area, (44/28) = ratio between the molecular weights of N_2O and N, (12/44) = ratio between the molecular weights of C and CO_2, (16/12) = ratio between the molecular weights of CH_4 and C, 296 = global warming potential of N_2O in relation to CO_2, 23 = global warming potential of CH_4 in relation to CO_2.

2.7. Calculation of soil carbon sequestration

As commented before, in order to calculate soil C sequestration, it is necessary to obtain data not only on soil C stocks, but also on trace gas emissions. In this study, with the GHG fluxes converted into C-equivalents, the soil C sequestration rate was expressed in Mg ha^{-1}. The soil C sequestration was determined by subtracting the N_2O and CH_4 fluxes in C-equivalents from the soil C accumulation rates. The CO_2 fluxes were not taken into account, since they are already included in the global balance of C in the soil–plant–atmosphere system, being considered in the soil C stock changes (Cerri et al., 2004; Bernoux et al., 2005). Nevertheless, CO_2 fluxes were included in the present work in order to illustrate the dynamics of this gas through time in different management practices.

2.8. Statistical analysis

The statistical analysis of data was performed on a completely randomized sampling design, with the assumption that the areas studied had the same topographic, edaphic and climatic conditions. Six areas, each with five pseudo-replicates, were evaluated.

Data from soil C stocks under different areas were analyzed for variance (ANOVA) to determine land use effects. A Tukey test was used to test for significant ($p \leq 0.05$) differences among treatments. All statistical analyses were performed using the SAS program, version 6.

3. Results and discussion

3.1. Soil carbon stocks

During the dry season, significant differences in soil C stocks as a function of land use change were observed (Table 2). In the 0–10-cm layer, the stock in CE was 21.5 Mg C ha^{-1}. After converting Cerrado into agricultural land, the soil C stock in 1CT was 16.3 Mg ha^{-1}, significantly smaller ($p < 0.05$) than the stocks in CE, 1NT and 3NT. Other studies have also shown the conversion of native Cerrado through slash and burn followed by soil cultivation to result in a decrease in C stocks associated with an increase in CO_2 emissions (Detwiler, 1986; Brown and Lugo, 1990; Lal, 2003; Franchini et al., 2007; Smith et al., 2008). Two years of rice cropping (2CT) raised the soil C stock to 21.3 Mg ha^{-1}, which was not statistically different than the other areas. The C stocks in the 0–10 cm layer under NT were 23.3, 21.3 and 23.9 Mg ha^{-1}, for the 1NT, 2NT and 3NT, respectively. In the areas under NT, there was a trend (though not statistically significant) for increase in soil C stocks as compared to the CE. Probably, due to the absence of tillage operations (i.e., NT), the lower decomposition rates favored the maintenance and accumulation of SOM (Bayer and Mielniczuk, 1999; Resck et al., 2000; Green et al., 2007) and guaranteed a continuous flow of subtracts and energy through the soil (Roscoe, 2005), leading to an increase in C stocks. The measured C stock data showed a high level of variation, which could be associated with the short amount of time since adoption of the NT system.

Table 2
Soil C stocks (Mg ha^{-1}) in the equivalent layers of 0–10 and 0–30 cm in the dry (July 2004) and wet (January 2005) seasons under Cerrado (CE), conventional tillage (1CT and 2CT) and no-tillage (1NT, 2NT and 3NT) in Vilhena, Rondonia State, Brazil.

Area	Soil C Stocks (Mg ha^{-1})			
	Soil equivalent layer (10 cm)		Soil equivalent layer (30 cm)	
	Dry season	Wet season	Dry season	Wet season
CE	21.5 ± 3.2a	21.3 ± 0.7a	50.0 ± 7.4b	48.1 ± 2.6a
1CT	16.3 ± 1.7b	17.8 ± 4.3a	47.6 ± 4.9b	47.4 ± 7.2a
2CT	21.3 ± 2.6ab	24.2 ± 6.4a	55.4 ± 8.5ab	58.5 ± 11.0a
1NT	23.3 ± 2.9a	24.2 ± 6.6a	66.5 ± 6.5a	65.6 ± 15.4a
2NT	21.3 ± 2.0ab	19.0 ± 5.2a	54.5 ± 5.6ab	47.4 ± 7.9a
3NT	23.9 ± 3.0a	19.5 ± 6.8a	67.5 ± 10.3a	59.0 ± 18.5a
LSD[a]	5.14	10.57	14.49	22.87
CV%[b]	12.4	25.7	13.0	21.5

The results are mean ($n = 5$) ± standard deviation. Means followed by the same letter are not significantly different according to Tukey's test at 5%.
[a] Least significant difference.
[b] Coefficient of variation.

In the 0–30-cm soil layer, the C stock in CE was 50 Mg ha^{-1}, significantly smaller than the stocks in 1NT and 3NT ($p < 0.05$), in the dry season. Corazza et al. (1999), studying a clayey Typic Hapludox under Brazilian Cerrado vegetation, measured a soil C stock in the 0–20-cm layer of 39.8 Mg ha^{-1}. Resck et al. (2000) measured in a Typic Hapludox under Brazilian Cerrado a C stock of 61 Mg C ha^{-1} in the 0–30-cm soil layer. In a Rhodic Hapludox with very clayey texture under Cerrado in Dourados (Mato Grosso do Sul State, Brazil), Salton et al. (2005) measured a soil C stock of 44.5 Mg ha^{-1} in the 0–20-cm layer. Bayer et al. (2006) reported a C stock of 54 Mg ha^{-1} in the 0–20-cm layer of a Typic Hapludox (650 g clay kg^{-1} soil) under Cerrado in Brazil. Despite the soil C contents were similar among the mentioned studies (ranging from 2.5 up to 3.1% comparable to the 2.9% of C for this research) the soil BD obtained here (weighted mean of 0.77 g cm^{-3} in the 0–30 cm soil depth) was lower than the values reported by Salton et al. (2005) and Bayer et al. (2006). Therefore, we suggest that the lower soil C stocks presented here are due to the lower BD compared to the last two studies cited above.

After the conversion of Cerrado into agricultural land, while the soil C stock in 1CT (47.6 Mg ha^{-1}) was significantly smaller than the stocks in 1NT and 3NT ($p < 0.05$), it is not statistically different from 2NT (Table 2).

In the wet season, 6 months after the first soil sampling, there were no significant differences among the areas in both the 0–10 cm and the 0–30 cm soil layers (Table 2).

Average soil C stocks in the 0–30 cm were calculated (Fig. 2) using the data of the two evaluated sampling times (dry and wet seasons presented in Table 2). When the average soil C stock was considered, some significant differences were observed. The C stock in 1CT was significantly smaller ($p < 0.05$) than the stocks in 2CT, 1NT and 3NT.

The average soil C stocks in 2CT and in the NT areas were used to calculate the annual C accumulation rates in the conversion of rice cropping under CT into soybean cropping under NT.

The C accumulation rate for the conversion of rice under CT into soybean under NT was obtained using the data shown in Fig. 2. Considering the average soil C stock values, the annual accumulation rate was 0.38 Mg ha^{-1} year^{-1}, as expressed in the equation $y = 0.38x + 58.34$.

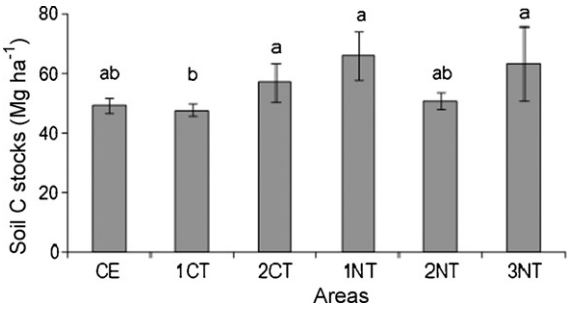

Fig. 2. Soil C stocks in the land use change areas in Vilhena, Rondonia State. The stocks are referred to the equivalent soil mass of 30 cm depth under Cerrado. CE (Cerrado); 1CT and 2CT (1 and 2 years of rice under CT); 1NT, 2NT and 3NT (1, 2 and 3 years of soybean under NT, after 2 years of rice under CT). Values are means ($n = 10$) ± the standard deviation. Averages followed by the same letter are not different by the Tukey test at 5%.

The C accumulation rates in this study are in agreement with results observed by other authors in soils under tropical Cerrado. Bayer et al. (2006), evaluating the C accumulation in areas under NT with a soybean/maize succession, reported annual accumulation rates of 0.30 and 0.60 Mg ha^{-1} year^{-1}, in areas with silty and clayey soils, respectively. In a Rhodic Hapludox under Cerrado in the State of Goiás, Brazil, Freitas et al. (2000) observed the rate of 0.25 Mg ha^{-1} year^{-1} in an area under NT with a maize/bean succession. Corazza et al. (1999), studying a clayey Oxisol in central region of Brazil reported an increase of 0.63 Mg of C ha^{-1} year^{-1} under NT.

On the other hand, Roscoe and Buurman (2003) did not observe an accumulation of C in NT as compared to CT in a Typical Haplustox with very clayey texture under Cerrado in Minas Gerais State, Brazil. The annual accumulation of soil C could vary regionally due to climate, soil, management, and mainly time since conversion to agriculture and time since adoption of NT. The accumulation of C in NT is very slow, and could take many years to reach expressive values (Bayer and Mielniczuk, 1999).

Moreover, studies carried out under temperate climate regimes have shown that in soils under NT soil C is concentrated in the surface layers, while in CT systems soil C is distributed deeper in the profile, so that apparent soil C gains from NT that are based only on near-surface samples disappear when deeper samples are also included (Dolan et al., 2006; Baker et al., 2007).

In recently converted areas such as in this study, the annual soil C accumulation could be attributed to the sum of several pools, such as the C in the form of remaining SOM from the Cerrado vegetation and the C derived from the original root system decomposition. After land-use-change, when the aboveground biomass is removed by slashing and burning, part of the root system remains in the soil and is decomposed and incorporated into the SOM, temporarily increasing the soil C stock. A third soil C pool deals with the inorganic C introduced into the soil by liming application, practice commonly applied after land use change in this study area. Finally, another important pool corresponds to the C introduced by the crop residues. Estimates by Cesar (1980) and Meirelles and Henriques (1992) showed that the residue added by aboveground native Cerrado vegetation varies from 1.2 to 3.7 Mg ha^{-1} year^{-1}, which is lower than the estimated mean annual residue addition by maize and soybean crops (4.0 and 4.3 Mg ha^{-1} year^{-1}, respectively). As it is well known, in the CT the residues are incorporated into the soil through plowing, whereas in the NT system the crop residues are maintained in the soil surface and the C will be gradually introduced into the soil.

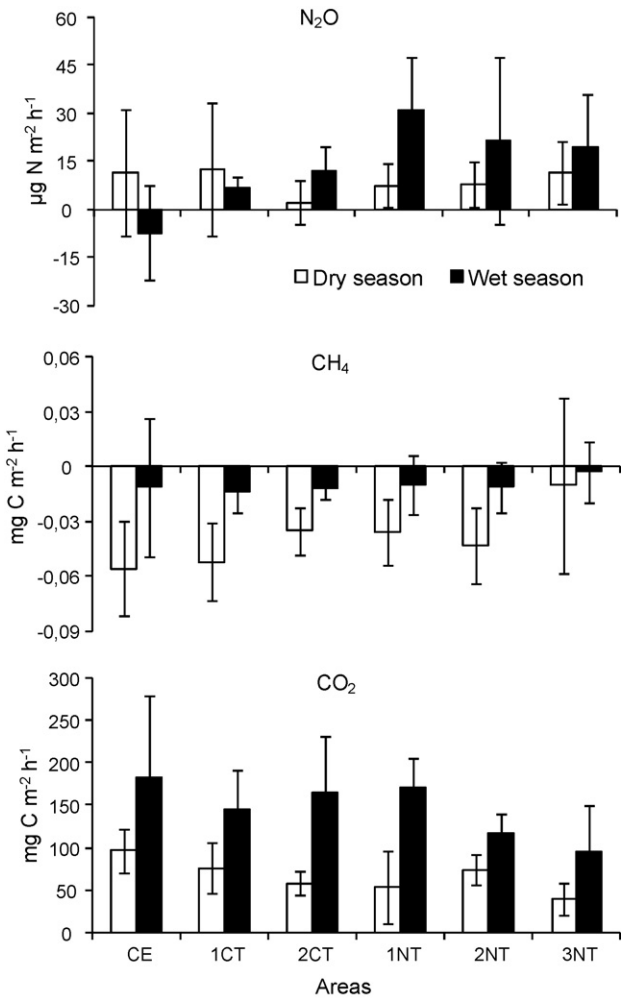

Fig. 3. Trace gas fluxes (CO_2, N_2O and CH_4) for dry (open bar) and wet (closed bar) seasons in the Cerrado (CE); conventional tillage (1CT and 2CT) and no-tillage (1NT, 2NT and 3NT) in Vilhena, Rondonia State, Brazil. Values are means ($n = 6$) ± the standard deviation.

3.2. Trace gas fluxes

In the dry season, the measured CO_2 and N_2O emission rates in the CE were 97.8 mg m^{-2} h^{-1} and 11.7 μg m^{-2} h^{-1}, respectively. The smallest CO_2 emissions were measured in the 1NT and 3NT sites (Fig. 3).

The average CO_2 emissions for the CE, CT and NT areas were 97.8, 67.8 and 56.5 mg C-CO_2 m^{-2} h^{-1}, respectively, in the dry season. The almost 17% lower emission rate from the NT system in relation to the CT would suggest a potential of the former to mitigate CO_2 emissions. Moreover, lime application is part of the land use change procedure and would certainly have implications for overall CO_2 emissions (IPCC, 2006 which includes CO_2 emission equation from lime application), but those fluxes were not evaluated in the present research.

Nevertheless, no soil disturbance, crop rotations and crop residue maintenance on the soil surface allow a slow and gradual decomposition of organic material, which, associated with the soil mineral fraction, leads to SOM accumulation, and thus, to an increase in soil C stocks (Bayer et al., 2000b; Amado et al., 2001; Sá et al., 2001). It is important to stress that the CO_2 emissions presented here express not only the soil respiration but also the root respiration. Thus, the direct equivalence between CO_2 emissions and C stock changes must be viewed with caution.

In 1CT, the N_2O emission rate was similar to the CE. This could be associated with the vegetation clearing and conventional soil tillage, which would have accelerated SOM mineralization, increasing the availability of soil mineral N, and therefore increasing soil N_2O emissions to the atmosphere. According to Stouthamer (1988), the N_2O produced by denitrifying soil organisms is controlled by the supply of nitrate and organic C, and by the availability of O_2 in the soil.

Despite the relatively large standard deviations, data suggest an increase in the N_2O emissions under NT compared to CT during the wet season (Fig. 3). According to Groffman (1985), NT, as compared to CT, increases soil C and N stocks, improving soil aggregate stability, leading to a rise in N_2O emissions from denitrification. The better soil aggregation, higher soil microporosity and water content favor the formation of anoxic microsites. The higher N_2O emissions under NT can also be related to N fixation by the soybean crop, which increases soil N content. Nobre (1994) observed an increase in N_2O emissions during the first 100 days after the establishment of soybean in tropical soils, probably due to higher N fixation in this crop.

Studies done by Six et al. (2002) compared NT and CT, from which the authors reported fluxes that were 3 kg N-N_2O ha^{-1} $year^{-1}$ higher in NT than in CT and pointed out that although this amount seems low when compared to the N input via fertilizer and crop residues, it can represent a considerable value from an environmental standpoint, since the GWP of N_2O is 296 times higher than the GWP of CO_2.

The CH_4 oxidation rates, evidenced by the measured negative fluxes, tend to be more intense under natural conditions, in which there are more appropriate conditions for the growth of methanotrophic bacteria (Duxbury et al., 1989).

In the wet season, the CO_2 emissions were about twice those in the dry season (Fig. 3). There was a trend for a reduction in CO_2 fluxes during the same time following land use change, in which emissions under CE, CT and NT areas were 183.6, 156.3 and 128.2 mg C m^{-2} h^{-1}, respectively.

The N_2O emissions in the wet season were different from those measured in the dry season. The reduction of N_2O into N_2 in the soil was shown by the negative fluxes in the CE area. The N_2O emissions were higher in the areas under NT, and in the wet season. In regions with marked dry and wet seasons, the higher soil water content has a positive effect on N_2O emissions to the atmosphere (Davidson et al., 1993).

In both seasons, and in all treatments, CH_4 fluxes were negative indicating that the soil was a sink for methane. Dry season negative fluxes were all greater than those in the wet season. Wet season fluxes were close to zero, indicating a balance between methanogenesis and methanotrophy activities. In the dry season there was a tendency for CH_4 sink strength to be greatest in the natural Cerrado soil, but none of the differences between treatments were statistically significant.

3.3. Conversion of N_2O and CH_4 fluxes into C-equivalents

N_2O and CH_4 fluxes expressed as C-equivalents (kg ha^{-1} $year^{-1}$) are presented in Table 3 on the assumption that the fluxes measured on 2 days in the dry and wet seasons were representative if each whole season. The limitation of this assumption is recognized but it is made as the best approach in the absence of additional data.

There was a balance between C-equivalents emission and consumption rates in the CE, since the N_2O emission values were similar to the CH_4 absorption rates in C-equivalents. The close to null value in C-equivalents in the soil under native vegetation confirms that the C stocks under native vegetation do not change with time, being in steady-state equilibrium.

Table 3
Conversion of N_2O and CH_4 fluxes into C-equivalent (kg ha^{-1} $year^{-1}$) in the soil–atmosphere system. The values represent the two seasons when samples were taken and the average of the two.

Sampling times	Areas					
	CE	1CT	2CT	1NT	2NT	3NT
N_2O in C-equivalent						
Dry season	130.46	137.91	25.56	83.23	86.79	128.46
Wet season	−81.34	76.46	134.35	346.38	240.37	214.14
Mean	24.56	107.18	79.96	214.81	163.58	171.30
CH_4 in C-equivalent						
Dry season	−41.03	−38.10	−25.64	−26.38	−31.50	−7.33
Wet season	−8.06	−10.26	−8.79	−7.33	−8.06	−2.20
Mean	−24.54	−24.18	−17.22	−16.85	−19.78	−4.76
Total emissions in C-equivalent						
Dry season	89.43	99.81	−0.08	56.86	55.29	121.14
Wet season	−89.40	66.20	125.56	339.06	232.31	211.94
Mean	0.02	83.00	62.74	197.96	143.80	166.54

CE (Cerrado); 1CT and 2CT (1 and 2 years of rice under CT); 1NT, 2NT and 3NT (1, 2 and 3 years of soybean under NT, after 2 years of rice under CT).

The annual N_2O and CH_4 emissions converted into C-equivalents were about 83, 63, 198, 144 and 167 kg ha^{-1} for 1CT, 2CT, 1NT, 2NT and 3NT, respectively. Considering that the values of C-equivalent presented in Table 3 were the same during all the years after conversion of Cerrado into agricultural land, the following values of C-equivalents were obtained: 83, 146, 344, 488 and 654 kg of C ha^{-1} for 1CT, 2CT, 1NT, 2NT and 3NT, respectively. In order to calculate the soil C sequestration, these C-equivalents values were subtracted from the C stocks presented in Fig. 2.

3.4. Soil carbon sequestration

The annual amount of C sequestered or emitted by the soil was obtained by subtracting the N_2O and CH_4 emission rates in C-equivalent from the soil C stocks, considering the assumptions and trends discussed before.

The average annual C sequestration rate estimated for the conversion of rice under CT into soybean under NT was 0.23 Mg ha^{-1} ($y = 0.23x + 58.33$). The N_2O and CH_4 emissions converted into C-equivalent, subtracted by the soil C stocks, resulted in a lower C sequestration rate when compared to the annual soil C accumulation rate.

According to Bernoux et al. (2006), caution must be taken when analyzing NT systems in terms of C sequestration. Comparisons should not be limited only to C storage in the soil but should also include associated trace gas fluxes. Associated fluxes of CH_4 and N_2O may change the final balance of C-CO_2 equivalents, based on the GWP of each gas.

Six et al. (2004) suggested that true global warming mitigation is only possible if the overall impact of NT adoption reduces the net GWP determined by fluxes of the three major biogenic GHG. The same authors also pointed out that newly converted NT systems increase GWP relatively to CT practices, in both humid and dry climate regimes, and longer-term adoptions, i.e., more than 10 years, only significantly reduces GWP in humid climates.

The present study aimed to estimate soil C sequestration when agricultural land under CT is converted to a more conservative management practice, i.e., NT. The native CE site was used only as a reference of original levels soil C stocks, but not included in the soil C sequestration rate estimates. Therefore, we did not take into account the C losses associated to the above ground biomass, which was slashed and burned in the conversion of Cerrado into agricultural land. However, studies that aim to perform a full balance of C in the soil–plant–atmosphere system should include

such emissions due to deforestation. An estimate of the amount of C exported from the area as native vegetation was cleared and plant residues were burned would almost certainly result in a negative C balance, with soil C sequestration, despite large C emissions in the implementation phase. Several studies have discussed such important issue (Fearnside et al., 1993; Kauffman et al., 1995; Graça et al., 1999; Dias-Filho et al., 2001). For instance, Kauffman et al. (1995) reported C emissions from cutting and burning a tropical forest in the Amazon ranging from 58 to 102 Mg ha^{-1}, while Dias-Filho et al. (2001) obtained emissions of 100–200 Mg ha^{-1} of C from above ground biomass in similar native vegetation in the Brazilian Amazon.

Moreover, conversion of native vegetation to agricultural land cause other negative effect, undermining the capacity of ecosystems to sustain food production, maintaining freshwater and forest resources, regulating climate and air quality, ameliorating infectious diseases, along with considerable losses of biodiversity (Foley et al., 2005).

4. Conclusions

The introduction of crop management practices after conversion of Amazon Cerrado was shown to influence soil C stocks, and have direct and indirect consequences on GHG emissions.

The soil C stocks indicated a trend of higher storage under NT compared to native Cerrado and soils under CT. Although the CO_2 emissions were not used to estimate C sequestration, in order to avoid double counting, the emissions of this gas tend to be smaller in the areas under NT, as compared to the CT areas and the Cerrado.

The annual C accumulation rate in the conversion of rice under CT into soybean under NT was 0.38 Mg ha^{-1} year^{-1}. However, when N_2O and CH_4 emissions in C-equivalent were subtracted, the soil C sequestration rate of the conversion of rice under CT into soybean under NT was 0.23 Mg ha^{-1} year^{-1}.

Although there were positive soil C sequestration rates, our results do not include a full C balance in soil management changes in the Amazon Cerrado. In order to better understand and monitor the full account of C in the soil–plant–atmosphere, it will be necessary to include not only the soil C sequestration rates, but also all GHG emissions (expressed in C-equivalents) resulting from land use change (deforestation of native vegetation through slash and burn), and the GHG emissions from fuels, electric power, fertilizers, pesticides, and other components of the production process.

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