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ABSTRACT

The impact of agricultural practices on CO₂ emissions from soils needs to be understood and quantified to enhance ecosystem functions, especially the ability of soils to sequester atmospheric carbon (C), while enhancing food and biomass production. The objective of this study was to assess CO₂ emissions in the soil surface following tillage abandonment and to investigate some of the underlying soil physical, chemical and biological controls. Maize (*Zea mays*) was planted under conventional tillage (T) and no-tillage (NT), both without crop residues under smallholder farming conditions in Potshini, South Africa. Intact top-soil (0–0.05 m) core samples (N = 54) from three 5 × 15 m² plots per treatment were collected two years after conversion of T to NT to evaluate the short-term CO₂ emissions. Depending on the treatment, cores were left intact, compacted by 5 and 10%, or had surface crusts removed. They were incubated for 20 days with measurements of CO₂ fluxes twice a day during the first three days and once a day thereafter. Soil organic C (SOC) content, soil bulk density (ρ_b), aggregate stability, soil organic matter quality, and microbial biomass and its activity were evaluated at the onset of the incubation. CO₂ emissions were 22% lower under NT compared with T with CO₂ emissions of 0.9 ± 0.10 vs 1.1 ± 0.10 mg C–CO₂ g C⁻¹ day⁻¹ under NT and T, respectively, suggesting greater SOC protection under NT. However, there were greater total CO₂ emissions per unit of surface by 9% under NT compared to T (1.15 ± 0.03 vs 1.05 ± 0.04 g C–CO₂ m⁻² day⁻¹). SOC protection significantly increased with the increase in soil bulk density ($r = 0.89$) and aggregate stability (from 1.7 ± 0.25 mm to 2.3 ± 0.31, $r = 0.50$), and to the decrease in microbial biomass and its activity ($r = -0.59$ and -0.57 , respectively). In contrast, the greater NT CO₂ emissions per m² were explained by top-soil enrichment in SOC by 48% (from 12.4 ± 0.2 to 19.1 ± 0.4 g kg⁻¹, $r = 0.59$). These results on the soil controls of tillage impact on CO₂ emissions are expected to inform on the required shifts in agricultural practices for enhancing C sequestration in soils. In the context of the study, any mechanism favoring aggregate stability and promoting SOC allocation deep in the soil profile rather than in the top-soil would greatly diminish soil CO₂ outputs and thus stimulate C sequestration.

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1. Introduction

The soil C pool, which is the largest terrestrial pool, storing 2344 Pg C (1 Pg = 10¹⁵ g = 1 billion tonnes) in the top three meters of the soil (Jobbagy and Jackson, 2000) exhibits direct and dynamic exchanges with the atmosphere through photosynthesis and organic matter decomposition. Because the soil C pool has been

drastically reduced by human activities, by approximately 66% (Lal, 2003), C sequestration in soils is thought to potentially mitigate the rising atmospheric CO₂ concentration (Batjes, 1996; Lal, 2003). Improving current land use and land management has the potential to reduce the rate of anthropogenic carbon loss in agricultural soils.

Among the land management practices commonly cited that could influence soil organic C (SOC) stocks are soil tillage, chemical and organic fertilizations, crop rotation, crop associations and planting density (Lal, 2003).

No-tillage (NT), where mechanical soil disturbance is minimized and permanent organic soil cover consisting of a growing crop or mulch of crop residues is maintained, is currently being promoted under both commercial and smallholder agriculture as it contributes to reduced labor and energy inputs (Huggins and Reganold, 2008). In addition, by keeping crop residues on the soil surface longer than under conventional tillage (T), NT has been demonstrated to prevent soil surface sealing (Bradford and Huang, 1994; Rao et al., 1998), reduce splash erosion (Bradford and Huang, 1994; Choudhary et al., 1997), and to enhance soil porosity and soil infiltration through improved biological activity (Doran, 1980; Granatstein et al., 1987; Meek et al., 1992; Edwards et al., 1992; Pierce et al., 1994; Feng et al., 2003; Gosai et al., 2010). In addition, the absence of soil disturbance in the NT system fosters SOC protection within aggregates, an important factor of C sequestration (Bolliger et al., 2006; Triplett and Dick, 2008).

However, despite numerous investigations, the impacts of tillage on soil C losses and especially CO₂ emissions have been largely inconsistent. Several studies have reported lower CO₂ emissions under NT (Fang and Moncrieff, 2001; Ussiri and Lal, 2009; Chaplot et al., 2012). For instance Reicosky and Archer (2007) reported cumulative 20 days emissions of 237 g CO₂ m⁻² for NT vs 891 g CO₂ m⁻² for T in Minnesota, USA. Similarly, on a clayey tropical soil from the Amazonian Basin, Carmo et al. (2007) reported that tillage cessation lowered CO₂ emissions from 235 to 181 mg CO₂-C m⁻² h⁻¹. However, several authors reported greater soil CO₂ emissions from NT soils (Aslam et al., 2000; Aslam et al., 2000; Carmo et al., 2007; Barreto et al., 2009; Lee et al., 2009; Sainju et al., 2010; Smith et al., 2011), among which Barreto et al. (2009) reported in Brazil an increase of CO₂ emissions by 700%.

The lower gaseous emissions from NT soils are assumed to come from the greater organic matter protection from microbial decomposition within undisturbed soil aggregates (e.g., Six et al., 2002). Other factors would however have to be considered to explain for instance the greater NT C losses such as soil compaction, soil surface crusting, organic matter quantity and quality, biological activity among others (Six et al., 2002; Von Lützow et al., 2006; Elder and Lal, 2008; Kögel-Knabner et al., 2008) and these need further investigation. Moreover, several authors have shown that changes in soil properties, especially the redistribution of organic carbon within soil profiles (e.g., Baker et al., 2007; Luo et al., 2010; Hou et al., 2012) require certain duration to occur, while others such as soil crusting can occur during a single rainstorm.

The relative contribution of these physical, biological or chemical factors of control to explain the changes in CO₂ emissions following tillage abandonment need further appraisal. What are the main soil factors that control changes in tillage-induced CO₂ emissions and their relative contribution? This constitutes an important research question that comes as a prerequisite to better understand the effects of land management on the global C cycle as well as for designing agricultural practices aiming at sequestering more C into soils while improving their overall functioning.

Consequently, the main objective of the present study was to investigate the underlying controls of the changes in soil CO₂ emissions consecutive to tillage abandonment. A research trial was

established in an agricultural maize field of South Africa to investigate the short-term (two years) consequences of tillage abandonment on CO₂ emissions and soil properties. In order to focus on soil CO₂ emissions and to avoid interferences with emissions from the mulch, we considered a system where crop residues are exported from fields. Additionally, this mimics the smallholder farming systems where, due to competing resource use, crop residues are used as livestock feed. The emissions from undisturbed 0 to 0.05 m T and NT soil samples (N=33) were compared to their SOC content, soil bulk density, SOC stocks, dissolved organic C quality, microbial biomass and activity, aggregate stability, and soil surface crusting. Additionally, undisturbed soil samples were compacted by 5–10% and some had their crust removed in an attempt to further the understanding of the controls of CO₂ outputs from soils.

2. Materials and methods

2.1. Study area

The study was conducted in a smallholder farming community of Potshini located 10 km south of the town of Bergville (Long: 29.38°; Lat: -28.81°), within the sloping lands of the KwaZulu-Natal province of South Africa.

Following the classification of Köppen, the climate of the area is temperate with cold dry winters and rainy summers (Peel et al., 2007). The mean annual precipitation is 684 mm, the potential evaporation is 1600 mm and the mean annual temperature is 13 °C. In the landscape, steep slopes with shallow soils are used for cattle grazing, whereas the lowland areas, characterized by gentle slopes (0–10%) and deep soils (>1.5 m), are used predominantly for rain-fed cropping with maize (*Zea mays*) production as the main crop in KwaZulu-Natal. Maize is generally planted around mid-November on lands prepared with draft oxen, but mechanization is becoming more common. Planting, weeding and harvesting are done manually. Little fertilizer is used due to the limited access to markets and lack of funds.

Soils are acidic Acrisols (WRB, 1998), which are generally deep (average soil depth of 1.2 m). The A-horizon is a brown sandy loam (7.5YR4/4, from 55 to 68% sand) and has a low clay percentage (17–19%) and a high content of fine sand (45–50%). This top-soil layer is 0.35-m thick, acidic (pH 4.9–5.2) and has a fine granular structure and is characterized by a low cation exchange capacity (2–4 cmol_c kg⁻¹) and SOC content (from 9 to 12 g C kg⁻¹). A sub-surface organo-mineral AB_w horizon (0.35–0.85 m) has similar texture but is lower in carbon (C) and nitrogen (N) and has a lower cation exchange capacity and base saturation. Beneath these horizons lie two clayey mineral sub-surface (B_w) horizons, which are reddish in color (5YR4/6), have an apedal structure, and are significantly clay-enriched (211–224 g kg⁻¹ for B_{w1} and 297–326 g kg⁻¹ for B_{w2}).

In the lowland areas the soils have been cultivated for several decades with the same farming system, marked by tillage and post-harvest residue grazing by cattle. No-tillage was introduced from 2010 as an experimental trial following its recent progression in Africa in both large-scale and small-scale farms (Huggins and Reganold, 2008). The fundamentals of a no-Till system are that a narrow furrow is made through the organic layer into the soil, the seed and fertilizer are placed into the furrow and then refilled, thereby reducing soil disturbance to a minimum.

2.2. The experimental design

A 450 m² (15 m × 30 m) area was divided into three till (T) and three no-till (NT) 5 m × 15 m plots. An intensive random soil sampling of the top 0.05 m was performed prior to the

implementation of the trial. It showed no significant difference among the six plots for soil texture, pH, SOC content (SOC_C), soil organic N content (SON_C) and stocks (SOC_S, SON_S), soil bulk density (ρ_b), Ca, P and K (data not presented here).

On three plots, tillage was performed manually to a 0.2 m depth, using a hand hoe, as commonly practiced in the area. In preparation for planting for the present study, the NT plots were weeded on the 20th September 2012 with glyphosate herbicide, at a rate of 4 L ha⁻¹. The T plots were conventionally tilled as per normal practice in the area, on the October 9, 2012. Planting was carried out on all experimental plots on the October 9, 2012. At planting, 40 kg N ha⁻¹ was applied on all treatments and 20 kg N ha⁻¹ were top-dressed at six weeks after planting. A cocktail of 2–4D, collisto, metagen and atrazine herbicides was applied twice to control weeds during the growing season, at a rate of 4 L ha⁻¹ and using backpack-mounted sprayers. The maize was harvested by hand on June 20, 2013.

2.3. Evaluation of soil bulk density and soil organic carbon stocks

Within each plot, undisturbed soils samples were collected on June 11, 2013 in the 0–0.05 m soil layer (N=36) using a 0.109 m diameter core. The samples were placed immediately after sampling in air-tight plastic bags with moistened paper towels (to maintain humidity, being careful not to let the towel touch the sample-coring) and placed in a cooler box at ~+4 °C and then transported to the laboratory to perform the incubation.

The determination of soil bulk density (ρ_b) was performed on a separate pseudo-replicate by oven drying at 105 °C for 24 h. This pseudo-replicate was air-dried to determine SOC_C and SON_C. The soil aggregates were crushed and sieved before total soil carbon and nitrogen evaluation through temperature combustion at 1500 °C, using a LECO CNS-2000 Dumas dry matter combustion analyzer (Leco Corp., St. Joseph, MI, USA). SOC stocks (SOC_S) in a given layer were then determined by the product of ρ_b , SOC_C and the thickness of the soil layer, and were expressed in kg C m⁻² as follows:

$$\text{SOC}_S = x_1 x_2 x_3 \left(1 - \frac{x_4}{100}\right) \times b \quad (1)$$

where SOC_S is the SOC stock (kg C m⁻²) of the 0–0.05 m soil layer; x_1 is the SOC concentration in the ≤ 2 mm soil material (g C kg⁻¹); x_2 is the soil bulk density (Mg m⁻³); x_3 is the thickness of the soil layer (m); and b is a constant equal to 0.001.

Because changes in bulk density between the T and NT treatments can result in errors in the estimation of SOC_S, namely overestimations in soils of higher bulk density (Balesdent et al., 1990), comparisons were made based on equivalent soil mass and not on volumes.

2.4. The pre-incubation treatments

Three pre-incubation treatments were carried out to assess the impact of (1) soil crusting, (2) soil compaction and (3) aggregate size fraction.

The crusts of three T and three NT cores were removed using a scalpel. Other core replicates were compacted by 5 and 10% (N=3 for T; N=3 for NT). The method used for compaction consisted in the gentle pushing of a metallic plain tube of the same diameter of the soil core into the undisturbed soil and to a depth corresponding to the required compaction percentage, i.e., from 2.5 mm for the compaction rate of 5% to 5 mm for 10%.

Finally, field moist soil was sieved through either a 2 mm or 8 mm sieve. Soil that passed through 8 mm sieve was further sieved through 2 mm, with material less than 2 mm being discarded. For both T and NT treatments, ring replicates were filled with soil and

aggregates of 0–2 mm diameter and three others with 2–8 mm aggregates. Due to the lack of soil material, two treatments did not have triplicates: NT-2 mm (N = 1) and NT-8 mm (N = 2). Overall, this led to a total of 33 cylinders for incubation.

2.5. Soil incubations

The soil samples (N=33) from both T and NT soils were incubated in 1000 ml open-top jars for 20 days at 20 ± 2 °C and in a 100% humidity environment, following Fang and Moncrieff (2001). Carbon dioxide fluxes were assessed on a daily basis, using a LI-6400XT portable photosynthesis analyzer (Li-cor Inc., Lincoln, NE) using the 6400-09 soil chamber based on non-dispersive infrared (NDIR) gas analyzer over a 2 min time interval. Measurements were performed twice a day during the first three days and once a day thereafter.

Samples were checked for water content by weighing the jars every day after each gas sampling and samples had, when required, water added to maintain a constant moisture content (Wienhold, 2007).

2.6. Soil aggregate stability

The stability of soil aggregates was evaluated on 2–5 mm aggregates to follow the Le Bissonnais (1996) laboratory procedure (ISO 10,930:2012). Aggregates 2–5 mm in size were obtained by hand breaking and dry sieving of bulk sampling collected nearby the soil coring sites. Aggregates were then oven-dried at 40 °C for at least 24 h. The fast wetting, slow wetting and stirring tests were applied, each corresponding to a specific disaggregation process (respectively slaking, differential clay swelling and mechanical breakdown). The fast wetting test consisted in the immersion of 5–10 g aggregates in 50 mL distilled water for 10 min. In the slow wetting test, aggregates were positioned on top of humid foam for 1 h. For the stirring test, the aggregates were first immersed in ethanol, then in water and shaken upside-down 10 times. Dry weights of aggregates collected on each sieve (sizes: 2 mm, 1 mm, 0.5 mm, 0.2 mm, 0.1 mm and 0.05 mm) were subsequently measured. The weight of aggregates <0.05 mm was calculated as the difference between the initial weight and the total weight retained in the sieves. Each fraction was expressed as the percentage of the initial sample dry mass to compute the mean weighted diameter (MWD) calculated as follows:

$$\text{MWD} = \frac{\sum(x_i \times w_i)}{100} \quad (2)$$

with x_i , the mean intersieve size and w_i the percentage of fragments retained the sieve i . The greater the MDW, the more is the aggregates resistant to disaggregation. The MWD was first calculated separately for each test. The mean of these three MWD was calculated.

2.7. Soil microbial biomass

To sample for microbial biomass and activity, the sampling procedure highlighted in Section 2.2 was followed. Soil microbial biomass C was assessed for all T and NT soil samples, using the chloroform fumigation–extraction method by Schinner et al. (1996). The method was modified slightly, where soil suspensions (in 0.5 M potassium sulphate as the extraction solution) were centrifuged at 10,000 rpm for 2 min before filtration and analysis. Organic C was determined immediately after filtration using a Shimadzu TOC-5000 analyzer with an ASI-5000 autosampler and Balston 78-30 high purity TOC gas generator (Shimadzu, Tokyo, Japan) and using standard solutions of 0, 10, 50 and 100 ppm C

made up using 0, 1, 5 and 10 mL of TOC stock solution prepared by dissolving 2.125 g of oven dry reagent grade potassium hydrogen phthalate ($C_8H_5KO_4$) in 1000 mL of distilled water.

In addition, soil microbial activity of the soil samples was estimated by assessing the hydrolysis of fluorescein diacetate (FDA) by soil enzymes, following Alef (1995).

2.8. Quantity and quality of soil dissolved organic C

Bulk top-soil samples from the T and NT plots were sifted through a 2-mm sieve and visible roots were removed. The amount of extractible dissolved organic C (DOC) in soil samples was considered to access the quantity and proportion of the most easily decomposable soil organic matter (Gregorich et al., 1998).

Production of DOC solutions was done by adding cold deionised water to soil (solid/solution ratio=0.1; Kalbitz et al., 2003). Samples suspensions were thoroughly mixed for 2 h, centrifuged (8000 rpm for 30 min at 10 °C) and filtered through 0.45 μ m filter (Millipore). After filtering, an aliquot of each sample was allowed to reach room temperature and analyzed for UV-vis absorbance. A single quartz cuvette (0.01 m, rinsed three times) was used to measure the absorbance spectra via UV-vis spectrophotometer with Milli-Q water as reference.

The remaining filtered samples were stored in the dark (+4 °C) for DOM analysis. Dissolved organic C concentration was measured on acidified samples (pH 2 with 2 N HCL) with Shimadzu TOC-V_{CSH} analyzer via a non-purgeable organic C method. All DOC data are the mean of 3 replicate injections for which the coefficient of variance was always less than 1%.

In addition, specific UV-absorbance ($SUVA_{254}$) values were used to estimate DOC aromaticity by dividing UV absorbance at a wavelength of 254 nm (A_{254}) by DOC concentration (Weishaar et al., 2003). The spectral slope parameter for the 275–295 nm ($S_{275-295}$) and 350–400 nm ($S_{350-400}$) wavelength ranges, which are linked to DOM aromatic (Helms et al., 2008; Spencer et al., 2009) were calculated by applying a non-linear exponential function regression to the absorbance spectrum over these ranges (Stedmon et al., 2000; Helms et al., 2008; Yamashita et al., 2013).

2.9. Statistical analysis

Basic descriptive statistics were reported for soil C-CO₂ emissions and soil characteristics of T and NT soils (Tables 1–3). Comparisons of mean values were carried out using *t* tests. A significance threshold of 5% was used throughout the study. Exploratory multivariate analyses were applied to the data through multivariate correlations (Table 5). Comparisons of C-CO₂ emissions were based on a regression analysis. First, each set of data was fitted separately. Then a regression was carried out using all

Table 2

Summary statistics for spectral slope ratio (S_r), average absorbance between 275 and 295 nm⁻¹ ($S_{275-295}$), specific UV-absorbance ($SUVA_{254}$) and concentration in water soluble DOC (WSDOC). All T and NT samples were considered, N = 18.

	WSDOC (mg CL ⁻¹)		S_r		$S_{275-295}$ (nm ⁻¹ (x10 ⁻³))		$SUVA_{254}$ (LmgC ⁻¹ m ⁻¹)	
	T	NT	T	NT	T	NT	T	NT
Mean	95	112	0.69	0.60	0.0158	0.0154	1.36	1.48
SD	12	20	0.20	0.13	0.0019	0.0011	0.31	0.24
Var.	142	405	0.04	0.02	0.0000	0.0000	0.10	0.06
Min	79	81	0.33	0.39	0.0133	0.0135	0.96	1.00
Q1	89	94	0.62	0.51	0.0151	0.0146	1.14	1.36
Median	93	112	0.74	0.64	0.0156	0.0152	1.38	1.53
Q3	98	126	0.82	0.69	0.0158	0.0163	1.61	1.56
Max	115	144	0.90	0.80	0.0192	0.0168	1.72	2.01
Skew	0.71	0.18	-1.24	-0.03	0.9045	-0.2193	-0.14	0.07
Kurt	1.41	-1.05	1.45	-1.15	2.1663	-1.4241	-2.38	1.44
SE	21	24	0.15	0.13	0.0004	0.0002	0.30	0.32
CV	13	18	30	22	12.2	7.4	23	16

Table 3

Summary statistics for CO₂ emission for undisturbed tilled (T) and no-tilled (NT) soils. N=6.

	CO ₂ -C					
	mgC-CO ₂ g ⁻¹ day ⁻¹		mgC-CO ₂ g C ⁻¹ day ⁻¹		gC m ⁻² day ⁻¹	
	T	NT	T	NT	T	NT
Mean	0.24	0.21	1.10	0.90	1.05	1.15
SD	0.15	0.03	0.19	0.10	0.19	0.15
Var.	0.02	0.00	0.00	0.00	0.04	0.02
Min	0.16	0.16	0.94	0.75	0.88	0.94
Q1	0.19	0.19	0.95	0.80	0.92	1.03
Median	0.21	0.21	1.03	0.94	1.03	1.13
Q3	0.33	0.23	1.24	0.97	1.08	1.26
Max	0.44	0.25	1.40	1.07	1.41	1.42
Skew	1.11	-0.18	0.84	4.42	1.44	0.44
Kurt	0.10	-0.93	-0.78	-1.46	2.61	-0.78
SE	0.02	0.05	0.24	0.25	0.04	0.03
CV	7.0	12.4	17.2	12.8	18.5	12.8

the data together. The null hypothesis was that the regression that used all the data performed as well as the ones taking the datasets separately. If the null hypothesis of a single regression was rejected, we concluded that the sets of data were different.

3. Results

3.1. Tillage impact on selected soil properties

3.1.1. Selected physical parameters

The general statistics of bulk density (ρ_b), SOC_C, SON_C, C:N ratio, SOC_S and mean weight diameter (MWD) for tilled (T) and no-tilled

Table 1

Summary statistics for bulk density, carbon content, nitrogen content, soil carbon stocks and mean weight diameter (MWD) from 0 to 0.05 m soil layer.

	ρ_b (g cm ⁻³)		SOC _C (g kg ⁻¹)		SON _C		C/N		SOC _S (kg m ²)		MWD (mm)	
	T	NT	T	NT	T	NT	T	NT	T	NT	T	NT
Mean	1.033	1.241	12.41	19.10	1.47	1.85	8.4	10.2	0.86	1.28	1.67	2.27
SD	0.231	0.104	1.04	2.06	0.15	0.16	0.97	0.29	0.17	0.17	0.94	0.74
Var.	0.053	0.011	1.07	4.26	0.02	0.03	0.95	0.08	0.03	0.03	0.88	0.54
Min	0.654	1.052	10.52	16.70	1.30	1.60	11.35	12.24	0.58	1.07	0.64	1.58
Q1	0.787	1.170	11.70	17.80	1.40	1.75	12.57	12.44	0.75	1.17	1.14	1.69
Median	1.122	1.197	11.97	18.60	1.40	1.80	12.93	12.58	0.91	1.23	1.28	1.84
Q3	1.203	1.332	13.32	19.65	1.55	1.90	13.40	12.79	0.96	1.30	2.58	3.16
Max	1.311	1.404	14.04	25.60	1.70	2.20	16.00	13.12	1.11	1.78	3.12	3.30
Skew	-0.828	0.087	0.09	1.77	0.44	0.66	1.15	0.41	-0.50	1.94	0.63	0.71
Kurt	-1.003	-1.022	-1.02	3.97	-1.10	0.35	3.74	-0.74	-0.90	4.73	-1.34	-1.72
SE	0.050	0.023	0.23	0.45	0.03	0.03	0.21	0.06	0.04	0.04	0.31	0.25
CV	0.224	0.0836	0.08	0.11	0.10	0.09	0.07	0.02	0.20	0.13	56.21	32.54

Table 5

Coefficients of determination (r) between soil CO₂ emission per g of carbon in the soil and per m² and multiple environmental factors.

	CO ₂ emissions	
	gC-CO ₂ m ⁻²	gC-CO ₂ gC sol ⁻¹
SOC _C	0.59	-0.26
SOC _C + ρ _b	0.60	-0.90
SOC _C + ρ _b + MWD	0.63	-0.91
SOC _C + ρ _b + MA	0.62	-0.91
SOC _C + ρ _b + MB	0.63	-0.91
SOC _C + ρ _b + S _R	0.60	-0.90
SOC _C + ρ _b + S ₂₇₅₋₂₉₅	0.58	-0.90
SOC _C + ρ _b + SUVA ₂₅₄	0.58	-0.90

(NT) soils are shown in Table 1. NT soils were 18% denser than the T ones (ρ_b = 1.03 vs 1.24 g cm⁻³). The SOC_C increased by 54% from 12.4 g kg⁻¹ under T to 19.1 g kg⁻¹ under NT, and the SOC_S were 32% higher under NT, all these differences being significant. SON_C was 26% greater under NT compared to T (1.85 vs 1.47 g kg⁻¹) and the resultant C:N ratio was the greatest under NT (10.2 vs 8.4). Finally, NT soil aggregates were significantly more stable than T ones as expressed by a MWD of 2.27 mm for NT compared to 1.67 mm for T (Table 1), with most of the differences between the two tillage treatments occurring for fast wetting (MWD of 0.8 for T and 1.7 for NT), followed by mechanical breakdown (1.33 mm for T and 1.7 for NT), and slow wetting (2.86 and 3.23 mm) (Fig. 1).

3.1.2. Dissolved organic C and microbial biomass C

The content of water extractable DOC increased significantly with the cessation of tillage from 95 ± 21 to 112 ± 24 mg CL⁻¹, which corresponded to a 18% difference (Table 2). Water extractable DOC was biochemically less stable under NT compared to T as expressed by a lower spectral slope ratio (S_R) (0.69 ± 0.15 vs 0.60 ± 0.13), but higher SUVA₂₅₄ values (1.48 ± 0.32 vs 1.36 ± 0.30 mg C⁻¹ m⁻¹). However, tillage impact on S₂₇₅₋₂₉₅ was not statistically significant.

Finally, both microbial biomass C and microbial activity significantly decreased with the abandonment of tillage. As shown by Fig. 2, microbial biomass C decreased significantly from 522.0 ± 31.3 mg C kg⁻¹ under T to 213.1 ± 10.9 mg C kg⁻¹ under NT, which corresponded to a 59% difference. Concomitantly, the cessation of tillage induced a significant and sharp fall in microbial activity with FDA hydrolysis values decreasing from 12.8 to 8.0 μg g⁻¹ h⁻¹ (Fig. 3), a 37% difference.

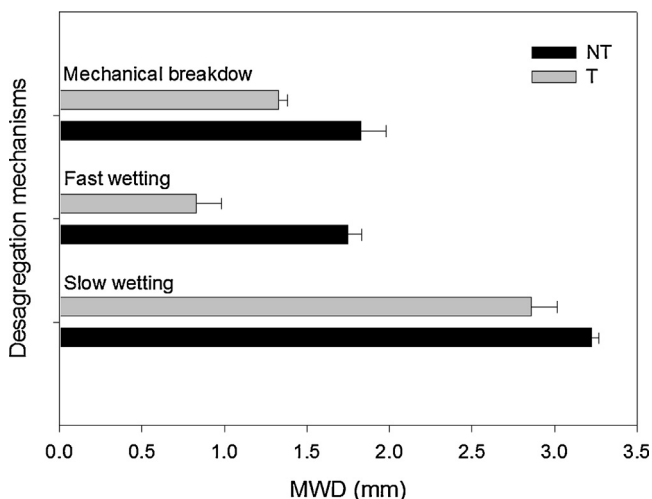


Fig. 1. Soil tillage impact on soil mean weight diameter (MWD) as a proxy of soil disaggregation potential by the three involved mechanisms. N = 9.

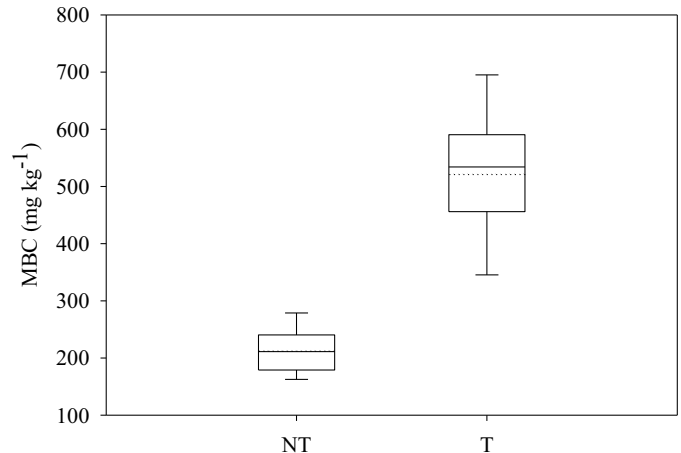


Fig. 2. Tillage impact on soil microbial biomass carbon (MBC). Plain lines correspond to 10th, 25th, median, 75th and 90th percentiles; dotted lines to the mean. N = 3.

3.2. Impact of tillage on CO₂ emissions

Table 3 shows the summary statistics of CO₂-C emission for both undisturbed T and NT soils. The average daily CO₂-C emission per g of C in the soil was significantly greater by 22% on T compared to NT (1.10 vs 0.90 mg C-CO₂ g C⁻¹ day⁻¹). After 20 days, T soil had emitted 0.0169 ± 0.00076 g CO₂-C g⁻¹ C, while the NT soils had released 0.0134 ± 0.00091 g CO₂-C g⁻¹ C (Fig. 4). However, expressed per area basis, the total CO₂-C emission appeared significantly greater from NT soils (1.15 ± 0.03 g CO₂-C m⁻² day⁻¹) compared to T soils (1.05 ± 0.04 g CO₂-C m⁻² day⁻¹), a 9% difference.

3.3. Impact of selected soil properties on CO₂ emissions

3.3.1. Impact of soil compaction

Results on the impact of soil compaction on soil C-CO₂ emissions are presented in Figs. 5 and 6A. Soil compaction decreased significantly soil CO₂ emissions from 0.85 mg CO₂-C g⁻¹ C day⁻¹ for the undisturbed T bulk soil to 0.79 mg CO₂-C g⁻¹ C day⁻¹ for the 5% compaction (T-c5) and to 0.72 mg CO₂-C g⁻¹ C day⁻¹ for the 10% treatment (T-c10), which in the latter case corresponded to an 18% difference.

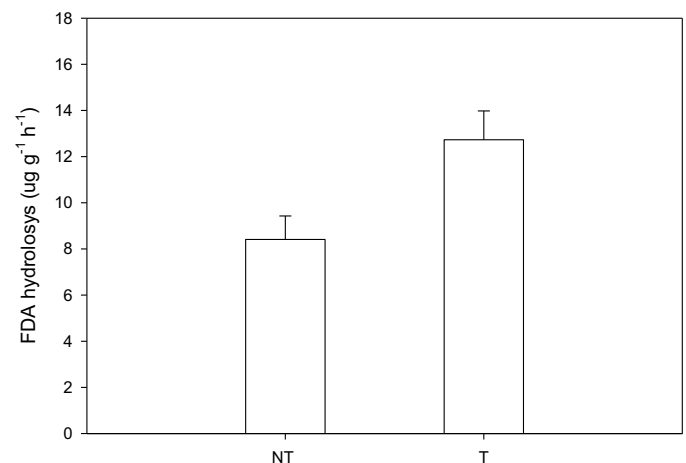


Fig. 3. Tillage impact on top-soil (0–0.05 m) microbial activity (MA) estimated by assessing the hydrolysis of fluorescein diacetate (FDA) by soil enzymes. N = 3.

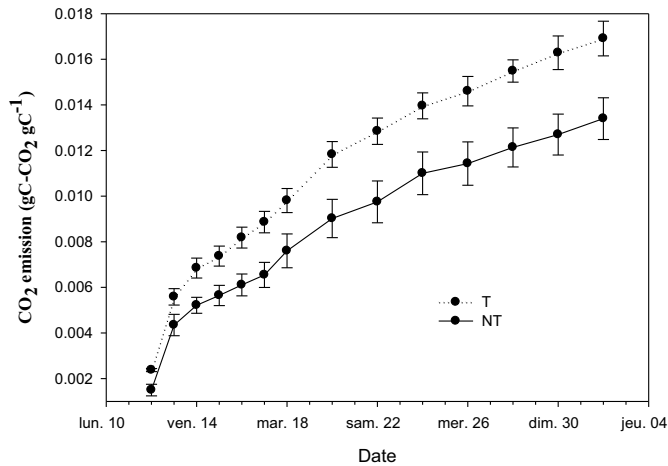


Fig. 4. CO₂ emissions from till (T) and no-till (NT) soils. N=3. Regression analysis showed that the datasets were significantly different.

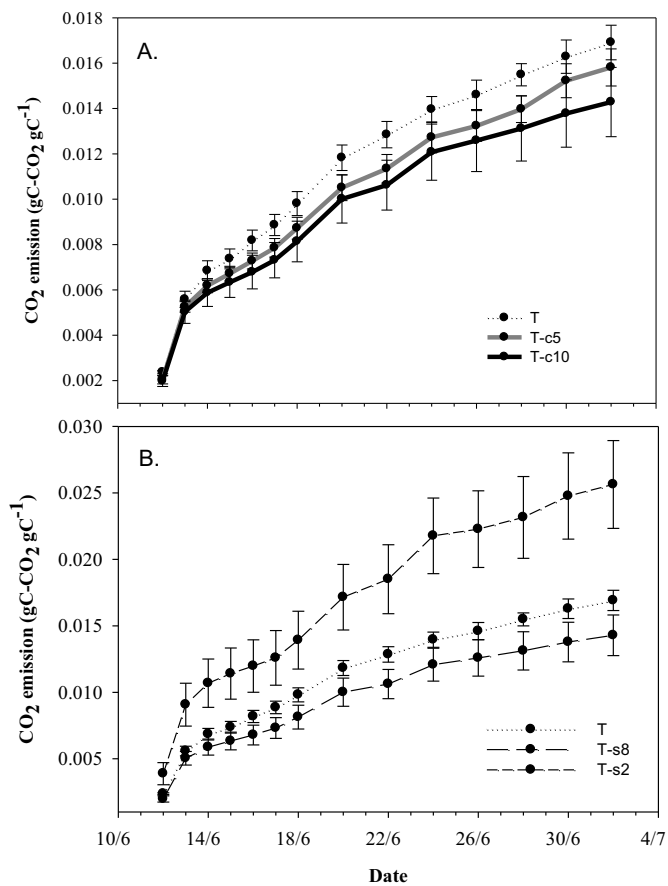


Fig. 5. CO₂ emissions from T soils (A) compacted by 0, 5 and 10% (T-c5 and T-c10, respectively), and (B) from soil aggregates sieved at 2 and 8 mm (T-s2 and T-s8, respectively). N=3. Regression analysis showed that the datasets shown on the same graph were significantly different.

In contrast, compaction of the NT soils resulted in an opposite effect on CO₂ emissions: the compaction decreased significantly the emission level.

3.3.2. Impact of the size of soil aggregates

Aggregates with a diameter from 2 to 8 mm represented 80% of the bulk soil mass under NT and 55% under T, the remaining

20 and 45% corresponding to aggregates with a diameter lower than 2 mm.

Aggregates with a 0–2 mm diameter for T soil (T-s2) were characterized by the highest CO₂ emissions (cumulative 20 days of 0.0256 g CO₂-C g⁻¹C), 51% significantly greater than soil aggregates with a 2–8 mm diameter (T-s8, 0.0143 g C-CO₂ g C⁻¹), while the bulk T soil showed intermediate gaseous emissions (0.0169 g CO₂-C g⁻¹C) (Fig. 5B). All the NT soil aggregates sizes under study also showed higher emissions than the bulk soil, (0.0066 g CO₂-C g⁻¹C for NT-s8 vs 0.0116 g CO₂-C g⁻¹C for NT-s2; Fig. 6B). Interestingly, the greatest CO₂ emissions from NT aggregates occurred for NT-s8 during the first day of the incubation, i.e., immediately after aggregate sieving.

3.3.3. Impact of soil crusting

Both tillage treatments exhibited significantly greater CO₂ emissions when a soil surface crust was present on the soil surface than when the crust was removed (Fig. 7A–B; Table 4). After 20 days of incubation, the cumulative CO₂ emissions from NT soils were 0.0134 g CO₂-C g⁻¹C with surface crust compared to 0.0119 g CO₂-C g⁻¹C when surface crust was removed, a 11% difference. Similarly, crust removal decreased significantly CO₂ emissions from T soils from 0.0169 g CO₂-C g⁻¹C to 0.0154 g CO₂-C g⁻¹C.

3.3.4. The main controls of CO₂ emissions

Soil CO₂ emissions per unit of surface appeared to significantly increase with the increase in SOC_C ($r=0.59$), followed by SON_C ($r=0.52$), SOC_S ($r=0.48$) but decreased with increasing MA ($r=-0.37$) and MB ($r=-0.34$), which corresponded in all cases to significant correlations (Table 4). In the mean time, emissions expressed in gC-CO₂ gC sol⁻¹ were shown to decrease the most with increasing ρ_b ($r=-0.89$) and SOC_S ($r=-0.81$), far in front of MWD ($r=0.50$), but decreased with increasing MB and MA (Table 4).

Table 5 presents the coefficients of determination (r) between CO₂ emissions per m² and per g of C in the soil, and multiple environmental factors. Variations in CO₂ emission per m² appeared to be explained the most by SOC_C, ρ_b and MWD with 63% of the data variance explained by these variables. In contrast, SOC_C and ρ_b explained 91% of the variance of CO₂ emissions per g of OC in the soil (Table 5).

4. Discussion

4.1. Bulk soil C-CO₂ emissions

There is a general consensus that the suppression of tillage decreases soil CO₂ emissions per unit of surface since, as suggested by Six et al. (2000), the aggregates' disruption due to tillage renders the initially protected organic matter (OM) accessible to decomposers.

This was supported by various studies worldwide and pointed to reductions of gaseous emissions per m² as high as 73% by La Scala et al. (2006) in Brazil, 66% in the study of Carbonell-Bojollo et al. (2011) in Spain, 35% by Jabro et al. (2008) in USA. However, several studies pointed to much lower reduction rates, as for instance 2% by Brye et al. (2006) and Sainju et al. (2010) in the USA. In contrast, greater CO₂ emissions from NT soils were reported. Barreto et al. (2009) showed for instance a 700% increase in C-CO₂ emissions by using NT. This rate was as much as 29% in a study by Aslam et al. (2000) in New Zealand, 15% in Brazil (Carmo et al., 2007), 9% by Smith et al. (2011) in the USA, 5% by Sainju et al. (2010) in the USA and 4% by Lee et al. (2009) in the USA. This is in line with the 17% observed in the present study.

The greater CO₂ emissions per m² from NT soils are often attributed to the presence of fresh organic material or mulch on the

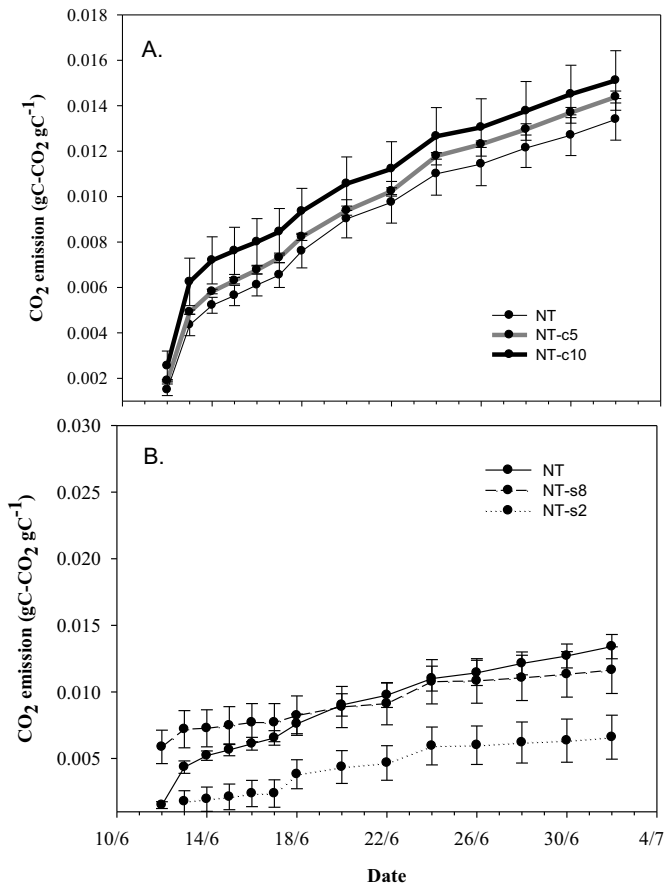


Fig. 6. CO₂ emissions from NT soils, (A) compacted by 5 and 10% (T-c5 and T-c10, respectively), and (B) from NT soil aggregates sieved at 2 and 8 mm (T-S2 and T-S8, respectively). N = 3 except from NT-s2 (N = 1) and NT-s8 (N = 2). Regression analysis showed that the datasets shown on the same graph were significantly different.

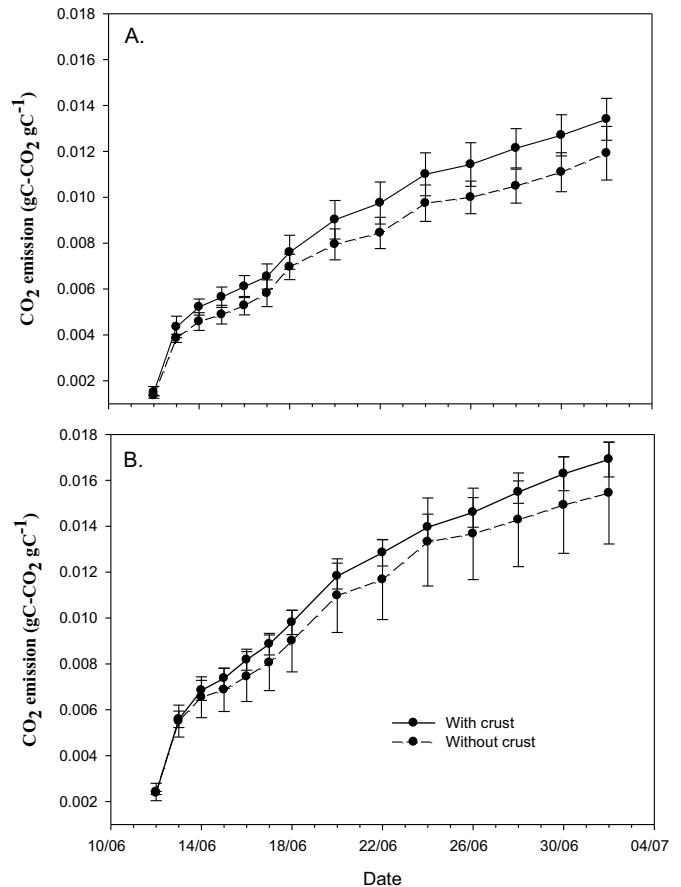


Fig. 7. Impact of soil surface crusting on soil CO₂ emissions for both NT (A) and T (B) soils. N = 3. Regression analysis showed that the datasets shown on the same graph were significantly different.

soil surface, which are easily accessible to decomposers (Oorts et al., 2007). Since, in the present study no mulch was present on the soil surface, what can explain the greater C–CO₂ emissions from NT soils?

Among the possible explanations of the greater soil C–CO₂ emissions from NT soils is their greater levels of microbial biomass and biomass activity as cited by Helgason et al. (2009), Spedding et al. (2004), Drigber et al. (2000), Shi et al. (2012), Zhang et al. (2005, 2012), Nakamoto et al. (2006) and Follett and Schimel (1989).

However, the present study pointed to a significant decrease in top-soil microbial biomass and biomass activity associated with the abandonment of tillage, a result similar to the one of Govaerts et al. (2007). Another possible explanation of the lower microbial activity under NT soils is the lower soil porosity and associated reduced oxygen inputs from the atmosphere, which limits the development and activity of living organisms (Pandey et al., 2014; Nsabimana et al., 2004). The presence of more recalcitrant OM, as found in this study could be another cause of lower biological activity and soil C–CO₂ emissions.

Here we show that the most significant control of the enhanced gaseous emissions per m² from NT soils lies into a the top-soil enrichment in SOC. Indeed, several authors reported that adopting NT changes the distribution of C in the soil profile with a rise in SOC stock in the top-soil and a decline in depth (Baker et al., 2007; Luo et al., 2010; Hou et al., 2012), which might significantly affect organic matter decomposition and gazes exchanges from soils the atmosphere. In particular, the increased proximity of OM to

decomposers and to the atmosphere, together with higher soil temperature (Hou et al., 2014), are likely to yield higher OM decomposition and CO₂ release to the atmosphere. In addition, the present study points to the presence under NT of a biological surface crust which appeared to play an important role in the release of greenhouse gases as lower emissions characterized the decrusted soils. While Mchunu et al. (2011) pointed to the presence of biological features on soil crusts such as algae, which might favor CO₂ emissions, more is to be done on soil crusting impact on soil C exports. It is not clear whether the increase in CO₂ emissions due to crusting is caused by aggregate disruption and soil compaction by raindrops or by the biological colonization of crusts, each producing potentially opposite effects.

4.2. Organic matter protection

Despite exhibiting greater C–CO₂ emissions per m² to the atmosphere, the NT system in this study yielded lower C exports per g of soil carbon and thus greater OM protection from decomposers. This was likely to come from the lower biota activity, a direct consequence of the higher soil aggregate stability and soil bulk density, both which protect OM from decomposers and gaseous exchanges to the atmosphere. While the present study pointed to the primary impact of soil compaction on lowering soil CO₂ emissions, C–CO₂ emissions surprisingly increased at the 10% compaction level of the NT treatment. This result could be explained by the breaking-down of soil aggregates and the associated release and decomposition of the encapsulated OM.

Table 4

Coefficients of determination (r) between soil CO₂ emission per g of carbon in the soil and per m² and multiple environmental factors: top-soil organic carbon and organic nitrogen content (SOC_C; SON_C), SOC stocks (SOC_S), soil bulk density (ρ_b); mean weighted diameter (MWD); spectral slope ratio (S_R) of $S_{275-295}$ and $S_{350-400}$; specific UV-absorbance (SUVA₂₅₄); water soluble dissolved organic carbon (WSDOC), microbial biomass carbon (MB); microbial activity (MA).

	SOC _C	SON _C	SOC _S	ρ_b	MWD	S_R	$S_{275-295}$	SUVA ₂₅₄	WSDOC	MB	MA
gC-CO ₂ m ⁻²	0,59 ^a	0,52 ^a	0,48 ^a	0,20	0,16	-0,17	-0,14	0,05	0,30	-0,34 ^a	-0,37 ^a
gC-CO ₂ gC sol ⁻¹	-0,26	-0,39 ^a	-0,81 ^a	-0,89 ^a	-0,50 ^a	-0,06	0,14	0,16	-0,47 ^a	0,59 ^a	0,57 ^a

^a Statistically significant determinants at 95% confidence level.

5. Conclusion

In this study of tillage impact on carbon outputs from soils, our main objective was to assess the changes in soil C-CO₂ emissions immediately following the suppression of tillage and to investigate some of the underlying physical, chemical and biological factors of control. Two main conclusions can be drawn from laboratory measurements of undisturbed soil samples and microcosms. The first one is that tillage abandonment significantly decreased soil CO₂ emissions per unit of soil carbon by an average of 22%. Such an improved OM protection correlated with the increase in aggregate stability and soil bulk density, and the decrease in both microbial biomass and microbial activity. The second conclusion was that despite tillage cessation promoted greater OM protection, the increase of SOC stocks in the top 0.05 m of the soil by about 50%, resulted in increased total soil C-CO₂ emissions by an average of 9.5%.

These results are expected to improve our understanding of the role of agricultural practices on the global carbon cycle. Furthermore, they are expected to inform on best management practices for improved C sequestration in soils. Under the conditions of the study (i.e., short-term cessation of tillage, crop residue removal, acidic and sandy conditions) a greater carbon sequestration could be probably achieved by increasing soil bulk density and allocating more organic carbon deep in the soil profile, rather in the top-soil. This might be achieved by planting for instance crops with high carbon allocation to deep roots and by providing a favorable environment for the soil engineers such as epigenic earthworms, to move OM deep in soil profiles. Strip-till might also be beneficial.

This study focused on short-term (2 years) tillage cessation and soils with no residues as a means to assess the soil processes originating the changes in CO₂ emissions. The origin of the differences in CO₂ emissions between tilled and no-tilled soils need to be investigated longer term and for different no-tilled systems such as these involving residue mulching, crop rotations and crop associations among others. Different climates and environmental conditions need to be investigated before incorporation could be made in carbon models. More research is also required to better understand the reasons and controlling factors of the observed physical, biological and chemical changes following shifts in land use or land management, as well as to adapt agricultural practices to local conditions for effective carbon sequestration.

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