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Nitrous oxide emission after the addition of organic residues on soil surface

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A R T I C L E I N F O

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A B S T R A C T

The addition of organic amendments with a low Carbon:Nitrogen (C:N) ratio to restore degraded soils may increase the emission of greenhouse gases and affect the storage of these elements in the topsoil. Our objectives were to evaluate the effect of the addition to the surface of organic amendments to a non-tilled and non-cultivated soil, compared with the addition of cover crop residues, with respect to: (i) N_2O emissions and their relationships with soil variables, (ii) C and N content in the topsoil. We conducted an experiment during two consecutive years in Paraná, Argentina (−31° 50.9′ S; −60°32.3′ W). Treatments included the addition of organic amendments (composted poultry litter and poultry manure) and cover crops residues [wheat (*Triticum aestivum* L.) and white sweet clover (*Melilotus albus* Medik.)]. Soil variables that are most commonly reported as critical for N_2O emissions, i.e. soil nitrates ($N\text{-}NO_3$) and water filled pore space (WFPS), showed more differences among treatments in Year 1 than in Year 2, which was associated with a higher frequency and amount of rainfall. N_2O flux ranged between 0.12-50 $\mu g\ N\ m^{-2} d^{-1}$ (Year 1) and 0.62-13.7 $\mu g\ N\ m^{-2} d^{-1}$ (Year 2). N_2O flux was significantly associated with WFPS in both Years ($P < 0.004$ in Year 1 and $P < 0.002$ in Year 2) and with $N\text{-}NO_3$ ($P < 0.045$) in Year 2. Although the N_2O emissions were extremely low, the highest values were recorded in poultry manure treatment, whereas, the lower values were recorded in crop residues and in the control treatment. Overall, our results suggest that the addition of organic amendments in our region, with a massive adoption of no-till, that were broadcasted before (i.e. 30–45 days) the sowing of summer crops, are associated with low N_2O emissions and potential improvements in soil quality. In addition, the experimental approach allowed us to more clearly identify the drivers of N_2O emissions and to better understand the soil processes that are involved in this particular situation, without the presence of a living vegetal cover.

1. Introduction

Increasing global demand for food, fibre and biofuels (Godfray and Garnett, 2014; Godfray et al., 2010; Popp et al., 2014) has led to an important change in the use of agricultural lands, which may affect soil quality, thus reducing their productivity and their ability to provide key ecosystems services (FAO, 2011; Popp et al., 2014). As a consequence, there is a growing need to develop agricultural systems having a more efficient use of resources and a lower impact on soil health and environment. Future agricultural systems should be able to maintain or even increase productivity while protecting biological diversity and reducing greenhouse gas (GHG) emissions (Tilman et al., 2011).

However, that desirable goal is far from being reached in most South American agroecosystems, in which the land use change has been characterised by a trend toward soybean monoculture (Wingeyer et al., 2015), with dramatic consequences in terms of soil degradation. In fact, an important decrease in soil carbon (C) stocks has been documented when the cropping systems largely rely on soybean as the sole crop of the year (Novelli et al., 2017, 2011; Studdert and Echeverria, 2000). Soil degradation in these agroecosystems has been associated with the action of intense erosion processes (Viglizzo et al., 2011; Wingeyer et al., 2015). These erosion processes are related with the low amount of crop residues remaining on the soil surface which are inherent in simple rotations (i.e. soybean monoculture) in comparison with more complex rotations (Novelli et al., 2017).

The most important functions of soils for environmental health are associated with the content of soil organic carbon (SOC) (Lal, 2005, 2004; Six et al., 2004). Degraded soils, which have lost an important amount of SOC, usually have reduced intrinsic capability to provide essential ecosystem services such as nutrient cycling, chemical detoxification (Kirk et al., 2004) and water filtration. Therefore, soil restoration appears as a critical issue in order to achieve more productive and sustainable agroecosystems. The addition of crop residues or organic amendments (Lal, 2010) may have an important impact on

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restoring some of the lost ecosystems services in degraded soils.

Soil restoration using organic amendments has been successful, not only to increase the SOC stock, but it has also favoured the improvement of other physical, chemical and biological properties of soil (e.g. Ros et al., 2003; Tejada et al., 2006) however these studies have been carried out by mixing the amendment with the soil using tillage operations. These practices are not compatible with the wide use of the no-till system in South America, having the main premise of maintaining a crop residue mulch on soil surface (Alvarez et al., 2011). Although the effects of no-till on crop production and soil health have been widely reported (e.g. Díaz-Zorita et al., 2002; Fabrizzi et al., 2005; Fernández Canigia et al., 2000), a considerable time-period is required before these effects become evident (Lal, 2010).

The impact of the addition of organic amendments on surface, with respect to the potential for environmental pollution, has led to growing concerns in areas where residues of intensive poultry farms are frequently added to soil. Although the composting of poultry litter and manure may reduce their pollutant load (Tyson and Cabrera, 1993), the impact of these composted amendments on the emission of GHG remains unknown when the residues are broadcasted on the soil surface. Moreover, since the high mineral N availability as N-NO_3, the low tension of atmospheric Oxygen (O_2) and the temperature are the main driver factors of nitrous oxide (N_2O) emission (Beauchamp, 1997; Alvarez et al., 2012; Cosentino et al., 2013), no-till systems may be more prone to high emission than conventional tillage systems, due to higher water infiltration and lower soil evaporation which reduce soil aeration (Fabrizzi et al., 2005; Rochette, 2008).

Reports on the impact of tillage systems on N_2O emissions are, however, contradictory. For instance, Liu et al. (2007) have reported higher fluxes of N_2O in no-till than in conventional tillage, whereas the opposite was reported by Perdomo et al. (2009). Others researchers have found no differences in the flux of N_2O emissions between tillage systems (Metay et al., 2007; van Kessel et al., 2013).

The quantification of the N_2O emissions associated to a soil restoration practice, such as organic amendment addition, is a key issue for the design of sustainable agricultural systems, due to the increasing need to reduce N_2O emissions from agriculture (Henault et al., 2012). Likewise, reduction of N emission from soil may increase the N-use efficiency of the agroecosystems, which is another benchmark to improve the whole system efficiency and sustainability. In our region, with a massive adoption of no-till, organic residues on surface are broadcasted far ahead, usually 30–45 days, before sowing of the next crop. However, most studies on N_2O emissions have been carried out in the presence of a living vegetal cover, which could affect N_2O emissions, due to changes in the soil environment (mainly N and C substrate availability) that are promoted by plant water and N uptake. Thus, an experimental approach without living vegetal cover may allow more clear identification of the drivers of N_2O emissions and to better understand the soil processes that are involved.

Our objectives were to evaluate the impact of the addition of organic amendments and vegetal residues to the soil surface, with no vegetal living cover on it: (i) N_2O emissions and its relationships with soil moisture and inorganic N, (ii) soil C and N content in topsoil after two consecutive years of addition.

2. Materials and methods

2.1. Experiments

We conducted an experiment during two consecutive years, from September 2014 to January 2015 (Year 1, encompassing 157 days) and from October 2015 to January 2016 (Year 2, encompassing 97 days), in a field of the experimental station of INTA Paraná (−31° 50.9′ S; −60° 32.3′ W), Entre Rios province (Argentina). The soil was classified as an Aquic Argiudoll (Plan Mapa de Suelos, 1998; Soil Survey Staff, 2010) under no-till since 1998.

The treatments included the addition of different residues on soil surface: two cover crop residues and two organic amendments. Crops residues were wheat (*Triticum aestivum* L.) and white sweet clover (*Melilotus albus* Medik) from cover crops cultivated elsewhere whereas organic amendments were composted poultry litter and poultry manure. A control treatment without residue addition was also included. The area of each plot was 2 m^2. We used a randomised complete block design with four replicates.

The experiment was kept free of crops and weeds over the two-year period because no crops were planted and weeds were removed using manual and chemical control methods. The location of the plot was the same over the two-year period, i.e. the treatments were consecutively replicated on the same plot.

The organic amendments were previously composted in order to reduce their pollutant load whereas crop residues were cut, oven-dried and stored until addition. Organic amendments were composted until their stabilization, according to the method proposed by Petric and Mustafić (2015), i.e. using an air flow rate of 0.43 l $min\ kg^{-1}$ and mean air temperature of 28 °C for poultry litter and poultry manure. The residues were added on 24 September 2014 (Year 1) and on 1 October 2015 (Year 2).

2.2. Residues characterization

Total nitrogen (N) and total carbon of the residues were determined by dry combustion using a LECO TRU SPEC autoanalyzer (Leco Corp., St. Joseph, MI, USA) (Table 1). The rate of residue addition was 5 t C ha^{-1}, therefore the total amount of dry matter of the added residues ranged from 12 to 12.5 t ha^{-1} for crops residues, from 20.2 to 20.8 t ha^{-1} for poultry manure, whereas the amount of added poultry litter was 10.7 t ha^{-1} in both years.

2.3. Measurements

Periodically, between 09 and 12 a.m. (Cosentino et al., 2012), we measured the emission of N_2O using the static chamber methodology (Conen and Smith, 1998). The chambers were designed according to the minimum established requirements for the protocol as proposed by Parkin et al. (2003). The chambers, with an area of 0.04 m^2, were carefully installed until they reached 0.05 m soil depth after residue addition. Air samples from the chamber were taken at time 0 for the starting values of the atmospheric concentration of N_2O and after 20 and 40 min following chamber closure. Samples were stored in vials until analysis in a cool, dry place.

The concentration of N_2O in the air samples was determined by gas chromatography using a gas chromatograph GC 7890 A with autosampler 7697 A (Agilent Network GC System, AECD, Santa Clara, CA, USA).

Coinciding with each air sampling date, soil samples were collected at 0.05 m depth in order to evaluate soil moisture and N-NO_3 concentration. Three or four samples were taken from the area surrounding the chamber, and mixed to form one composite sample per plot. Soil moisture was determined using the gravimetric method, i.e. weighing the sample immediately after sampling and after 48 h of being oven-

Table 1
Nitrogen concentration (N), carbon concentration (C) and the C:N ratio of the added residues.

	N (%)		C (%)		C:N ratio	
	Year 1	Year 2	Year 1	Year 2	Year 1	Year 2
Poultry manure	2.2	2.2	24.8	24.0	10.8	11.3
Poultry litter	2.5	2.5	47.0	47.0	18.8	18.8
White sweet clover	1.7	3.3	39.2	42.2	22.6	12.8
Wheat	1.0	1.5	42.7	40.2	42.7	27.0

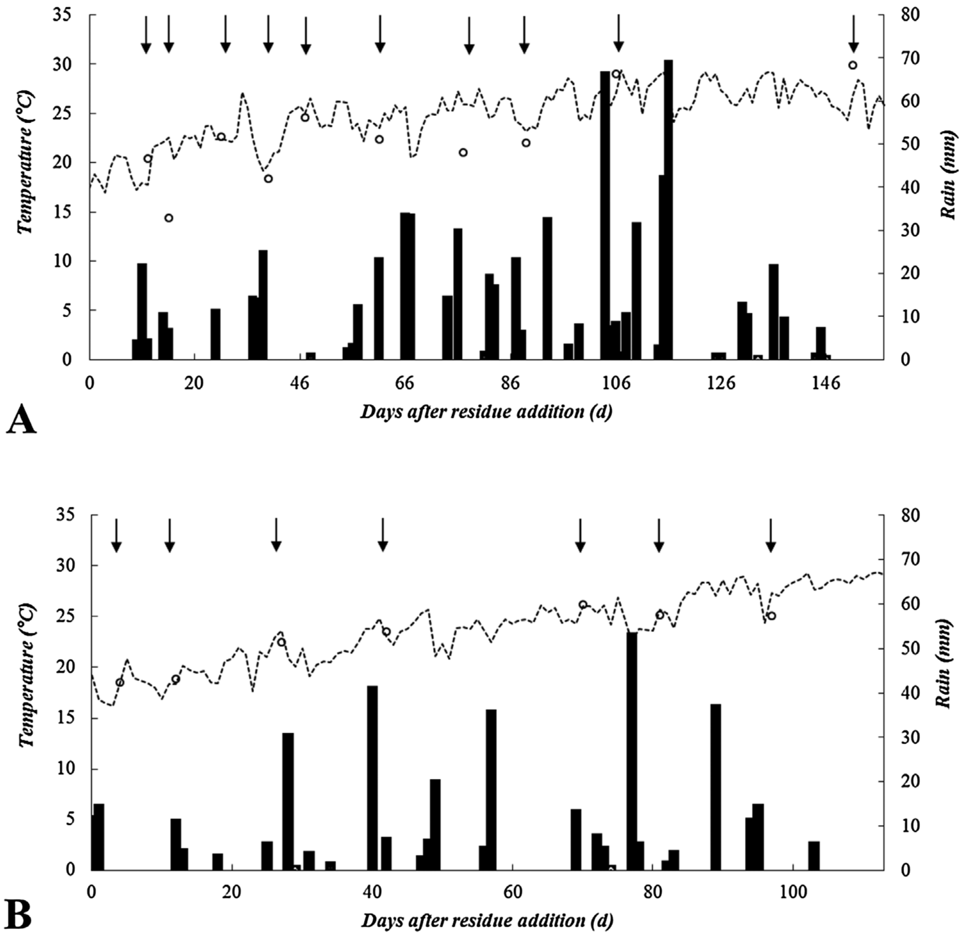

Fig. 1. Mean soil temperature at 0.05 m depth (dotted line), air temperature at each measurement date (open circles) and daily rainfall (black bars) during the experimental period in (a) Year 1 and (b) Year 2. Arrows indicate measurement dates of N_2O, N-NO_3 and WFPS.

dried at 105 °C. An aliquot of the sample was air-dried, ground and sieved through a 2 mm screen to determine N-NO_3 concentration by a colorimetric method (Bremner, 1965).

Soil samples at 0.05 m depth were also taken at the onset of the experiment (initial) and at the end of each evaluation period. The composite samples comprised at least 10 sub-samples that were collected from different locations in each plot. These soil samples were air-dried, ground, and sieved through a 0.5-mm screen. Total C content and total nitrogen were determined by dry combustion using a LECO TRU SPEC autoanalyzer (Leco Corp., St. Joseph, MI, USA). Bulk density in each plot and sampling date was determined by the core method (Blake and Hartge, 1986) using cores of length 0.03 m and diameter 0.054 m.

2.4. Calculations

The N_2O fluxes (*f*) was calculated using the ideal gases equation:

$$f = \left(\frac{\Delta C}{\Delta t}\right) * \frac{V}{A} * \frac{m}{Vm}$$

Where:

c$\Delta C/\Delta t$/ is slope of linear increase in the concentration of N_2O during the closing period of the chamber.

V is volume of chamber (2.01 l).

A is area covering chamber floor (0.04 m^2).

m is molecular mass of N_2O (44 µg μmol^{-1}).

Vm is molar volume of N_2Ocorrected by air temperature.

Emission of gases was calculated using the average temperature between 09 and 12 am. The air temperature was obtained from an agrometeorological station located at 150 m from the experiment.

We calculated soil water-filled pore space (WFPS, %) from the soil moisture and the bulk density.

Weighted means of N-NO_3 and WFPS were calculated, considering the value of the variable in each measurement date and duration of period between two consecutive measurements.

2.5. Statistical analyses

We used analysis of variance (ANOVA) to evaluate the effect of treatments in the studied variables. The software used was INFOSTAT (Di Rienzo et al., 2011). When the ANOVA indicated significant differences ($P < 0.05$), we compared the treatment means using the Duncan test ($\alpha = 0.05$). Linear regressions and correlation analysis were also performed using INFOSTAT.

3. Results

3.1. Climatic conditions

In Year 1, during the measurement period, cumulative rainfall was 665 mm. This value was similar to the historical (1967–2014) average (614 mm) for the same period. There were two rainfall events higher than 50 mm, whereas the other 44 events averaged 12 mm (Fig. 1a). In Year 2, the cumulative rainfall was 347 mm with only one rainfall event of 50 mm, whereas the other 27 events averaged 3 mm (Fig. 1b). The cumulative rainfall represented only 57% of the historical average for the same period. The mean air temperature at measurement time ranged from 14 to 29 °C (Year 1) and from 18 to 26 °C (Year 2). The average mean soil temperature at 0.05 m depth was 24.8 °C in Year 1 and 23.7 °C in Year 2 (Fig. 1a and b).

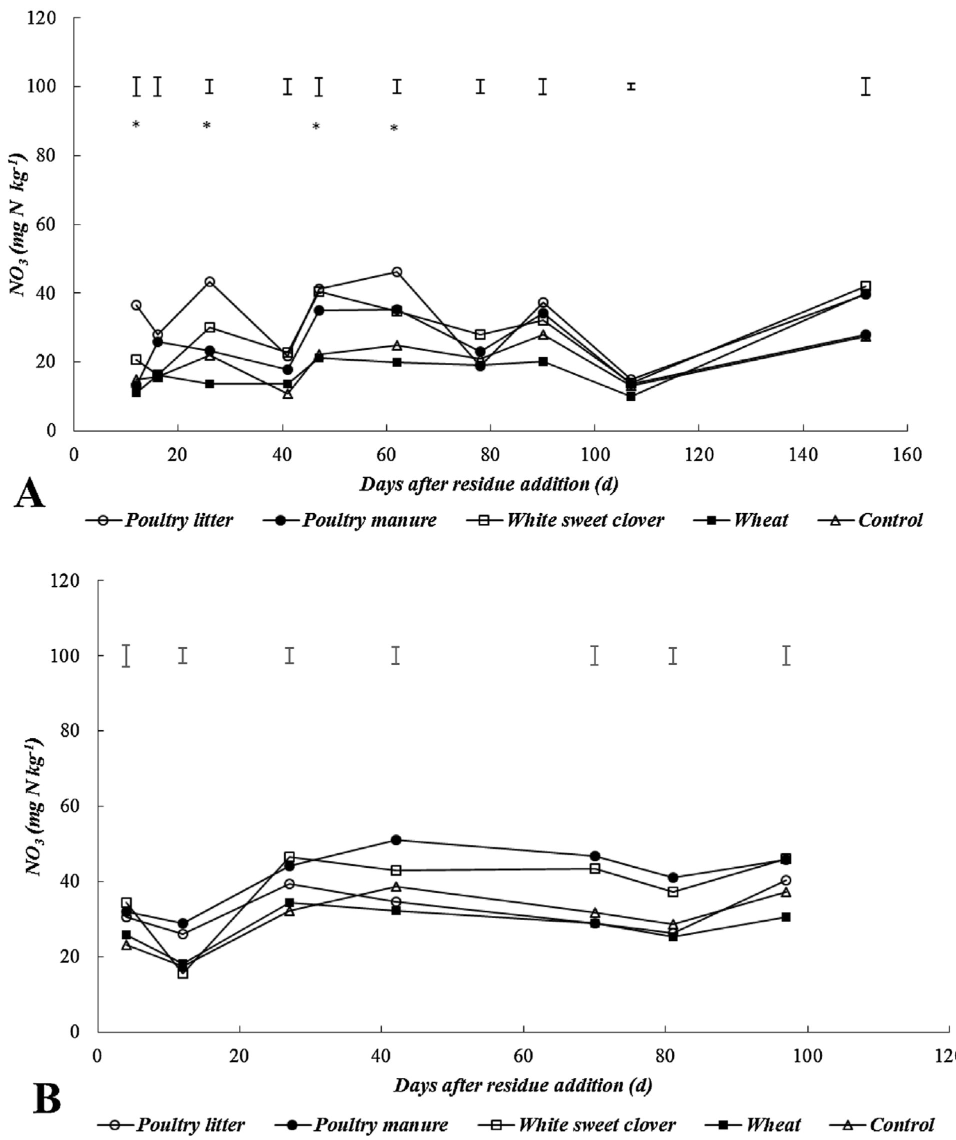

Fig. 2. Evolution of NO_3 after residue addition in (a) Year 1 and (b) Year 2. Error bars indicate standard error of the mean SEM. * indicates significant differences between treatments for each measurement date.

Table 2
Weighted means of N-NO_3 and soil water-filled pore space (WFPS) content in experiments carried-out in two years in Paraná, Argentina.

	N-NO_3 (mg kg^{-1})				WFPS (%)			
	Year 1		Year 2		Year 1		Year 2	
Poultry litter	36.0	a	32.9	ab	37.7	a	30	ab
Poultry manure	27.5	abc	43.6	a	34.7	ab	29	ab
White sweet clover	33.1	ab	39.8	ab	29.7	c	26	b
Wheat	24.4	bc	28.9	b	31.0	bc	26	b
Control	23.6	c	31.7	b	34.5	abc	32	a

Different letters besides means within a column indicate significant differences according to Duncan test (α = 0.05).

3.2. Soil inorganic N and water filled pore spaces

The soil inorganic N as nitrate (N-NO_3) in Year 1 differed between treatments ($P < 0.05$) in four out of ten sampling dates, in which poultry litter had the higher values (Fig. 2a). There were differences ($P < 0.05$) between treatments for the weighted mean of N-NO_3 across sampling dates. In fact, the weighted mean of N-NO_3 for poultry litter, poultry manure and white sweet clover was higher than for the control treatment (Table 2). On the other hand, in Year 2 the N-NO_3 content showed no significant differences between treatments (Fig. 2b) in any sampling date, although the weighted mean of N-NO_3 differed between treatments ($P < 0.05$) showing a similar ranking between treatments (Table 2) than in Year 1.

In Year 1, the water-filled pore space (WFPS) had significant differences ($P < 0.05$) between treatments in six out of ten measurement dates. On the first measurement date, control, poultry manure and poultry litter treatment had significantly higher WFPS compared with the other treatments. On the second measurement date, poultry litter had the highest value, whereas on the other measurement dates poultry manure and poultry litter had the highest values (Fig. 3a). The weighted mean of WFPS was highest for poultry litter (Table 2). In Year 2, WFPS only differed ($P < 0.05$) between treatments on one measurement date, where the control had the higher value and white sweet clover had the lower value (Fig. 3b). The weighted mean of WFPS for Year 2 was highest ($P < 0.05$) in the control treatment (Table 2).

Water-filled pore space ranged from 12 to 54% (on average 34%) in Year 1 and from 15 to 40 (on average 29%) in Year 2. In Year 1, there was a negative relationship between WFPS and N-NO_3 ($P < 0.0001$, $R^2 = 0.44$), whereas in Year 2 these two variables were unrelated ($P > 0.1$).

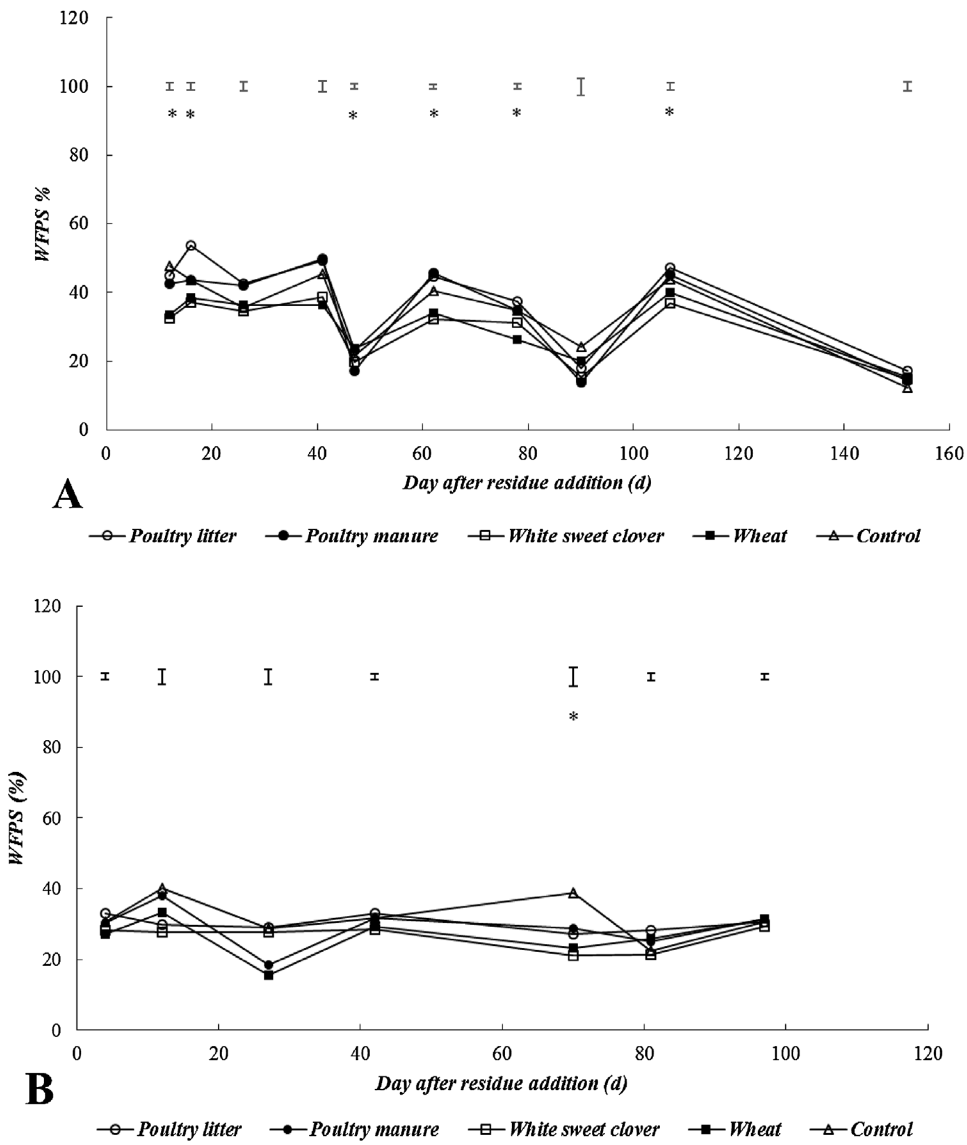

Fig. 3. Evolution of water-filled soil pore space (WFPS) after residue addition in (a) Year 1 and (b) Year 2. Error bars represent standard error of the mean SEM. * indicates significant differences between treatments for each measurement date.

3.3. Nitrous oxide emissions

Flux of N_2O ranged from 0.12 to 50 μg N $m^{-2}d^{-1}$ in Year 1 (Fig. 4a) and from 0.62 to 13.7 μg $m^{-2}d^{-1}$ in Year 2 (Fig. 4b). In Year 1, N_2O flux differed ($P < 0.05$) between treatments on four out of ten measurement dates. At 26 days after addition (DAA), white sweet clover, control and poultry manure had the highest values, whereas poultry manure had higher flux at 41, 48 and 71 DAA (Fig. 4a).

In Year 2, the N_2O flux was extremely low, and only differed ($P < 0.05$) between treatments at 12 and 81 DAA. Wheat and control had higher N_2O fluxes (2.7 and 3.1 μg N m^{-2} d^{-1}) at 12 DAA, whereas poultry manure had higher N_2O flux (13 μg N m^{-2} d^{-1}) at 81 DAA (Fig. 4b).

In year 1, the cumulative N_2O flux had significant differences ($P < 0.05$) only after 62 DDA (Fig. 5a). Total cumulative N_20 flux at the end of the evaluation period in this year was higher in poultry manure (237 g N ha^{-1}) than in the other treatments (Fig. 5a).

In year 2, although the values were extremely low, the cumulative N_2O flux differed significantly between treatments only at 12 DDA ($P < 0.05$), when the control and wheat had higher cumulative fluxes (10.9 and 11.9 g N ha^{-1}) (Fig. 5b). Total cumulative N_2O flux at the end of the measurement period in this year ranged from 52 to 84 g N ha^{-1}, but there was no significant difference between treatments (Fig. 5). Total cumulative N_2O flux in the control treatment in year 1 was double the value in year 2 (104 v. 52 g N ha^{-1}), whereas for poultry manure the total cumulative flux was almost three-fold higher in year 1 than in year 2 (237 v. 83 g N ha^{-1}).

Pooling all data, i.e. taking into account all measurement dates, N_2O flux was significantly ($P < 0.004$; $R^2 = 0.18$ in Year 1 and $P < 0.002$; $R^2 = 0.29$ in year 2) associated with WFPS.

Cumulative N_2O flux in Year 1 was not significantly associated with the weighted mean of WFPS or N-NO_3, whereas in the Year 2 it was significantly associated ($P < 0.045$; $R^2 = 0.2$) with the weighted mean of N-NO_3.

3.4. Topsoil C and N

Total soil N in topsoil, i.e. at 0.05 m depth, only differed significantly ($P < 0.05$) between treatments at the end of Year 2. The treatment with wheat residue addition had higher soil N concentration compared to the other treatments (Table 3). In addition, the same treatment had higher total soil C ($P < 0.05$) than the other treatments in both years. The C:N ratio also only showed significant differences ($P < 0.05$) between treatments in Year 2, when the wheat treatment and control had higher C:N ratios (Table 3).

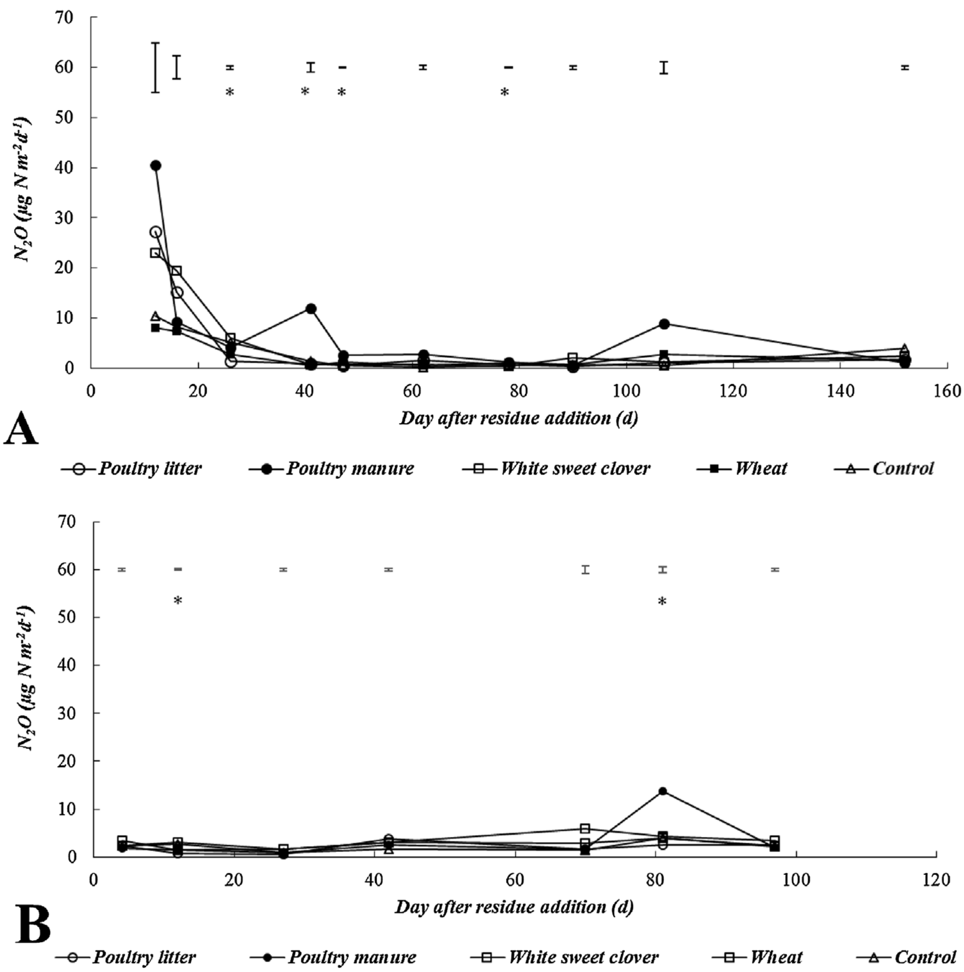

Fig. 4. Flux of nitrous oxide (N_2O) after residue addition in (a) Year 1 (b) Year 2. Error bars represent standard error of the mean SEM. * indicates significant differences among residues on each date.

4. Discussion

Our experimental approach, without crops or living vegetal cover during two years, allowed us to evaluate the actual effect of the residue addition on the studied variables. This approach, however, does not take into account the important buffer effect of the living vegetal cover with respect to several key aspects such as soil moisture, soil N-NO_3, soil temperature and microbiological activity. This buffer effect of living vegetal cover on N_2O emissions has been clearly reported by López-Fernández et al. (2007) and Barton et al. (2008, 2011).

In our study, the addition of residues as either organic amendments or crop residues led to changes in soil N-NO_3 and WFPS (Figs. 2, 3 and Table 2), both of which are commonly reported as being associated with N_2O emissions (e.g. Alvarez et al., 2012; Cosentino et al., 2013). Various other studies have also reported that soil N-NO_3 was altered after the addition of either organic amendments (Masaka et al., 2016; Pino et al., 2008; Roig et al., 2012) or crop residues (Díaz Zorita and Grove, 2002).

The values of N_2O flux ranged from 0.1 to 71.9 µg N m^2 h^{-1} in Year 1 and from 0.2 to 20.2 µg N m^2 h^{-1} in Year 2, which are closer to the reported values for fallows (e.g. Barton et al., 2011; Alvarez et al., 2012), natural grasslands (e.g. Perdomo et al., 2009) and manure than with the values reported for cropped lands (Alvarez et al., 2012; Perdomo et al., 2009). In fact, our average values of 3.78 µg N m^2 h^{-1} (Year 1) and 2.77 µg N m^2 h^{-1} (Year 2) were noticeably lower than the mean flux of 21.4 ± 5 µg m^2 h^{-1} that were reported in our region for cropped lands under similar conditions, i.e. when mean temperature was higher than 23 °C and WFPS was lower than 58% (Cosentino et al., 2013). The lack of a living vegetal cover that buffers soil evaporation rate, and high temperatures in topsoil (Fig. 1), may have led to a fast reduction of WFPS (Fig. 3 and Table 2) in the topsoil, even in Year 1 which was the rainier.

However, N_2O flux was associated with WFPS in both years, as has been reported in various other studies (Alvarez et al., 2012; Baral et al., 2016; Metay et al., 2007; Saggar et al., 2004). Moreover, N_2O flux has been reported as being linearly increased with increase of WFPS in the range of 30 to 60% (Linn and Doran, 1984). Accordingly, in our study, cumulative N_2O emission was 26% higher in Year 1 than in Year 2 for the control treatment and was as high as 82% for the poultry manure treatment. These differences in emissions could be mainly attributable to the higher WFPS in Year 2 than in Year 1 (Table 2), since the differences in soil temperature were negligible between years.

Cumulative values of N_2O emissions ranged from 84 to 237 g N ha^{-1} in Year 1 and from 52 to 85 g N ha^{-1} in Year 2, which are much lower than the reported values for several cropped species of the family Fabaceae, such as alfalfa (1450 g N ha^{-1}) and soybean (3080 g N ha^{-1}) (Rochette et al., 2004). However, our accumulated values were at the lower limit of the range reported by Masaka et al. (210–740 g N ha^{-1}) for different quality of manure addition in rape (*Brassica napus* L.) and tomato (*Solanum lycopersicum* L.), but were higher than those reported for cover crops under direct seeding (31 To 35 g N ha^{-1}) by Metay et al. (2007) in Brazil.

Emission of N_2O has been reported as being mainly generated from denitrification processes when WFPS > 60% (Baral et al., 2016; Linn and Doran, 1984; Metay et al., 2007), whereas with lower WFPS levels, nitrification processes become relatively more important. Emissions of N_2O from nitrification are usually less than those of denitrification (Castaldi, 2000; Smith et al., 2003), which provide an explanation of our extremely low N_2O emissions values. Thus, our low WFPS values (Fig. 3) (< 53%) suggest a predominance of the nitrification process

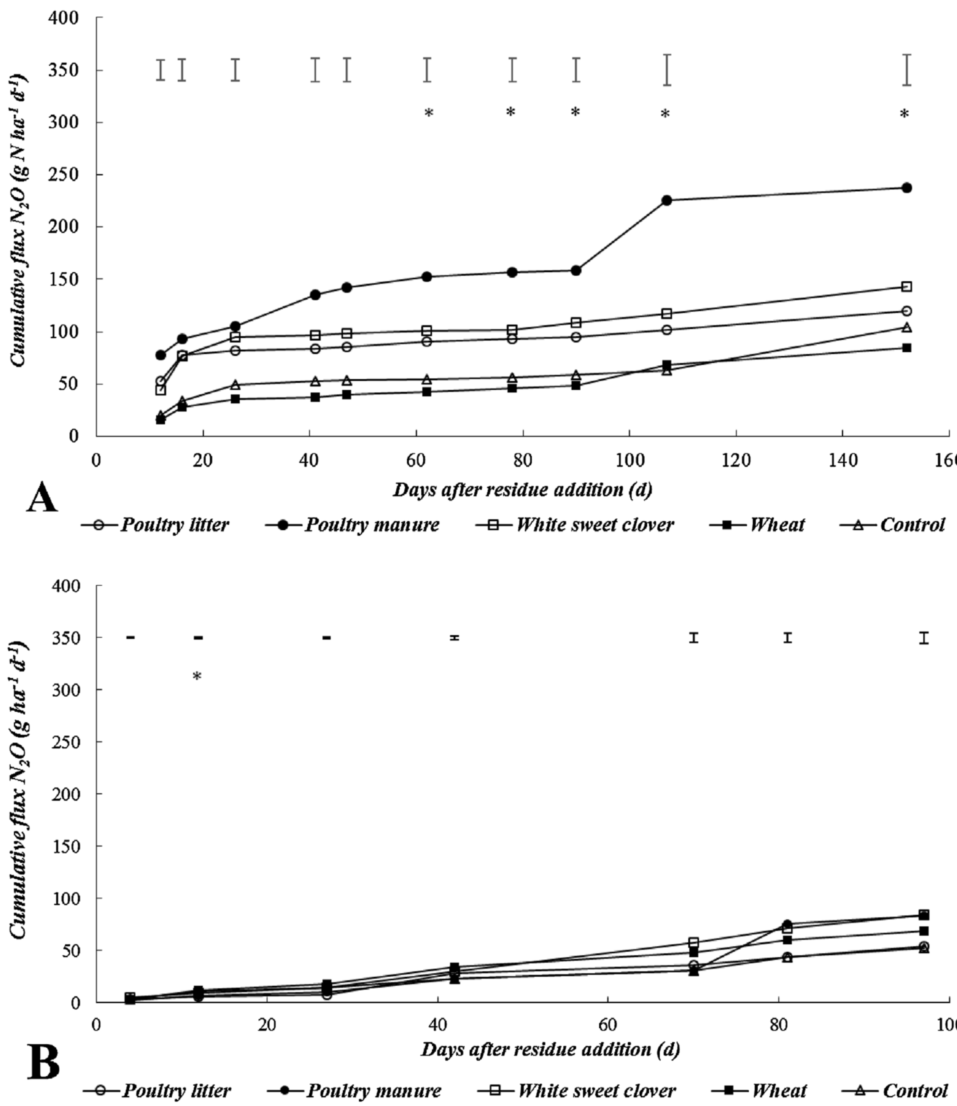

Fig. 5. Cumulative N_2O flux after residue addition in (a) Year 1 and (b) Year 2. Error bars indicate standard error of the mean SEM. * indicates significant differences between treatments for each measurement date.

Table 3
Topsoil (0–0.05 m depth) total nitrogen (N), total carbon (C) and the C:N ratio at the onset (initial), at the end of Year 1 and the end of Year 2 in five treatments of residue addition in experiments carried-out in two years in Paraná, Argentina.

	N (%)					C (%)					C:N				
	Initial	Year 1		Year 2		Initial	Year 1		Year 2		Initial	Year 1		Year 2	
Poultry manure	0.14	0.13	a	0.11	c	1.63	1.47	b	1.25	b	11.8	11.7	b	11.8	a
Poultry litter		0.15	a	0.11	c		1.70	ab	1.32	b		11.4	b	11.8	a
White sweet clover		0.14	a	0.13	b		1.60	ab	1.63	a		11.7	b	12.6	a
Wheat		0.15	a	0.15	a		1.97	a	1.86	a		12.6	a	12.4	a
Control		0.13	a	0.11	c		1.65	ab	1.28	b		12.7	a	11.8	a

Different letters besides means, within a column, indicate significant differences according to Duncan test (α = 0.05).

(Linn and Doran, 1984; Smith et al., 2003), which may lead to our extremely low N_2O emission values (Fig. 4). Accordingly, the highest N_2O emission in Year 1 than in Year 2 could be attributed to higher WFPS, which was related with more abundant and frequent rainfall. Thus, the variation in WFPS between years may have altered the ratio between denitrification and nitrification processes.

In the rainier Year 1, mineral N may have been lost from the 0.05 m topsoil through processes other than N_2O emissions, such as N-NO_3 leaching or runoff, which may have reduced soil N-NO_3 concentration when WFPS was higher, as was demonstrated by the detected negative correlation between these two variables (not shown). Additionally, this N-NO_3 reduction with the increase in WFPS provides an explanation for the lack of association between N-NO_3 and N_2O flux in this Year.

In both years, poultry manure had the higher N_2O flux at some measurement dates (Fig. 4), although it was not related with soil N-NO_3 nor with WFPS at a depth of 0.05m. Probably, the chemical composition of this organic amendment (Table 1), characterised by a C:N ratio that is similar to the soil, could be conducive to the growing of a different microbiological community than in the other treatments. The lack of association between N_2O flux and soil N-NO_3 has already been reported in some situations, such as Alvarez et al. (2012) during fallow periods in Argentina. Moreover, Rochette et al. (2004) concluded that mineral N was a poor predictor of N_2O emissions in prairies and crops of Canada. Therefore, the relationship between instantaneous N_2O flux and soil N-NO_3 seems to be not strongly sustained for the previous studies or our results, then the flux could be related with others soil conditions such as WFPS, temperature and substrate composition. These conditions may affect the relative abundance of nitrifiers and denitrifiers, where N-NO_3 may be alternatively the substrate or the final product. Thus, a soil environment that is favourable to nitrifiers could result in higher levels of N-NO_3 with low levels of instantaneous N_2O flux.

On the other hand, we found that the cumulative N_2O flux was positively associated with the weighted mean of N-NO_3 in Year 2, characterised by lower rainfall than in Year 1. This corresponds with results by Aita et al. (2015), who reported a close association between cumulative emission and exposure to mineral N, an index similar to our weighted mean. This finding reflects that, although mineral N does not necessarily need to be linked with instantaneous N_2O flux, the high level of mineral N during the entire measurement period can be a determining factor of total emissions. Surprisingly, N_2O flux and weighted mean of N-NO_3 in rainier Year 1 were unrelated, which could be attributed to the lack of temporal coincidence between a high WFPS with high N-NO_3, as reflected by the recorded negative relationship between N-NO_3 and WFPS.

In spite of the fact that there was no living vegetal cover during the two-year period, the addition of wheat residues increased topsoil C and N (Table 3), when considering the final and initial values, compared with the other treatments. In Year 2, the addition of white sweet clover residues, however, kept topsoil C unchanged with a reduction of N, compared with initial values (Table 3).

These results reflect the importance of chemical composition of added residues on soil C and N balance, which was contrasting between wheat and white sweet clover (Table 1). On the other hand, in treatments with organic amendment addition, as well as in the control, a reduction in topsoil C and N was recorded after the two-year period (Table 3) in spite of the addition of 10 t C ha^{-1} (in the two-year period) on a topsoil with an stock of only 7.4 t C ha^{-1}.

Reduction of topsoil C and N in the control treatment is an anticipated consequence of the lack of input of both elements during a two-year period. In fact, a minimum annual input of C has been suggested in order to maintain the soil C level, by several reports (e.g. Andriulo et al., 1999). However, reduction of topsoil C and N in organic amendment treatments suggests a priming effect (Kuzyakov, 2010; Kuzyakov et al., 2000), which involves the utilisation of native soil C by microbiota, driven by the excess of added N, which is reflected by the higher C:N ratio of the residues (Table 1).

Collectively, our results demonstrate the actual effect of residue addition on N_2O emissions and on topsoil C and N without the interference of living vegetal cover on key variables. For these experimental conditions, only the addition of a residue with a high C:N ratio matched an increase in topsoil C and N with low N_2O emissions.

5. Conclusions

Although N_2O emissions were extremely low, the highest values were recorded in organic amendments, mainly in poultry litter, whereas the lower values were recorded in crop residues and in the control treatment.

The residue addition affected the soil variables that are commonly reported as being associated with N_2O emissions, i.e N-NO_3 and WFPS, although this was dependent on the amount and frequency of rainfall. In spite of the fact that WFPS was associated with N_2O flux in the two-year period, emissions were higher in rainier Year 1 than in Year 2.

Although all treatments added 10 t C ha^{-1} after two years, an increase in topsoil C and N was only recorded in the wheat treatment with higher C:N ratio, which suggests a priming effect in the other treatments.

Overall, our results suggest that in our region the widespread adoption of no-till management with the addition of organic amendments broadcast 30–45 days before sowing of summer crops is associated with low N_2O emissions and potential improvements in soil quality. Further, our approach of not having any plant cover allowed us to identify more clearly the drivers of N_2O emissions and also to better understand the involved soil processes in this particular situation, without the presence of a living vegetal cover. It provided new knowledge to develop practices that are oriented to soil restoration with low pollution, since we used composted amendments to reduce their pollutant load and evaluated the N_2O fluxes, a mandatory issue in sustainable soil management.

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