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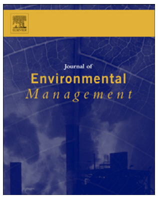

Research article

Potential role of compost and green manure amendment to mitigate soil GHGs emissions in Mediterranean drip irrigated maize production systems

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A B S T R A C T

Organic fertilization can preserve soil organic matter (SOM) and is foreseen as an effective strategy to reduce green house gases (GHGs) emissions in agriculture. However, its effectiveness needs to be clarified under specific climate, crop management and soil characteristics. A field experiment was carried out in a Mediterranean drip irrigated maize system to assess the pattern of soil CO_2 and N_2O fluxes in response to the replacement of a typical bare fallow–maize cycle under urea fertilization (130 kg N ha^{-1} y^{-1}) (CONV) with: (i) bare fallow-maize cycles under two doses of compost (COM1 and COM2, 130 and 260 kg N ha^{-1} y^{-1}, respectively) and (ii) a vetch-maize cycle, with vetch incorporation as green manure (130 kg N ha^{-1} y^{-1}) (GMAN).

Along the maize period (MP), reduced daily N_2O emissions were detected in organic treated soils compared to CONV, mainly in the first stages of the cultivation, thanks to the slow release of available nitrogen from the organic substrates. Cumulative N_2O fluxes (kg N_2O-N ha^{-1}) in MP scored to 0.24, 0.14, 0.12 and 0.085 for CONV, COM1, COM2 and GMAN, respectively, with significantly lower emissions in GMAN respect to CONV. CO_2 fluxes partially reflected the ranking observed for maize yields, with cumulated values (Mg CO_2-C ha^{-1}) of 2.2, 1.5, 2.1, 2.1 for CONV, COM1, COM2 and GMAN, respectively, and significantly lower in COM1 respect to the other treatments.

During the fallow period (FP), compared to CONV (0.77 Mg CO_2-C ha^{-1} and 0.25 kg N_2O-N ha^{-1}), enhanced GHG fluxes were detected in COM treatments (about 0.90 Mg CO_2-C ha^{-1} and 0.37 kg N_2O-N ha^{-1}, as averaged values from COM1 and COM2), likely driven by the slow prolonged mineralization of the added organic matter. GMAN showed comparable CO_2 (0.82 Mg CO_2-C ha^{-1}) and N_2O emissions (0.30 kg N_2O-N ha^{-1}), in consequence of restrained post-harvest residual N coupled with the counteracting effect of vetch uptake.

Respect to the total yearly GHG emissions in CONV (about 194 kg CO_2 eq ha^{-1} y^{-1}), the overall results showed commensurate slightly higher GWP in COM treatments (+11% as averaged value from COM1 and COM2). The yield-scaled global warming potential (GWP) resulted 60% higher and nearly doubled for COM2 and COM1 respectively, according to the lower COM yields, markedly dampening at halved compost dose. GMAN appeared the best performing organic treatment, with lower GWP (−27%) and competitive yields respect to CONV.

All treatments showed N_2O emission factors consistently lower compared with the default IPCC 1% value.

1. Introduction

In the recent scientific debate on Greenhouse Gases (GHGs) mitigation strategies, solid organic amendments are foreseen as effective sustainable tools able to preserve soil fertility and enhance

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soil carbon sequestration of atmospheric CO_2, at the same time potentially constraining soil GHG emissions in agroecosystems (Aguilera et al., 2013; Alluvione et al., 2010; Diacono and Montemurro, 2010; Franzluebbers, 2005; Freibauer et al., 2004; Gregorich et al., 2005; Lal, 2004; Li et al., 2013; Smith, 2004; Su, 2007; Tuomisto et al., 2012).

Currently, GHGs from the agriculture sector contributes to about 10–12% of total global anthropogenic GHG emissions (IPCC, 2007). Nitrogen input is a major driver of agricultural N_2O-N losses (Tuomisto et al., 2012), contributing to about 60% of the global anthropogenic N_2O emission (IPCC, 2007). Otherwise, the net CO_2 flux from cropped systems is assumed to be nearly balanced (IPCC, 2007), even if croplands can be both a source and a sink for CO_2 depending on the specific agronomic management (La Scala et al., 2006; Mangalassery et al., 2014; Tuomisto et al., 2012).

Organic fertilizations through compost and legume green manure can maintain the organic C levels in arable soils, increase soil C sequestration and sustain crop production by recycled and biologically fixed nitrogen (Alluvione et al., 2013; Grignani et al., 2007; Li et al., 2013; Melero et al., 2007; Tejada et al., 2008; Triberti et al., 2008).

Composting (of municipal solid waste-MSW, sewage sludge, green waste, etc.), highly recommended by the European policy on waste management (2008/98/EC), can both reduce the use of artificial fertilizers and preserve or enhance soil fertility increasing organic C and total N (Fagnano et al., 2011; Mantovi et al., 2005; Ros et al., 2006; Smith, 2004; Tuomisto et al., 2012). Moreover the use of compost can: (i) protect stabilized SOM (Lynch et al., 2006; Piccolo et al., 2004; Spaccini et al., 2002) and (ii) increase aggregate stability (Diacono and Montemurro, 2010; Spaccini and Piccolo, 2012). Green manure from legume intercrops: (i) enhance soil aggregate stability and protect soil from erosion (Gómez et al., 2009); (ii) promote N retention both in the fallow and cash crop periods (Cherr et al., 2006; Gabriel and Quemada, 2011).

However, the effective nexus between potential increase in C storage and decrease of soil GHG emissions is still controversial (Li et al., 2005). For instance, depending on the specific pedo-climatic context and the relative substrates composition in easily biodegradable or recalcitrant C compounds, organic fertilizers may contextually enhance soil CO_2 emissions (Ding et al., 2007; Li et al., 2013) and N_2O fluxes (Aguilera et al., 2013; Rochette et al., 2007) respect to mineral fertilization management in maize cultivation systems. Taking into account that N_2O global warming potential (GWP) is 265 times greater than CO_2, N_2O emissions might be enough elevated to negate some of the beneficial effects of organic amendments on soil properties (Heller et al., 2010).

The controversial topic need to be further specifically addressed in the Mediterranean environment (Aguilera et al., 2013). In this climatic areas the high summer temperatures coupled with low precipitations strongly influence nutrient and water availability, crop development, and microbial activity. As a result, the pattern of soil GHG emissions (especially N_2O fluxes) is strongly driven by the interaction of N fertilization with irrigation water inputs (Aguilera et al., 2013; Ranucci et al., 2011; Sánchez et al., 2001; Vallejo et al., 2004). In this regard the irrigation management appeared able to tune the potential of N_2O mitigation by organic fertilization, with the slowest progression of N_2O fluxes under low water input regimes (Aguilera et al., 2013; Sánchez-Martín et al., 2008, 2010a).

However, the overall effectiveness of organic fertilization might be limited by: (i) the effect of post-harvest residual mineral N leading to potential relevant N_2O-N losses during the fallow intercrop period (Aguilera et al., 2013); (ii) lower agronomic yield relative to the conventional mineral management, with a consequent levelling of the environmental benefits (including GHG mitigation) of alternative organic fertilizers respect to synthetic ones (Diacono and Montemurro, 2010; Meier et al., 2015; Tuomisto et al., 2012). In this regard crop-specific and yield-scaled GHG emissions appear necessary for a proper assessment of potential tradeoffs between reduced synthetic N input, yield performance and GHG emissions (Aguilera et al., 2013; Meier et al., 2015; Sanz-Cobena et al., 2014; Venterea et al., 2011).

Currently, there are only few studies investigating the effect of compost from MSW and cover crop green manuring (vetch included) on CO_2 and N_2O emissions in Mediterranean maize cropping systems. Cumulated nitrous oxide losses during the maize crop season under fertilization with composted MSW resulted lower in comparison to the conventional management with synthetic fertilizer, however no information were available for the intercrop period (López-Fernández et al., 2007; Meijide et al., 2007). Differently the net GHG benefit of maize managed with cover crop green manuring (as alternative to synthetic fertilization), referred to a whole year time window, appeared less marked, due to potential counteracting GHG emission patterns along the different main crop and intercrops periods (Guardia et al., 2016; Sanz-Cobena et al., 2014).

The objective of this work was to address the effect of compost fertilization (at two levels) and vetch green manuring (versus conventional fertilization with urea) on soil area and yield-scaled CO_2 and N_2O emissions in a Mediterranean drip-irrigated maize cultivation system, along both the main spring-summer crop and the intercrop periods. This, in order to achieve a deeper evaluation of the potential GHG mitigation for the investigated organic fertilized crop production systems, which is a major driver of their whole foreseen sustainability.

2. Materials and methods

2.1. Study site

The experimental farm was located in the coastal plain of the Sele River, Southern Italy (40°37′N, 14°58′E, 30 m above sea level) which is characterized by Mediterranean climate, with dry summer and precipitation mostly occurring in autumn-winter. See Alluvione et al. (2013) for further details.

Soil at the study site is a Vertic Haploxeralf (USDA soil taxonomy; Soil Survey Staff, 2014), with a sandy-clay-loam texture. Main soil properties of the plowed layer (0–30 cm) are: sand 46.5% silt 22.3% clay 31.2%, bulk density 1.42 g cm^{-3}, pH 7.4 (1 soil:2.5 water), 7.5 g organic C kg^{-1}, 0.9 g total N kg^{-1}, 31.5 mg Olsen P kg^{-1}, and 90.6 cmol exchangeable K kg^{-1}.

2.2. Experimental design and crop management

The study focused on maize cultivation (*Zea mays* L) under drip irrigation since: (i) maize for silage is widespread in the Campania region and in non-zootechnics farms could be potentially interested in the use of compost or green manure as alternatives to mineral fertilization; (ii) drip irrigation is a diffused water-saving management in Southern Italy (Natali et al., 2009), currently contemplated for maize cultivation in the study area (Regione Campania, 2014).

Soil GHGs monitoring were conceived in the framework of the wider MESCOSAGR project, which aimed to assess, in the short term (within a three-year time frame window), the effectiveness of alternative C friendly strategies in increasing soil organic matter and supporting proper N fertility for maize yields, under different soil properties and climatic conditions.

An agricultural field previously cropped with durum wheat was converted to maize cultivation in fall 2006 subject to alternative N fertilizer managements: (i) a conventional bare fallow—maize cycle under fall mouldboard plowing (30 cm depth) and urea fertilization at maize sowing (130 kg N ha^{-1} yr^{-1}) (**CONV**); (ii) bare fallow-maize cycles under fall mouldboard plowing (30 cm depth) and different doses of compost incorporation at maize sowing (**COM1** and **COM2**, corresponding to a N supply of about 130 and 260 kg N ha^{-1} yr^{-1}, respectively); (iii) a *Vicia villosa* L -maize cycle, under fall mouldboard plowing (30 cm depth) and vetch chopping and incorporation as green manure at maize sowing (**GMAN**, corresponding to a N supply of about 130 kg ha^{-1} yr^{-1}, mainly from legume N_2 fixation). A non N-fertilized control mouldboard plowed at 30 cm was also arranged (**CONT**), in order to correct cumulative N_2O fluxes for the baseline flux.

All treatments were arranged in a randomized complete block design with four replicates on 30 m^2 (6 × 5 m) plots and repeated on the same plots during three consecutive years from 2006 to 2008.

Each year, all treatments were mouldboard plowed at 30 cm on the same day in mid-October. In GMAN, a further pass of rotary hoe was carried out for vetch seedbed preparation and sowing in late October- early November; no fertilizers were supplied for vetch cultivation. Along the maize period, agronomic practices for seedbed preparation, sowing and irrigation were carried out similarly and simultaneously for all treatments. Two passes of rotary harrow were performed in early May for seedbed preparation and incorporation of fertilizers, which included for all treatments additional 100 kg P_2O_5 ha^{-1} y^{-1} as triple superphosphate and 200 kg K_2O ha^{-1} y^{-1} as muriate of potash. All fertilizers were incorporated on the same day they were distributed. The compost was always stable and fully mature given its C/N ratio close to 10 (Silva et al., 2007). The hairy vetch was shredded just a few days before soil incorporation at bud stage (early May), with a C/N ration of about 16. Based on a vetch aboveground yield of about 0.33 kg m^{-2} (dry basis) with about 3.5% total aboveground N, to compute the N supplied to the field through the vetch incorporation, the study considered a shoot to root N ratio of 2.1 and N-fixation as about 70% of the uptake (Büchi et al., 2015; Gabriel and Quemada, 2011). Corn (*Zea mays* L.) was sown to a density of 7.4 seeds per m^2 on the same date the seedbed preparation and the fertilization were performed.

Water supply was carried out by means of a drip irrigation system, with one drip tape line for each row of plants (dripline spacing 0.7 m, emitter spacing 0.40 m). Water inputs were managed to balance the estimated evapotranspiration losses (Hargreaves and Samani, 1985). Specifically, eleven irrigations (about 20–35 mm of water for each application) were scheduled weekly and biweekly from May 12 and August 13 in 2008, for a total irrigation water supply of 362 mm, contributing to about 80% of the total water input (irrigation + rain) during the maize period.

Further details about the experimental design can also be found in Alluvione et al. (2009, 2013), Grignani et al. (2012) and Fierro and Forte (2012).

2.3. Soil CO_2 and N_2O fluxes measurements

Soil CO_2 and N_2O fluxes were monitored during a one-year time window along the alternation of the main spring-summer crop cultivation and the intercropping fallow period. The monitoring activities were carried out two years after the beginning of the project, in order to constrain possible residual effects resulting from the previous soil management. The campaigns started in October 2007 (October 11th) and continued in November, December, January and late April- early May, to monitor the GHGs trend along the 2007/2008 fallow period (FP), from the time for fall mouldboard ploughing up to spring, soon before maize sowing. Afterwards, GHGs were monitored throughout the following spring-summer maize period (MP), from sowing (May 2008 6th) to August 2008, soon before harvest (August 20th). At this stage the monitoring campaigns were protracted post harvest till early October 2008 (soon before the successive yearly main tillage), to catch the residual GHG emissions throughout the drought post-harvest phase and the following first autumn rains in the 2008/2009 FP.

Measurements of daily bi-hourly CO_2 and N_2O emissions were carried out by means of automated closed static stainless steel chambers ($Ø = 30$ cm, $h = 10$ cm), equipped with white teflon coated lids automatically opening 180° (to avoid shading on monitored soil) and controlled by a multipoint auto-sampler coupled to a 1412- Photoacoustic Field Gas Monitor (Innova). The monitoring was performed by a single chamber for each treatment, installed in the inter-row area next to the drip tape lines, in order to catch the water distribution gradient around the drip points. Each chamber was equipped with a vent valve to avoid pressure variations inside (Bain et al., 2006) and inlet and outlet tubes allowing air circulation from chamber to detection instrument. Due to the unsatisfactory automated corrections for water and cross interference performed by the photoacoustic system, a silica-gel buffer was mounted in-line to minimize water vapour fluctuations. Moreover, a correction factor was derived for N_2O (+0.05 ppb for each ppm of CO_2), through calibration by means of pure N_2O gas and mixed CH_4-N_2O-CO_2 gas standards and further output data check by GC analyses (See Forte et al., 2016 for details).

Soil gases fluxes were calculated for each chamber, considering a cycle of 3–5 measurements with open chamber and 10–12 measurements with closed chamber, covering a total time of about 10 min. The gas flux was expressed as:

$$F_{(GHG)} = (\Delta[GHG]/ \Delta t) \text{ x } (V/S)$$

where $F_{(GHG)}$ was the soil flux of gas, $\Delta[GHG]$ was its variation of concentration expressed as mg m^{-3}, in the time interval Δt, V was the chamber's volume and S the covered soil surface.

The soil gases fluxes were computed on one-hour time scale according to the following equation:

$$[GHG]_t = a(t\text{-}t_0)$$

where $[GHG]_t$ and $[GHG]_0$ were the gas concentrations at time t and on first measurement performed with closed chamber, respectively. Hourly fluxes were calculated by multiplying the angular coefficient *a* with *S/V*.

The linear regression of GHGs accumulation was checked and the measurement cycles with R^2 less than 0.8 were rejected. Threshold detection values for N_2O fluxes and CO_2 emissions were 0.01 g CO_2-C m^{-2} d^{-1} and 0.05 μg N_2O-N m^{-2} d^{-1}, respectively.

Bi-hourly measurements were summed up to calculate daily-integrated GHG fluxes and cumulative emissions, with linear interpolation of gaps along the acquisition period.

Even if the GHG acquisition was performed by a single permanent chamber for each treatment, monitoring campaigns along the fallow and maize periods were performed to get an estimate of the spatial variability of the process. Specifically GHG fluxes (and linked auxiliary parameters including soil temperature and WFPS) were periodically monitored from other two supplementary chambers in each treatment, in order to compute coefficient of variations (CV%) for both CO_2 and N_2O fluxes in the two different monitored periods (See Forte et al., 2016 for further details). Values detected for CVs% in the FP were homogeneously in the mean range of about 20% and 30% for CO_2 and N_2O fluxes, respectively. In the MP, average values of CVs% for CO_2 emissions were about 11%, 27% and 13% for CONV, COM treatments and GMAN, respectively; average CVs% for N_2O

emissions were about 34%, 57% and 37% for CONV, COM treatments and GMAN. CVs% were used to derive the confidence interval related to the detected GHG fluxes according to Forte et al. (2016).

Emission factors (EFs) related to the whole one-year time frame were computed dividing the yearly fertilized induced N_2O-N losses (FIE) by the amount of fertilizer N added at maize sowing. In this regard, FIE were computed by subtracting the cumulative baseline flux detected in CONT from the cumulative N_2O fluxes recorded in each different treatment.

To convert the total cumulate GHGs emissions into Carbon dioxide equivalents (CO_2eq), Global Warming Potential (GWP) of gases were retrieved from the most recent pertinent IPCC guidelines (IPCC, 2006), using the relevant 100-year timeframe (GWP100).

2.4. Crop and soil parameters

Crop yield was estimated by collecting aboveground biomass at dent stage from an area of 15 m^2 (5 m in 4 rows with 0.75 m interrow spacing) per plot. Plant samples were oven-dried at 70 °C until a constant weight and C and N content in plant tissues was measured by using a CHN elemental analyzer (NA 1500 N analyzer, Thermo Fisher Scientific, Waltham, MA) after the dynamic flash combustion of the samples.

Maize crop yields, in the year of GHGs monitoring (2008), were 15.5 Mg d.m. ha^{-1}, 6.9 Mg d.m. ha^{-1}, 9.3 Mg d.m. ha^{-1} and 12.5 Mg d.m. ha^{-1}, for CT, COM1, COM2 and GMAN, respectively.

Crop evapotranspiration was calculated by multiplying the reference evapotranspiration (Hargreaves and Samani, 1985) by crop coefficients (Allen et al., 1998).

During the FP, soil temperature was measured on campaign basis down to a depth of 10 cm by means of a thermo-pHmeter (Hanna Instruments). On the same sampling dates, intact soil cores were collected and processed to derive the soil water filled pore space (WFPS), by dividing the volumetric water content (product of gravimetric soil moisture content and bulk density) by total soil porosity (Forte et al., 2016; Sánchez-Martín et al., 2008; 2010a,b; Vallejo et al., 2004).

During the MP, soil moisture and temperature were monitored by a data-logger CR1000 (Campbell Scientific Ltd., Shepshed, UK), equipped with six probes for temperature detection (107 Temperature Probe Campbell Scientific) and three reflectometers for the detection of soil moisture (CS616 Water Content reflectometers – TDR, Campbell Scientific).

Soil nitrates were analysed according to the Hach® method by spectrophotometry (Hach DR, 2000; Hach Company, Loveland, CO) (Alluvione et al., 2013). Soil ammonium was determined by K_2SO_4 extraction (0.5 M) and Orion ion-selective electrodes (Castaldi and Aragosa, 2002).

2.5. Data analysis

The differences of the mean values among the treatments were assessed by means of the "One Way Analysis of Variance" ($P < 0.05$). The relationships among different measured variables were investigated through correlation analysis (Pearson Product Moment Test, $P < 0.05$). For both parametric tests, the normality and equal variance assumptions were checked through the Kolmogorov-Smirnov and Levene's tests, respectively. If necessary, data were log-transformed to satisfy assumptions of normality and homoscedasticity. All the statistical analyses were performed using the Sigma Plot package (Systat Software, Inc, Germany, version 11.0).

3. Results

3.1. Environmental conditions and evolution of mineral N

T_{soil} and WFPS showed homogeneous patterns among the different treatments, with a clear seasonal trend reflecting the patterns of air and soil temperature (Fig. 1).

During the FP, due to the autumn-winter rainfall, the values of WFPS ranged from 57% to 77%, with 67% as average value. Differently, during the MP, WFPS was driven by the irrigation water input and ranged from about 40%, in between irrigation events, to about 53% soon after the water supply.

As it relates to the trend of mineral N in soil, higher nitrate and organic N concentration were detected in the organic treatments respect to CONV at the beginning of the fallow period in 2007, post maize harvest (Table 1). At this stage, the main tillage causes a fast release of mineral N, with the lowest average ammonium concentrations detected in COM treatments. Afterwards an overall homogeneous lowering of soil mineral N was recorded throughout the autumn-winter period, with no significant differences between treatments. At increasing of the spring air and soil temperature (just before the site preparation for maize sowing), the availability of soil mineral nitrogen returned higher in the organic treatments. In the MP, a marked increase of soil mineral N was recorded in CONV, soon after urea incorporation. Also at stem extension, ammonium concentration in CONV remained significantly higher respect to the organic treatments. From tasseling onward, soil mineral N availability was homogeneously low, whilst at physiological maturity, following the second mechanical weeding, increased rates of ammonium release were detected from all plots. At the end of the maize crop cycle, after the harvest, soil mineral (and organic N) returned higher under the organic management (Tables 1 and 2).

3.2. GHG pattern along crop and intercrop periods

Fig. 1 shows the one-year dynamic of daily CO_2 and N_2O fluxes from soil of the different treatments along the alternation of fallow and maize periods, from the autumnal main tillage in 2007, throughout the spring-summer maize cultivation in 2008, up to the drought post-harvest phase and the first autumn rains, before the following deep ploughing in 2008. The GHGs trend is reported in relation to the pattern of climatic, crop and soil physical parameters.

At the beginning of the monitoring activities, just before the main tillage in 2007, faster GHGs emissions were detected in COM2 and GMAN compared to CONV (Fig. 1, Tables 3 and 4), at higher soil nitrate availability (Table 1). Soon after the deep ploughing, coupled with the fast release of ammonium (Table 1), it was recorded a short pulse of GHG in CONV, less evident for GMAN and almost negligible for COM treatments (Fig. 1, Tables 3 and 4). During the first month after tillage, GHG emissions remained higher in CONV respect to the organic treatments (Fig. 1, Tables 3 and 4). From the second to the fourth month after tillage, for both the COM trials the pattern reversed toward flux activities markedly higher in comparison to CONV, without significant differences between the different compost doses (Fig. 1, Tables 3 and 4). For GMAN, the same time-window corresponded to the emergence and further growth of vetch and showed comparable GHG fluxes with CONV (Fig. 1, Tables 3 and 4). In the seventh month after tillage, just before the site preparation for maize sowing, a general increase of soil CO_2 emissions was recorded and a trend toward higher average fluxes persisted for COM treatments (Fig. 1, Table 3). Differently, N_2O fluxes appeared homogeneous and basal, at WFPS close to 40% (Fig. 1, Table 4).

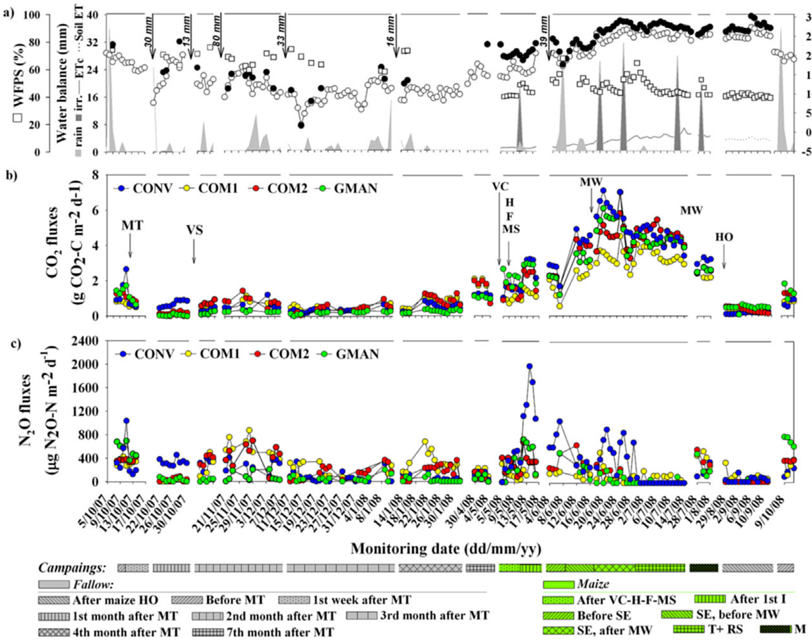

Fig. 1. The figure shows: (a) the pattern of climatic variables (daily average air temperature and rainfall), soil parameters (daily mean soil temperature and water filled pore space at 0–10 cm depth), irrigation water input, crop and soil evapotranspiration; (b) soil CO_2-C emissions; (c) soil N_2O-N emissions. The data are referred to the fallow and maize periods for each treatment. In absence of significant differences, daily mean soil temperature and water filled pore space (WFPS) at 0–10 cm depth are reported as average value for all treatments. Legend: *MT, main tillage (11-10, 2007); VS, vetch sowing (07-11, 2007); VC, vetch chopping (03-05, 2008); H harrowing (05-05, 2008); F, fertilization (05-05, 2008); MS, maize sowing (06-05, 2008); I, irrigation; SE, stem extension; MW, mechanical weeding; T, tasseling; RP, reproductive stages; HO, harvest operations (20-08, 2008).*

Table 1
Mean value and standard error of the mean (SE) for soil mineral nitrogen in the 0–15 cm layer of each treatment during the maize cultivation and the fallow periods. Different apical letters indicate significant differences (One way ANOVA, $P < 0.05$, $n = 4$).

Date	Soil Mineral N (μg g^{-1})															
	NO_3-N (μg g^{-1})								NH_4-N (μg g^{-1})							
	CONV		COM1		COM2		GMAN		CONV		COM1		COM2		GMAN	
	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Fallow 2007/2008																
07/10/2007	6.3^{a}	0.9	8.0^{ab}	0.7	9.3^{b}	0.9	13^{bc}	3	7.5	1.7	8.0	1.5	7	2	10	4
11/10/2007	11.0	1.2	8.0	1.0	10.0	1.3	7.0	1.2	37	4	28	5	25	5	46	7
17/10/2007	8.0	0.8	7.0	0.9	7.0	0.9	8.0	1.4	19.0	1.9	23	4	18	3	22	3
20/12/2007	5.0	0.5	5.0	0.6	6.0	0.8	6.0	1.0	7.0	0.8	3.0	0.6	3.0	0.5	8.0	1.2
09/01/2008	8.3	0.9	8.0	1.0	9.0	1.2	9.0	1.5	9.0	0.9	1.0	0.2	12	2	3.0	0.5
21/01/2008	6.0	0.7	6.0	0.7	8.0	1.1	7.0	1.2	3.7	0.4	5.0	0.9	6.0	1.2	3.0	0.5
03/05/2008	9^{a}	3	15.7^{b}	1.4	14^{b}	2	18.4^{b}	1.3	11	5	8.0	0.7	4.6	0.6	6.8	1.7
Maize 2008																
10/05/2008	26^{a}	3	15.0^{b}	1.8	13.5^{b}	1.8	19.3^{b}	1.3	19^{a}	3	7.5^{b}	1.4	4.08^{b}	0.8	7.0^{b}	1.1
17/06/2008	7.5	0.8	6.0	0.7	5.0	0.7	8.0	1.3	12.5^{a}	1.4	3.5^{b}	0.6	5.0^{b}	1.0	7.0^{b}	1.1
07/07/2008	6.0	0.7	5.5	0.4	4.5	0.6	4.5	1.3	8.5	0.5	7.5	1.6	8.5	1.4	10.7	0.9
31/07/2008	5.0	50.8	7.0	0.4	7.0	0.9	7.0	1.3	16	0.8	17.5	3	17.0	1.2	17.0	1.4
Fallow 2008/2009																
28/08/2008	9.7^{a}	1.0	14.8^{b}	1.6	16^{b}	3	16.4^{b}	1.3	12.6	1.6	6	2	6.7	1.6	8.4	1.0
05/09/2008	11.0	1.3	7.0	0.8	8.0	1.3	8.0	1.3	11	0.8	9	3	13	3	12.5	1.5
16/09/2008	5.0	0.5	4.0	0.4	6.0	1.0	6.0	1.3	9.0	0.5	15	5	14	2	12	1.5

Table 2
Mean value and standard error of the mean (SE) for soil organic N in the 0–15 cm layer of each treatment at the end of the maize cropping cycle (post-harvest) in 2007 and 2008. Different apical letters indicate significantly differences (One way ANOVA, $P < 0.05$, $n = 4$).

Sampling time	Soil Organic N (g kg^{-1})							
	CONV		COM1		COM2		GMAN	
	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Post maize harvest 2007	0.73^{a}	0.04	0.90^{b}	0.06	0.99^{b}	0.04	0.81^{a}	0.09
Post maize harvest 2008	0.83^{a}	0.07	0.94^{ab}	0.09	1.01^{b}	0.09	0.87^{a}	0.13

Table 3
Mean value and standard error of the mean (SE) for soil daily CO_2 emissions in CONV, COM1, COM2 and GMAN treatments along the different monitoring campaigns during the maize cultivation and the fallow periods. Different apical letters indicate significant differences between groups (One way ANOVA, $P < 0.05$).

Monitoring period	CO_2 fluxes (g CO_2-C m^{-2} d^{-1})							
	CONV		COM1		COM2		GMAN	
	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Fallow 2007/2008								
Before MT	0.91^{a}	0.01	0.81^{a}	0.01	1.26^{b}	0.01	1.34^{b}	0.08
1st week after MT	1.7	0.6	0.68	0.08	1.13	0.09	1.43	0.19
1st month after MT	0.77^{a}	0.06	0.11^{b}	0.01	0.25^{c}	0.02	0.12^{b}	0.01
2nd month after MT	0.47^{a}	0.06	0.56^{b}	0.06	0.79^{b}	0.06	0.22^{a}	0.02
3rd month after MT	0.33^{a}	0.02	0.49^{b}	0.05	0.41^{b}	0.05	0.20^{c}	0.02
4th month after MT	0.51^{a}	0.05	0.78^{b}	0.06	0.78^{b}	0.10	0.25^{c}	0.02
7th month after MT	1.23	0.02	1.7	0.2	1.7	0.2	1.08	0.03
Maize 2008								
After VC-H-F-S	1.62^{a}	0.05	0.99^{b}	0.05	1.50^{a}	0.14	2.06^{a}	0.12
After 1st I	3.0^{a}	0.2	1.40^{b}	0.08	2.3^{a}	0.2	2.8^{a}	0.2
Before SE	2.11^{a}	0.14	1.3^{bc}	0.3	2.5^{ab}	0.3	2.0^{ab}	0.3
SE, before MW	4.44^{a}	0.14	2.42^{b}	0.12	4.07^{c}	0.09	3.28^{d}	0.11
SE, after MW	5.7^{a}	0.3	3.34^{b}	0.16	4.52^{b}	0.19	5.1^{a}	0.3
T + RS	4.47^{a}	0.13	3.41^{b}	0.09	4.67^{a}	0.11	4.22^{a}	0.06
M	3.13^{a}	0.08	2.24^{b}	0.06	2.61^{c}	0.07	2.61^{c}	0.06
Fallow 2008/2009								
After maize HO[1]	0.09^{a}	0.05	0.25^{b}	0.02	0.28^{b}	0.04	0.41^{c}	0.04
Before MT	0.74^{a}	0.10	0.78^{a}	0.05	1.27^{b}	0.01	1.36^{b}	0.19

[1] Data were log-transformed to meet statistical normality and homoscedasticity assumptions; untransformed data are presented for clarity.

Table 4
Mean value and standard error of the mean (SE) for soil daily N_2O emissions in CONV, COM1, COM2 and GMAN treatments along the different monitoring campaigns during the maize cultivation and the fallow periods. Different apical letters indicate significant differences between groups (One way ANOVA, $P < 0.05$).

Monitoring period	N_2O fluxes (μg N_2O-N m^{-2} d^{-1})							
	CONV		COM1		COM2		GMAN	
	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Fallow 2007/2008								
Before MT	277	43	328	75	359	11	642	36
1st week after MT	592	246	329	48	359	15	545	97
1st month after MT	331^{a}	19	9^{b}	6	36^{b}	11	30^{b}	10
2nd month after MT	221^{a}	38	467^{b}	46	398^{b}	47	60^{a}	17
3rd month after MT	69^{a}	18	208^{b}	32	135^{b}	28	47^{a}	12
4th month after MT	63^{a}	19	301^{b}	43	178^{c}	30	23^{a}	10
7th month after MT	114	25	152	31	91	33	66	24
Maize 2008								
After VC-H-F-S	373	52	94	19	235	52	235	60
After 1st I	1439^{a}	173	220^{b}	49	425^{b}	65	542^{b}	106
Before SE	770^{a}	107	169^{b}	10	302^{b}	66	508^{b}	50
SE, before MW	309^{a}	41	131^{b}	29	370^{ab}	84	219^{ab}	46
SE, after MW	488^{a}	80	113^{b}	39	114^{b}	31	54^{b}	22
T + RS	0.02	0.01	60	19	0.02	0.01	11	8
M	191	52	416	109	339	54	240	81
Fallow 2008/2009								
After maize HO[1]	36	13	67	22	0.28	0.15	6	4
Before MT	204^{a}	49	285^{a}	59	361^{b}	4	705^{b}	39

[1] Data were log-transformed to meet statistical normality and homoscedasticity assumptions; untransformed data are presented for clarity.

At the beginning of the maize period, all treatments, with the only exception of COM1, showed a generalized slightly increase of soil GHG fluxes immediately after soil N addition and incorporation at different forms, with a following fast peak emission after the first irrigation event (Fig. 1, Tables 3 and 4). In this reactive first phase, CONV showed significantly higher N_2O fluxes respect to the organic treatments (Table 4). Out of these, GMAN exhibited the highest N_2O-N evolution rate, followed by the COM treatments, with flux intensities proportional to the compost doses (COM2>COM1). The pattern of soil CO_2 emissions appeared similar, but the differences among treatments resulted less marked. The slowest progression of fluxes was recorded in COM1 (Fig. 1, Table 3).

After maize emergence, before the phase of stem extension, the trend of soil GHG emissions remained quite similar (Fig. 1, Tables 3 and 4). N_2O fluxes underwent an overall progressive reduction, with persisting higher rates in CONV (Fig. 1, Table 4).

At stem extension, a general increase of CO_2 emissions was observed and differences between CONV and COM treatments resulted amplified (Fig. 1, Table 3). Specifically, average emissions were lowered by about 40%, 20% and 10% in comparison to CONV, for COM1, COM2 and GMAN, respectively. As it relates to N_2O emissions, pulses were recorded in CONV after each irrigation event till the end of the stem extension. Nevertheless, after the mechanical weeding at stem extension, a peak at third day was recorded in COM1, with a rate of N release of about 400 μg N_2O-N m^{-2} day^{-1} (Fig. 1, Table 4).

When maize plant reached its full height, from tasseling throughout the reproductive growth stages, the differences of GHG emissions between treatments dampened, except for COM1 which continued to show the slowest CO_2 fluxes (Fig. 1, Table 3). At low soil mineral N availability, N_2O fluxes lowered to basal uniform values, with a tendency toward higher N evolution from COM1 and GMAN (Fig. 1, Table 4).

An overall decrease of CO_2 fluxes was observed during the physiological maturity, in proximity of harvest, which remained higher in CONV respect to the organic treatments, markedly respect to COM1 (Fig. 1, Table 3). At this stage, the reversion of N_2O pattern toward enhanced emissions in the organic trials became more evident, with averaged daily N_2O fluxes nearly doubled respect to CONV (Fig. 1, Table 4).

Immediately after harvesting, in warm dry conditions, soil GHGs fluxes resulted basal in all treatments, with a significant trend toward higher CO_2 emissions in the organic treatments, coupled with small N_2O pulses in COM1. Differently, after the first autumnal rains, faster GHG emission rates were recorded, markedly for GMAN and COM2.

The overall results showed a clear seasonal pattern for CO_2 fluxes (Fig. 1), with average emissions from all treatments in MP about 4 times higher than in FP, in tight connection with the trend of T_{soil} and WFPS (Table 5). Moreover, CO_2 emissions resulted significantly correlated: (i) to soil mineral N, along the fallow period and pre-maize germination and (ii) to crop evapotranspiration, after maize germination (Table 5). The seasonal pattern appeared less evident for N_2O fluxes (Fig. 1), which resulted interrelated with soil chemical-physical parameters (Table 5). During the MP, results highlighted N_2O emissions concentrated in very few days after the first irrigations following the fertilization event, at not limiting value of soil mineral N (>20 μg g^{-1}) and WFPS (>42%). In this regard soil water moisture, besides being driven by the irrigation water input, resulted negatively correlated with Etc after maize germination ($R = -0.47$; $P < 0.01$; Pearson Product- Moment Test). N_2O fluxes negatively correlated with Etc (Table 5).

3.3. Area and yield scaled cumulated GHG emissions

With regard to the total cumulated GHGs emissions, during the FP, no significant differences were detected among treatments, even if respect to CONV, higher ranges of GHG fluxes were recorded in plots under organic fertilization by compost (Fig. 2). The GHGs cumulated along the MP showed respect to CONV consistent lower fluxes of CO_2 in COM1 and N_2O in GMAN. On average, also the cumulated N losses by the COM treatments appeared restrained. (Fig. 2).

Total CO_2 emitted appeared comparable among treatments,

Table 5
Correlation coefficients for significant correlations between log-transformed daily GHG emissions (g CO_2-C m^{-2} d^{-1} and μg N_2O-N m^{-2} d^{-1}) and driving ancillary parameters. Pearson Product Moment Test: *$P < 0.05$; **$P < 0.01$; ***$P < 0.001$. The numbers reported in parentheses show the samples size. T_{air}: air temperature; T_{soil}: soil temperature; WFPS: water filled pore space; Min- N: mineral soil N, as sum of NO_3-N and NH_4-N.

	T_{soil} (daily mean)	WFPS	Mineral N	NO_3-N	NH_4-N	Etc
CO_2	0.55***(361)	0.320***(142)	0.57*** (158) (Fallow; Pre-maize germination)	0.46*** (158) (Fallow; Pre-maize germination)	0.43* (158) (Fallow; Pre-maize germination)	0.19** (260) (maize crop)
N_2O	0.32*(46) (WFPS> 42%; 20 μg g^{-1} <Min-N< 30 μg g^{-1})	0.36*** (104) (Min-N> 20 μg g^{-1})	0.61*** (98) (WFPS> 42%)	0.46*** (98) (WFPS> 42%)	0.43*** (98) (WFPS> 42%)	−0.46*** (256)

with the only exception of COM1 (about 18% lower) (Table 6). Cumulated N_2O fluxes resulted on average about 25% higher and 28% lower in COM1 and GMAN respectively, in comparison to CONV (Table 6). The resulting yearly EFs were comparable among CONV and COM1 and considerably higher than values produced by COM2 and GMAN. Respect to CONV, the final total GWP, resulted 24% higher in COM1 and 28% lower in GMAN (Table 6).

On a yield basis, cumulative yearly GHG in COM treatments nearly doubled respect to CONV. In this regard the yield-scaled GWP-100 confirmed the lower impact of GMAN, whilst evidenced impacts about 60% higher and almost doubled for COM2 and COM1, respectively (Table 6).

4. Discussion

4.1. Maize period (MP)

Soil GHGs dynamic after fertilizers incorporation evidenced

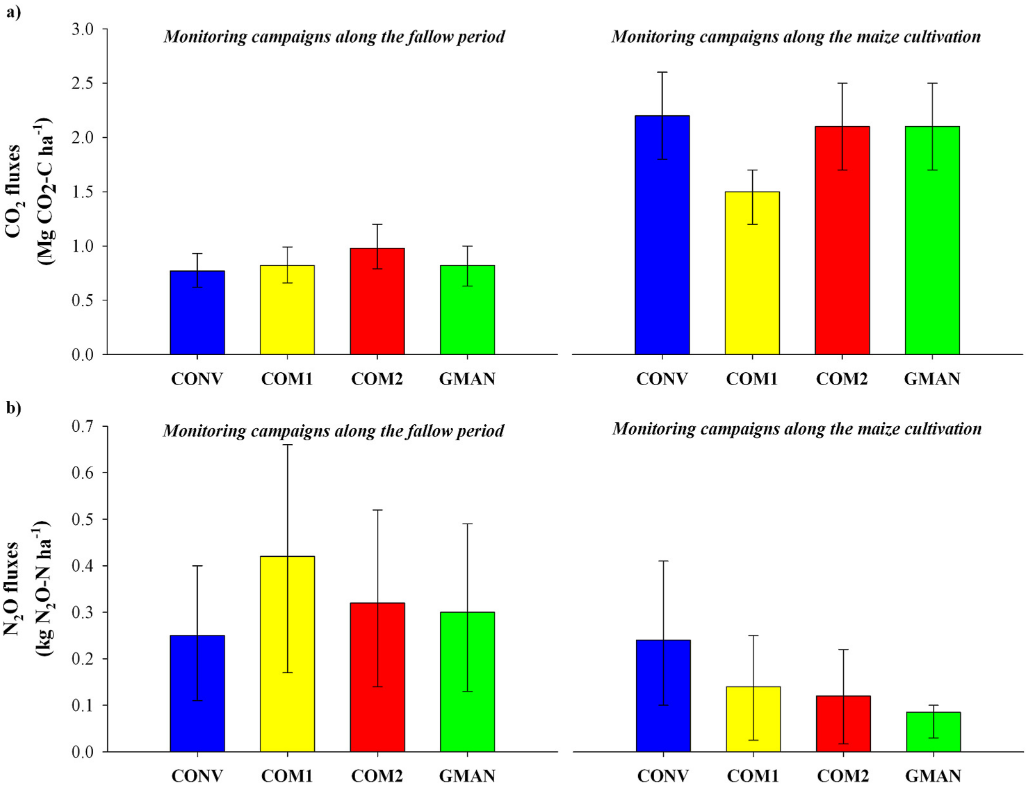

Fig. 2. Cumulative a) soil CO_2-C and b) N_2O-N emissions for each treatment along the monitored fallow and maize periods. The figures also shows the lower and upper bounds of the related CI95%.

Table 6
Summarized for each treatment: (i) the yearly area-scaled and yield-scale cumulated GHG fluxes, with overall EF% for N_2O emissions (in brackets); (ii) the total area-scaled and yield-scaled GWP.

Treatment	Yearly cumulated CO_2 emissions		Yearly cumulated N_2O emissions		Yearly cumulated GWP	
	Area scaled emissions (Mg CO_2-C ha^{-1} y^{-1})	Yield scaled emissions (Mg CO_2-C ton_{bd}^{-1})	Area scaled emissions (kg N_2O-N ha^{-1} y^{-1})	Yield scaled emissions (kg N_2O-N ton_{bd}^{-1})	Area scaled GWP (kg CO_2 eq ha^{-1} y^{-1})	Yield scaled GWP (kg CO_2 eq t_{db}^{-1})
CONV	5.5×10^0	3.7×10^{-1}	7.1×10^{-1} (0.35)	4.6×10^{-2} (0.55)	194	13
COM I	4.5×10^0	6.6×10^{-1}	8.9×10^{-1} (0.48)	1.3×10^{-1} (0.93)	240	35
COM II	5.8×10^0	6.2×10^{-1}	6.9×10^{-1} (0.16)	7.4×10^{-2} (0.26)	189	20
GMAN	5.2×10^0	4.1×10^{-1}	5.1×10^{-1} (0.19)	4.0×10^{-2} (0.39)	140	11
CONT			2.6×10^{-1}			

lower N_2O-N losses in the organic amended plots respect to CONV and appeared driven by the interacting effect of (i) different short term availability of soil nutrients in mineral and organic treatments and (ii) the low water input management.

The fast urea mineralization produced larger peaks of N_2O fluxes (Alluvione et al., 2010; Bertora et al., 2008) gathered in a first phase after fertilizer addition (Fig. 1, Table 4), until the maize roots began competing with nitrifiers and denitrifiers for mineral N.

Conversely, the lower N_2O fluxes from compost and green manure amended soil (Fig. 1, Table 4) resulted from the slower release of N from the applied organic forms (Aguilera et al., 2013; Alluvione et al., 2010; Rahn et al., 2003), with a rank of short-term daily fluxes reflecting the hierarchy of expected mineralization rates.

Mineralization of organic amendments depends on substrate composition and pedo-climatic conditions (Masunga et al., 2016; Rochette et al., 2007). For different organic materials (including plant residues and compost), short-term mineralization rate and linked GHG emissions appeared negatively correlated with the C:N ratio, since substrates with lower C:N decompose rapidly and promote NH_4^+ release and higher water-soluble organic C (Gomes et al., 2009; Huang et al., 2004; Masunga et al., 2016). The short-term daily fluxes from organic treatments (higher in GMAN than in compost amended plots) did not reflect the inverse dependence on the hierarchy of C:N ratios (16 and 10 for GMAN and COM treatments, respectively). Rather, they appeared driven by other key parameters modulating the mineralization rate of organic fertilizers such as fiber and lignin concentration, biodegradable fractions and labile N (Gomes et al., 2009; Griffin and Hutchinson, 2007; Masunga et al., 2016). Specifically, the higher soluble C/total C ratio and available inorganic N and the lower lignin content of vetch manure compared to compost (Alluvione et al., 2010) suggested that vetch tissue: (i) could have been mineralized faster than the stable organic matter of the compost (Alluvione et al., 2010) and (ii) might have provided additional readily available C and N for bacterial activity (Baruah et al., 2016; Gomes et al., 2016; Guardia et al., 2016; Huang et al., 2004), thereby leading to higher N_2O losses. Accordingly, the short-term differences of heterotrophic CO_2 fluxes (before maize germination) among the organic treatments highlighted the highest average fluxes for the substrate most prone to mineralization (green manure) and the lowest ones for the single-dose of compost amendment (Fig. 1, Table 3).

More than 90% of total nitrogen content in compost is in organic form with slow release by mineralization in soil (about 10%) in the first months after soil incorporation (Diacono and Montemurro, 2010; Masunga et al., 2016; Ryals et al., 2014). Differently, organic residues supplied by green manuring are biolabile, low in lignified tissues and easily decomposable (Alluvione et al., 2010; Masunga et al., 2016; Ryals et al., 2014).

Drip irrigation probably tuned the response of GHG fluxes to the replacement of urea with green manure and compost. It represents an increasingly popular water-saving technique to preserve agronomic yield and reduce GHG emission under Mediterranean maize cropped soils (Bozkurt et al., 2006; El-Hendawy and Schmidhalter, 2010; Forte et al., 2016). Indeed, the restrained volume of the soil wetting zone (Vázquez et al., 2005) can strongly modulate the relationship between soil bacterial activity, nutrient availability and temperature and restrict GHG emissions to small pulses in response to rewetting cycles (Forte et al., 2016; Kallenbach et al., 2010; Peterson et al., 2016; Sánchez-Martín et al., 2008, 2010a,b). The issue has received more attention with regard to the effect of drip irrigation on N_2O fluxes from Mediterranean crops under mineral fertilizer management, which highlighted reduced N-N_2O losses respect to the high water input management (Sánchez-Martín et al., 2008, 2010a; Aguilera et al., 2013; Kennedy et al., 2013). Conversely, scanty data are available on nutrient cycling from solid organic fertilizers and linked GHG emissions under surface drip irrigation schemes in Mediterranean agroecosystems (Aguilera et al., 2013; Peterson et al., 2016). However, throughout the late spring-summer period, the low water input by drip irrigation is not expected to radically change the constrained decomposition and N-mineralization rates observed in Mediterranean climate, where drought conditions limit microbial growth and activity and nutrients diffusion in the soil pore space (Criquet et al., 2004; Fioretto et al., 2005; Marion, 1982). Therefore, drip irrigation might have contributed to reduce short term GHG emissions from organic fertilizers, limiting the bacterial oxidation of the added organic matter to small pulses promoted by irrigation water and thus further delaying the release of available N into the soil respect to the mineral fertilizer (Peterson et al., 2016). In this regard, for similar crop phase and fertilizations management under temperate climate, Alluvione et al. (2010) recorded enhanced GHG fluxes from the soil fertilized by vetch green manure respect to the compost and urea amended soils, due to the optimal WFPS allowing the organic substrates to decompose according their specific composition in easily biodegradable and recalcitrant C compounds.

From maize emergence onward, the contribution of root respiration arose (Forte et al., 2016; Lee et al., 2009) and interacted with the effect of organic fertilization on heterotrophic soil respiration, leading to pattern of CO_2 emissions among treatments which reflected the hierarchy of maize production levels. The negative functional relationship detected for all treatments between soil N_2O fluxes and Etc (Table 5) proved a “crop effect” on soil N_2O-N losses as well. Indeed, Etc represents a process including the complexity of soil-plant system affecting production and emission of soil N_2O. High Etc values indicate an intense root activity, entailing soil water depletion via evapotranspiration and active roots N uptake, mainly surrounding the rhizosphere where microbial activity is higher.

4.2. Fallow period (FP)

After maize harvest, without root competition for N uptake, the pattern toward higher soil mineral N availability and GHGs emissions in the organic treatments after the first autumnal rains (Fig. 1, Tables 3 and 4) supported the hypothesis of a delayed mineralization of the supplied organic matter, amplified by the spring-summer drip irrigation management, able to promote residuals GHGs pulses in the autumn fallow period at not limiting values of soil temperature and WFPS (Aguilera et al., 2013; Mancinelli et al., 2010).

Respect to CONV, the GHG response of the organic fertilized plots to the fall deep ploughing appeared restrained (for GMAN) and almost negligible (for COM treatments), suggesting a protecting effect of the organic amendments against the fast SOM mineralization usually induced by the mouldboard plough (Bronick and Lal, 2005; Wright and Hons, 2005). Additionally, the amplitude of the short-term tillage-induced GHGs pulses inversely reflected the higher levels of SOM aggregation and stabilization detected in COM treatments in comparison to either GMAN and CONV (Spaccini and Piccolo, 2012).

For both compost doses, flux activities remained remarkable throughout the FP, with mean values higher respect to CONV and to the same treatments during the MP. This finding was likely ascribable to the slow mineralization rates of compost which resulted in higher N organic concentration at the end of each maize crop cycle respect to both GMAN and CONV (Table 2), and consequently might have given rise to a delayed mineralization of organic matter along each following intercrop period, also promoted by the main tillage. The average lower N_2O-N losses in COM2 respect to COM1 (by about 25%), might have been connected with the enhanced capability of the doubled dose to prevent organic N from mineralization, since the compost humified organic matter can promote the building of large hydrophobic domains preserving the biolabile soil C from degradation (Spaccini et al., 2002).

Contrary to Sanz-Cobena et al. (2014), but in line with Guardia et al. (2016), the vetch cover crop (and the linked autotrophic root respiration) did not affect soil CO_2 fluxes respect to the CONV bare soil (Fig. 1, Table 3). Additionally, this study supported the evidence of comparable N_2O emissions for legume cover crops and bare soils in Mediterranean semi-arid environment (Barton et al., 2011). Other studies for the Mediterranean context highlighted higher direct N_2O fluxes from vetch cover crop respect to fallow bare soil (Guardia et al., 2016; Sanz-Cobena et al., 2014). However in these latter cases, additional mineral N was supplied at maize sowing, which might have led to higher surplus N in the forthcoming fallow period respect to the vetch N uptake, resulting in higher N_2O-N losses (Guardia et al., 2016; Sanz-Cobena et al., 2014).

Respect to COM treatment, the potential residual slow mineralization of vetch tissues along the fallow period likely played a constrained role, as suggested by the comparable values of post-harvest organic N concentrations in GMAN and CONV (Table 2). Nonetheless, it might be assumed that eventual delayed mineralization of organic N might has been counteracted by the vetch uptake, retaining the potential post-harvest surplus inorganic N (Sanchez-Martin et al., 2010a,b).

4.3. Cumulative area and yield scaled GHGs emissions

At comparable CO_2 emissions, GMAN showed respect to CONV a significant reduction (by about 65%) of cumulative area-scaled N_2O fluxes during the MP, without significantly increasing N_2O-N losses along the FP (Fig. 2). This supported the evidence that green manure from legume cover crops may prevent N losses along the intercrop period and supply (after incorporation) a slow release of biologically fixed N for the forthcoming spring-summer cash crops (Cherr et al., 2006; Gabriel and Quemada, 2011).

In COM treatments area-scaled cumulated N_2O emissions resulted about 50% lower (as averaged value for both compost doses) respect to CONV (Fig. 2), in line with the range of N_2O-N losses reduction (by 55%–77%) reported for other Mediterranean maize cultivations fertilized by MSW compost as alternative to mineral fertilization (López-Fernández et al., 2007; Meijide et al., 2007). Differently, remarkable area-scaled cumulated GHGs were evidenced in the FP, respect to CONV and to the yielded emissions in the MP (Fig. 2), according to the significant N_2O-N losses detected during the winter fallow of other Mediterranean spring-summer cropping system under organic fertilization (Aguilera et al., 2013; Kallenbach et al., 2010; Lee et al., 2009; Sánchez-Martín et al., 2010b).

This findings confirmed that the complex dynamic of N mineralization, rather than N fertilization per se, regulates soil mineral N availability and cumulated GHG fluxes from agricultural soils (Masunga et al., 2016; Robertson et al., 2000). Apparently, results would also confirm for the organic treatments the inverse relationship between cumulated GHG emissions and C:N ratio (Baruah et al., 2016; Gomes et al., 2009; Huang et al., 2004). However, compared to GMAN, the higher cumulated GHG emissions in COM treatments were ascribable to the slower mineralization rates which did not to match plant uptake and resulted in higher residual GHG fluxes. In this regard the study confirmed only for GMAN the potential GHG mitigation recently highlighted for organic fertilization under low water input management in Mediterranean climate (Aguilera et al., 2013). Otherwise, the suboptimal soil moisture conditions for N_2O production during the days of fertilization resulted for all treatments in low cumulated N losses (Aguilera et al., 2013; Forte et al., 2016; Kennedy et al., 2013; Sánchez-Martín et al., 2008, 2010a,b) which supported the restrained EFs highlighted for Mediterranean spring-summer low-water input cultivations subject to either inorganic or organic fertilization (Table 6) (Aguilera et al., 2013; Castaldi et al., 2015).

According to Alluvione et al. (2009) and Grignani et al. (2012) organic treatments entailed an average increase of soil organic C concentration in the 0–30 cm layer respect to CONV (significantly higher for COM2 respect to CONV), corresponding to about 0.4, 1.3 and 0.8 Mg C ha^{-1} y^{-1} additionally stored in COM1, COM2 and GMAN soil organic matter, respectively. This additional soil C storage in the organic treatments might be accounted as corresponding CO_2-C saved emissions for GHGs inventory purposes. In this regard, the overall results inherent the total area scaled GWPs (Table 6) confirmed the potential GHG profile of COM2 and indicated that other additional 0.1 CO_2-C eq ha^{-1} y^{-1} might be saved by the GMAN management, thanks to the restrained biogenic N_2O-N fluxes in the MP. Otherwise, they warned about the counteracting losses of about 0.2 CO_2-C eq ha^{-1} y^{-1} in COM1, due to the highest fallow N_2O-N emissions.

A deeper understanding of the possible tradeoffs between GHG benefits and constraints of the organic fertilization requires to evaluate impacts per product unit (Aguilera et al., 2013; Meier et al., 2015; Tuomisto et al., 2012). The timing of soil mineral N released from vetch green manure mineralization roughly coincided with the pattern of crop uptake, supporting comparable crop productivity respect to the mineral management (Miguez and Bollero, 2005; Tonitto et al., 2006). Accordingly, the overall yield-scaled GHG results (Table 6) confirmed that legume cover crop as green manure for maize cultivation can provide an effective balance between agronomic efficiency and GHGs emissions (Guardia et al., 2016; Masunga et al., 2016).

Differently, the low N mineralization rate of compost, besides leading to higher SCS, also affected the low crop productivity and low N content in tissues (Alluvione et al., 2013; Diacono and Montemurro, 2010). Besides climatic conditions and irrigation management (Peterson et al., 2016), also the low soil OC content and the clay mediated SOC physical protection regulated the process, promoting low organic N mineralization and soil mineral N immobilization (Alluvione et al., 2013). Otherwise, under temperate climate and for different soil texture, the same compost management did not depress maize growth (Alluvione et al., 2013). As a result, respect to CONV, the yield-scaled GWP of COM treatments resulted 60% higher and nearly doubled for COM2 and COM1 respectively, according to the lower COM yields markedly dampening at halved compost dose (Table 6). This finding questioned the effective GHG gas benefits of the organic fertilization by compost in the specific investigated pedo-climatic conditions.

However, the response of maize productivity and soil GHG emissions to compost amendments needs to be verified in the longer term since the best compost agronomic performances have been often gained under high and frequent applications (Diacono and Montemurro, 2010). In this regard, the negative effect of compost on maize yield resulted the highest in the first year, whilst it lowered in the third year, following the repeated additions, markedly for COM2 (Alluvione et al., 2009, 2013). From a total life cycle perspective also the upstream emissions of the composting process should be considered, since potential relevant contributors to the overall GHG profile of alternative agronomic N management (Martínez-Blanco et al., 2009; Meier et al., 2015; Zucaro et al., 2015). Finally, the effect of replacing the bare fallow with a cover crop should be investigated, in order to beneficiate of the slow progressive N release from compost which could support winter productions and compensate the low maize productivity.

5. Conclusions

The results produced in this study showed for the maize crop period an essential effect of compost amendment and vetch green manure at reducing soil N_2O emissions in comparison to conventional mineral N management, without contextually increase CO_2 emissions. This particularly applied to the first stages of maize development, thanks to the slow release of nitrogen available for soil bacterial activity. In this regard drip irrigation and soil texture likely interacted with N fertilizer management, affecting the dynamic of organic substrates mineralization and consequent linked GHG fluxes by bacterial processes competing with the plant system for nutrient uptake.

Along the fallow period, respect to the conventional bare soil management, we can conclude: (i) the trend of GHGs emissions from the bare soil of compost amended treatments reversed toward higher N_2O-N and CO_2-C losses, likely due to the residual effect of the slow progressive mineralization of the supplied organic matter; whilst (ii) GHGs emissions from vetch cropped soil resulted comparable, likely also thanks to the retention of surplus residual N. The overall results in terms of total cumulated GHGs (as CO_2 eq) highlighted for the vetch-corn system lower area and yield-scaled GWP respect to the reference bare fallow-maize scenario under mineral fertilization. Otherwise they questioned the GHG mitigation potential of compost amendment under the specific investigated pedo-climatic conditions, due to: (i) enhanced emissions during the fallow period, counteracting the entailed GHG benefits along the maize period; (ii) lower maize yield performance relative to the reference mineral fertilization management, which markedly worsened the GHG profile on a yield basis. In this regard, there is the need to further investigate the longer term response of maize yield to continuous compost amendments and the potential benefits from the replacement of the bare fallow soil with a winter crop, in order to limit residual N_2O emissions and at the same time to take advantages of residual fertility, compensating the low maize productivity.

The markedly lower N_2O emission factors, compared with the default IPCC 1% value confirmed the need for local crop specific EF, representative of site pedo-climatic conditions.

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References

Aguilera, E., Lassaletta, L., Sanz-Cobena, A., Garniere, J., Vallejo, A., 2013. The potential of organic fertilizers and water management to reduce N_2O emissions in Mediterranean climate cropping systems. A review. Agr. Ecosyst. Environ. 164, 32–52.

Allen, R., Pereira, L., Raes, D., Smith, M., 1998. Crop Evapotranspiration: Guidelines for Computing Crop Water Requirements. FAO Irrigation and Drainage Paper no. 56, Rome.

Alluvione, F., Fiorentino, N., Fagnano, M., Bertora, C., Zavattaro, L., Grignani, C., 2009. Corn management: can compost help to substitute mineral fertilizers? In: Maggioli, S.p.A. (Ed.), Ecomondo 2009, pp. 353–358. Santarcangelo di Romagna, Rimini ISBN 8838759359.

Alluvione, F., Bertora, C., Zavattaro, L., Grignani, C., 2010. Nitrous oxide and carbon dioxide emissions following green manure and compost fertilization in corn. Soil Sci. Soc. Am. J. 74, 384–395.

Alluvione, F., Fiorentino, N., Bertora, C., Zavattaro, L., Fagnano, M., Quaglietta Chiarandà, F., Grignani, C., 2013. Short-term crop and soil response to C-friendly strategies in two contrasting environments. Eur. J. Agron. 45, 114–123.

Bain, W.G., Hutyra, L., Patterson, D.C., Bright, A.V., Daube, B.C., Munger, W.J., Xu, L., Furtaw, M.D., Madsen, R.A., Garcia, R.L., Anderson, D.J., McDermitt, D.K., 2006. On maintaining pressure equilibrium between soil CO_2 flux chamber and the ambient air. J. Geophys. Res. 111, 37–48.

Barton, L., Butterbach-Bahl, K., Kiese, R., Murphy, D.V., 2011. Nitrous oxide fluxes from a grain-legume crop (narrow-leafed lupin) grown in a semiarid climate. Glob. Change Biol. 17, 1153–1166.

Baruah, A., Bordoloi, N., Baruah, K.K., 2016. Effect of organic amendments with varied C-N ratios on grain productivity and nitrous oxide (N_2O) emission from wheat grown in alluvial soil. A. J. C. S. 10 (4), 460–469.

Bertora, C., Alluvione, F., Zavattaro, L., van Groenigen, J.W., Velthof, G., Grignani, C., 2008. Pig slurry treatment modifies slurry composition, N_2O, and CO_2 emissions after soil incorporation. Soil Biol. Biochem. 40, 1999–2006.

Bozkurt, Y., Yazar, A., Gençel, B., Sezen, M.S., 2006. Optimum lateral spacing for drip-irrigated corn in the Mediterranean Region of Turkey. Agr. Water Manag. 85, 113–120.

Bronick, C.J., Lal, R., 2005. Soil structure and management: a review. Geoderma 124, 3–22.

Büchi, L., Gebhard, C.A., Liebisch, F., Sinaj, S., Ramseier, H., Charles, R., 2015. Accumulation of biologically fixed nitrogen by legumes cultivated as cover crops in Switzerland. Plant Soil 393 (1), 163–175.

Castaldi, S., Aragosa, D., 2002. Factors influencing nitrification and denitrification variability in a natural and fire-disturbed Mediterranean shrubland. Biol. Fert. Soils 36, 418–425.

Castaldi, S., Alberti, G., Bertolini, T., Forte, A., Miglietta, F., Valentini, R., Fierro, A., 2015. N_2O Emission Factors for Italian Crops. In: Valentini, R., Miglietta, F. (Eds.), The Greenhouse Gas Balance of Italy. Environmental Science and Engineering, Springer-Verlag Berlin Heidelberg, pp. 135–144. http://dx.doi.org/10.1007/978-3-642-32424-6_9. Print ISBN: 978-3-642-32423-9.

Cherr, C.M., Scholberg, J.M.S., McSorley, R., 2006. Green manure approaches to crop production: a synthesis. Agron. J. 98, 302–319.

Criquet, S., Ferre, E., Farnet, A.M., Le Petit, J., 2004. Annual dynamics of phosphatase activities in an evergreen oak litter: influence of biotic and abiotic factors. Soil Biol. Biochem. 36, 1111–1118.

Diacono, M., Montemurro, F., 2010. Long-term effects of organic amendments on soil fertility. A review. Agron. Sustain. Dev. 30, 401–422.

Ding, W., Cai, Y., Cai, Z., 2007. Soil respiration under maize crops: effects of water, temperature, and nitrogen fertilization. Soil Sci. Soc. Am. J. 71, 944–951.

Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on waste and repealing certain. O. J. L. 312, 22.11.2008, pp. 3–30.

El-Hendawy, S.A., Schmidhalter, U., 2010. Optimal coupling combinations between irrigation frequency and rate for drip-irrigated maize grown on sandy soil. Agr. Water Manag. 97, 439–448.

Fagnano, M., Adamo, P., Zampella, M., Fiorentino, N., 2011. Environmental and agronomic impact of fertilization with composted organic fraction from municipal solid waste: a case study in the region of Naples, Italy. Agr. Ecosyst. Environ. 141, 100–107.

Fierro, A., Forte, A., 2012. Measurements of CO_2 and N_2O emissions in the agricultural field experiments of the MESCOSAGR project. In: Piccolo, A. (Ed.), Carbon Sequestration in Agricultural Soils. Springer-Verlag Berlin Heidelberg, pp. 229–259.

Fioretto, A., Di Nardo, C., Papa, S., Fuggi, A., 2005. Lignin and cellulose degradation and nitrogen dynamics during decomposition of three leaf litter species in a Mediterranean ecosystem. Soil Biol. Biochem. 37, 1083–1091.

Forte, A., Fiorentino, N., Fagnano, M., Fierro, A., 2016. Mitigation impact of minimum tillage on CO_2 and N_2O emissions from a Mediterranean maize cropped soil under low-water input management. Soil Till. Res. http://dx.doi.org/10.1016/j.still.2016.09.014. Available online 30 September 2016.

Franzluebbers, A.J., 2005. Soil organic C sequestration and agricultural greenhouse gas emissions in the southeastern USA. Soil Till. Res. 83 (1), 120–147.

Freibauer, A., Rounsevell, M.D.A., Smith, P., Verhagen, J., 2004. Carbon sequestration in the agricultural soils of Europe. Geoderma 122 (1), 1–23.

Gabriel, J.L., Quemada, M., 2011. Replacing bare fallow with cover crops in a maize cropping system: yield, N uptake and fertiliser fate. Eur. J. Agron. 34, 133–143.

Gomes, J., Bayer, C., de Souza Costa, F., de Cássia Piccolo, M., Zanatta, J.A., Vieira, F.C.B., Six, J., 2009. Soil nitrous oxide emissions in long-term cover crops-based rotations under subtropical climate. Soil Till. Res. 106, 36–44.

Gregorich, E.G., Rochette, P., Vanden Bygaart, A.J., Angers, D.A., 2005. Greenhouse gas contributions of agricultural soils and potential mitigation practices in Eastern Canada. Soil Till. Res. 83, 53–72.

Griffin, T.S., Hutchinson, M., 2007. Compost maturity effects on nitrogen and carbon mineralization and plant growth. Compost Sci. Util. 15 (4), 228–236.

Grignani, C., Zavattaro, L., Sacco, D., Monaco, S., 2007. Production, nitrogen and carbon balance of maize-based forage systems. Eur. J. Agron. 26, 442–453.

Grignani, C., Alluvione, F., Bertora, C., Zavattaro, L., Fagnano, M., Fiorentino, N., Mori, M., Amato, M., Lupo, F., Bochicchio, R., 2012. Field plots and crop yields under innovative methods of carbon sequestration in soil. In: Piccolo, A. (Ed.), Carbon Sequestration in Agricultural Soil (A Multidisciplinary Approach to Innovative Methods. Springer Heidelberg, Dordrecht, London, New York, pp. 39–60.

Guardia, G., Abalosb, D., García-Marco, S., Quemadaa, M., Alonso-Ayusoa, M., Cárdenas, L.M., Dixonc, L.E., Vallejo, A., 2016. Integrated soil fertility management drives the effect of cover crops on GHG emissions in an irrigated field. Biogeosciences 13, 5245–5257.

Hargreaves, G., Samani, Z., 1985. Reference crop evapotranspiration from temperature. Appl. Eng. Agric. 1, 96–99.

Heller, H., Bar-Tal, A., Tamir, G., Bloom, P., Venterea, R.T., Chen, D., Zhang, Y., Clapp, C.E., Fine, P., 2010. Effects of manure and cultivation on carbon dioxide and nitrous oxide emissions from a corn field under Mediterranean conditions. J. Environ. Qual. 39 (2), 437–448.

Huang, Y., Zou, J.W., Zheng, X.H., Wang, Y.S., Xu, X.K., 2004. Nitrous oxide emissions as influenced by amendment of plant residues with different C: N ratios. Soil Biol. Biochem. 36, 973–981.

IPCC, 2006. 2006 Intergovernamental panel of climate change guidelines for national greenhouse gas inventories. In: Eggleston, H.S., Buendia, L., Miwa, K., Ngara, T., Tanabe, K. (Eds.), Agriculture, Forestry and Other Land Use: N2O Emissions from Managed Soils, and CO2 Emissions from Lime and Urea Application. Institute for Global Environmental Strategies (IGES), Japan, pp. 1–54 (Chapter 11).

IPCC, 2007. Summary for policymakers. In: Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M., Miller, H.L. (Eds.), Climate Change 2007: Synthesis Report of Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom, p. 22.

Kallenbach, C.M., Rolston, D.E., Horwath, W.R., 2010. Cover cropping affects soil N_2O and CO_2 emissions differently depending on type of irrigation. Agr. Ecosyst. Environ. 137, 251–260.

Kennedy, T.L., Suddickb, E.C., Sixc, J., 2013. Reduced nitrous oxide emissions and increased yields in California tomato cropping systems under drip irrigation and fertigation. Agr. Ecosyst. Environ. 170, 16–27.

La Scala, N., Bolonhezi, D., Pereira, G.T., 2006. Short-term soil CO_2 emission after conventional and reduced tillage of a no-till sugar cane area in southern Brazil. Soil Till. Res. 91, 244–248.

Lal, R., 2004. Soil carbon sequestration impacts on global climate change and food security. Science 304, 1623–1627.

Lee, J., Hopmans, J.W., van Kessel, C., King, A.P., Evatt, K.J., Louie, D., Rolston, D.E., Six, J., 2009. Tillage and seasonal emissions of CO_2, N_2O and NO across a seed bed and at the field scale in a Mediterranean climate. Agr. Ecosyst. Environ. 129, 378–390.

Li, C.S., Frolking, S., Butterbach-Bahl, K., 2005. Carbon sequestration in arable soils is likely to increase nitrous oxide emissions, offsetting reductions in climate radiative forcing. Clim. Change 72, 321–338.

Li, L.-J., You, M.-Y., Shi, H.-A., Ding, X.-L., Qiao, Y.-F., Han, X.-Z., 2013. Soil CO_2 emissions from a cultivated Mollisol: effects of organic amendments, soil temperature, and moisture. Eur. J. Soil Biol. 55, 83–90.

López-Fernández, S., Díez, J.A., Hernaiz, P., Arce, A., García-Torres, L., Vallejo, A., 2007. Effects of fertiliser type and the presence or absence of plants on nitrous oxide emissions from irrigated soils. Nutr. Cycl. Agroecosys. 78, 279–289.

Lynch, D.H., Voroney, R.P., Warman, P.R., 2006. Use of 13 C and 15 N natural abundance techniques to characterize carbon and nitrogen dynamics in composting and in compost-amended soils. Soil Biol. Biochem. 38, 103–114.

Mancinelli, R., Campiglia, E., DiTizio, A., Marinari, S., 2010. Soil carbon dioxide emission and carbon content as affected by conventional and organic cropping systems in Mediterranean environment. Appl. Soil Ecol. 46, 64–72.

Mangalassery, S., Sjögersten, S., Sparkes, D.L., Sturrock, C.J., Craigon, J., Mooney, S.J., 2014. To what extent can tillage lead to a reduction in greenhouse gas emissions from temperate soils? Sci. Rep. 4 (4586), 1–8.

Mantovi, P., Baldoni, G., Toderi, G., 2005. Reuse of liquid, dewatered, and composted sewage sludge on agricultural land: effects of long-term application on soil and crop. Water Res. 39, 289–296.

Marion, G.M., 1982. Nutrient mineralization processes in Mediterranean-type ecosystems, 58, 313–320. USDA forest service general technical report PSW e United States, Pacific Southwest Forest and Range Experiment Station; June 1982.

Martínez-Blanco, J., Munoz, P., Antón, A., Rieradevall, J., 2009. Life cycle assessment of the use of compost from municipal organic waste for fertilization of tomato crops. Resour. Conserv. Recy. 53, 340–351.

Masunga, R.H., Uzokwe, V.N., Mlay, P.D., Odeh, I., Singh, A., Buchan, D., De Neve, S., 2016. Nitrogen mineralization dynamics of different valuable organic amendments commonly used in agriculture. Appl. Soil Ecol. 101, 185–193.

Meier, M.S., Stoessel, F., Jungbluth, N., Juraske, R., Schader, C., Stolze, M., 2015. Environmental impacts of organic and conventional agricultural products– Are the differences captured by life cycle assessment? J. Environ. Manag. 149, 193–208.

Meijide, A., Díez, J.A., Sánchez-Martín, L., López-Fernández, S., Vallejo, A., 2007. Nitrogen oxide emissions from an irrigated maize crop amended with treated pig slurries and composts in a Mediterranean climate. Agr. Ecosyst. Environ. 121, 383–394.

Melero, S., Madejón, E., Ruiz, J.C., Herencia, J.F., 2007. Chemical and biochemical properties of a clay soil under dryland agriculture system as affected by organic fertilization. Eur. J. Agron. 26, 327–334.

Miguez, F.E., Bollero, G.A., 2005. Review of corn yield response under winter cover cropping systems using meta-analytic methods. Crop Sci. 45, 2318–2329.

Natali, F., Dalla Marta, A., Orlando, F., Orlandini, S., 2009. Water use in italian agriculture: analysis of rainfall patterns, irrigation systems and water storage capacity of farm ponds. Inter. J. Am. 55 (3), 56–59.

Peterson, C.A., Soares, T., Torbert, E., Herrera, I., Scow, K.M., Gaudin, A.C.M., 2016. Drip irrigation effect on soil function, root systems and productivity in organic tomato and corn. In: Proceedings of the Organic Agriculture Research Symposium Pacific Grove, CA. January 20, 2016.

Piccolo, A., Spaccini, R., Nieder, R., Richter, J., 2004. Sequestration of biologically labile organic carbon in soils by humified organic matter. Clim. Change 67, 329–343.

Rahn, C.R., Bending, G.D., Turner, M.K., Lillywhite, R.D., 2003. Management of N mineralization from crop residues of high N content using amendment materials of varying quality. Soil Use Manag. 19, 193–200.

Ranucci, S., Bertolini, T., Vitale, L., Di Tommasi, P., Ottaiano, L., Oliva, M., Amato, U., Fierro, A., Magliulo, V., 2011. The influence of management and environmental variables on soil N_2O emissions in a crop system in Southern Italy. Plant Soil 343, 83–96.

Regione Campania, 2014. Disciplinari di produzione integrate. DGR n. 50 del 28/02/2014. http://www.agricoltura.regione.campania.it.

Robertson, G.P., Paul, E.A., Harwood, R.R., 2000. Greenhouse gases in intensive agriculture: contribution of individual gases to the radiative forcing of the atmosphere. Science 289, 1922–1925.

Rochette, P., Angers, M., Chantigny, H., Gagnon, B., Bertrand, N., 2007. N_2O fluxes in soils of contrasting textures fertilized with liquid and solid dairy cattle manures. Can. J. Soil Sci. 88, 175–187.

Ros, M., Klammer, S., Knapp, B., Aichberger, K., Insam, H., 2006. Long-term effects of compost amendment of soil on functional and structural diversity and microbial activity. Soil Use Manag. 22, 209–218.

Ryals, R., Kaiser, M., Tor, M.S., Berhe, A.A., Silver, W.L., 2014. Impacts of organic matter amendments on carbon and nitrogen dynamics in grassland soils. Soil Biol. Biochem. 68, 52–61.

Sánchez, L., Díaz, J.A., Vallejo, A., Cartagena, M.C., 2001. Denitrification losses from irrigated crops central Spain. Soil Biol. Biochem. 33, 1201–1209.

Sánchez-Martín, L., Arce, A., Benito, A., Garcia-Torres, L., Vallejo, A., 2008. Influence of drip and furrow irrigation systems on nitrogen oxide emissions from a horticultural crop. Soil Biol. Biochem. 40, 1698–1706.

Sánchez-Martín, L., Meijide, A., Garcia-Torres, L., Vallejo, A., 2010a. Combination of drip irrigation and organic fertilizer for mitigating emissions of nitrogen oxides in semiarid climate. Agr. Ecosyst. Environ. 137, 99–107.

Sánchez-Martín, L., Sanz-Cobena, A., Meijide, A., Quemada, M., Vallejo, A., 2010b. The importance of the fallow period for N2O and CH4 fluxes and nitrate leaching in a Mediterranean irrigated agroecosystem. Eur. J. Soil Sci. 61, 710–720.

Sanz-Cobena, A., García-Marco, S., Quemada, M., Gabriel, J.L., Almendros, P., Vallejo, A., 2014. Do cover crops enhance N_2O, CO_2 or CH_4 emissions from soil in Mediterranean arable systems? Sci. Total Environ. 466, 164–174.

Silva, M.T.B., Menduina, A.M., Seijo, Y.C., Viqueira, F.D.-F., 2007. Assessment of municipal solid waste compost quality using standardized methods before preparation of plant growth media. Waste Manage. Res. 25, 99–108.

Smith, P., 2004. Carbon sequestration in crop lands: the potential in Europe and the global context. Eur. J. Agron. 20, 229–236.

Soil Survey Staff, 2014. Keys to Soil Taxonomy, twelfth ed. USDA-Natural Resources Conservation Service, Washington, DC.

Spaccini, R., Piccolo, A., Conte, P., Haberhauer, G., Gerzabek, M.H., 2002. Increased soil organic C sequestration through hydrophobic protection by humic substances. Soil Biol. Biochem. 34, 1839–1851.

Spaccini, R., Piccolo, A., 2012. Carbon sequestration in soils by hydrophobic protection and in situ catalyzed photo-polymerization of soil organic matter (SOM). Chemical and physical-chemical aspects of SOM in field plots. In: Piccolo, A. (Ed.), Carbon Sequestration in Agricultural Soils. A Multidisciplinary Approach to Innovative Methods. Springer-Verlag, Berlin Heidelberg, pp. 61–105.

Su, Y.Z., 2007. Soil carbon and nitrogen sequestration following the conversion of cropland to alfalfa forage land in northwest China. Soil Till. Res. 92 (12), 181–189.

Tejada, M., Gonzalez, J.L., Garcia-Martinez, A.M., Parrado, J., 2008. Effects of different green manures on soil biological properties and maize yield. Bioresour. Technol. 99, 1758–1767.

Tonitto, C., David, M.B., Drinkwater, L.E., 2006. Replacing bare fallows with cover crops in fertilizer-intensive cropping systems: a meta-analysis of crop yield and N dynamics. Agr. Ecosyst. Environ. 112 (1), 58–72.

Tuomisto, H.L., Hodge, I.D., Riordan, P., Macdonald, D.W., 2012. Does organic farming reduce environmental impacts? A meta-analysis of European research. J. Environ. Manag. 112, 309–320.

Triberti, L., Nastri, A., Giordani, G., Comellini, F., Baldoni, G., Toderi, G., 2008. Can mineral and organic fertilization help sequestrate carbon dioxide in cropland? Eur. J. Agron. 29, 13–20.

Vallejo, A., Diez, J.A., Lopez-Valdivia, L.M., Cartagena, M.C., Tarquis, A., Hernàiz, P., 2004. Dentitrification from an irrigated soil fertilized with pig slurry under Mediterranean conditions. Biol. Fert. Soils 40, 93–100.

Vázquez, N., Pardo, A., Suso, M.L., Quemada, M., 2005. A methodology for measuring drainage and nitrate leaching in unevenly irrigated vegetable crops. Plant Soil 269, 297–308.

Venterea, R.T., Maharjan, B., Dolan, M.S., 2011. Fertilizer source and tillage eff ects on yield-scaled nitrous oxide emissions in a corn cropping system. J. Environ. Qual. 40, 1521–1531.

Wright, A.L., Hons, F.M., 2005. Soil carbon and nitrogen storage in aggregates from different tillage and crop regimes. Soil Sci. Soc. Am. J. 69, 141–147.

Zucaro, A., Forte, A., Fagnano, M., Fierro, A., 2015. Life cycle assessment of maize cropping under different fertilization alternatives. I. J. P. E 10 (4), 427–436.