Nitrous oxide and carbon dioxide emissions following green manure and compost fertilization in corn

Mitigating the impact of fertilization on global warming: can leguminous green manure and compost help to reduce N_2O and CO_2 emissions?

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ABSTRACT

Alternative nitrogen fertilizers that stimulate low greenhouse gas emissions from soil are needed to reduce the impact of agriculture on global warming. Corn (*Zea mais*, L.) grown in a calcareous silt loam soil in northwestern Italy was fertilized with a municipal solid waste compost and vetch green manure (*Vicia villosa*, Roth.). Their potential to reduce N_2O and CO_2 emissions was compared to that of urea (130 kg N ha^{-1}). Gaseous fluxes were measured for two years in the spring (after soil incorporation of fertilizers) and in summer. In spring, the slow mineralization of compost reduced N_2O emissions (0.11 % of supplied N) relative to urea (3.4 % of applied N), without an increase in CO_2 fluxes. Nitrous oxide (2.31 % of fixed N) and CO_2 emissions from rapid vetch decomposition did not differ from urea. When N_2O and CO_2 fluxes were combined, compost reduced by 49% the CO_2 equivalent emitted following urea application. Vetch did not show such an effect. In summer, no fertilizer effect was found on N_2O and CO_2 emissions. Compost proved to be potentially suitable to reduce CO_2 equivalent emitted after soil incorporation while vetch did not. For a thorough evaluation, net greenhouse gas emissions assessment should be extended to the entire N life cycle. Differences between calculated N_2O emission factors and the default Tier 1 IPCC value (1%) confirmed the need for site- and fertilizer-specific estimations.

Abbreviations: ΔCO_2eq, increase of emitted CO_2 equivalent due to the applied fertilizer N respect to absence of fertilization; COM, fertilization with compost; CK, check plots not fertilized with N; EF, nitrous oxide emission factor of fertilizer N applied; GHG, greenhouse gas; LGM, fertilization with a leguminous green manure (hairy vetch); Ndfa, N derived from the atmosphere and fixed by a legume; PD, potential denitrification; PMN, N potentially mineralizable in anaerobiosis; SOC, soil organic carbon; SON, soil organic nitrogen; UR, fertilization with urea; WFPS, water-filled pore space.

INTRODUCTION

Optimal nitrogen (N) fertilization management is crucial not only to crop production improvement and cropping systems energy efficiency (Fluck, 1992; Sauerbeck, 2001; Mosier, 2002), but also to ammonia and greenhouse gas (GHG) emission reduction (Freney, 1997; Bussink and Oenema, 1998; Follett et al., 2005; Snyder et al., 2009). Greenhouse gases are produced during mineral N fertilizer synthesis by burning fossil fuels (Jenssen and Kongshaug, 2003) and following their application in the field. After N fertilizer distribution, nitrous oxide (N_2O) and nitrogen oxide emissions typically increase due to enhanced nitrification and denitrification processes (Byrnes, 1990; Mosier and Schimel, 1991; Stehfest and Bouwman, 2006). On the other hand, adequate fertilization can contribute to soil carbon (C) sequestration through a greater crop residue soil return (Halvorson et al., 1999; Follett, 2001; Alvarez, 2005). Similarly, green manures and organic fertilizers (such as animal manure, slurry, and compost) increase soil C storage and sustain crop nutrition with recycled and/or biologically (not industrially) fixed N (Campbell et al., 2001; Grignani et al., 2007; Melero et al., 2007; Triberti et al., 2008).

Different N sources have already been evaluated for their crop production sustainability while reducing their global warming impact. Among them, leguminous green manure and compost from urban waste represent viable options as mineral N fertilizer substitutes (Bøckman, 1997; Eriksen et al., 1999; N'Dayegamiye and Tran, 2001; Tejada and Gonzalez, 2003; Tejada et al., 2008). In addition to microbial activity stimulation, soil fertility increase, pest control, and reduced soil erosion, leguminous green manures can prevent nutrient leaching during the non-crop growing season and supply low energy cost N to the subsequent crop (Crews and Peoples, 2004; Cherr et al., 2006a). Compost fertilization not only has a low groundwater N pollution risk (Eva Erhart, 2007), if applied at a rate that meets crop needs (Hadas et al., 2004), but also great nutritive element recycling (Ikumo, 2005; Hargreaves et al., 2008), soil C sequestering and protection (Spaccini et al., 2002; Piccolo et al., 2004; Lynch et al., 2006) potentials .

71 Organic N sources used to sustain crop nutrition can reduce N_2O and CO_2 emissions after
72 soil incorporation depending on their N and C availability to soil microorganisms. (Granli and
73 Bøckman, 1994; Baggs et al., 2000; Huang et al., 2004). Many indicators can be used to estimate
74 such availability. Among all, soil mineral N, potentially mineralizable N in anaerobios (PMN), and
75 potential denitrification (PD) are common. Soil mineral N is an indicator of the N readily available
76 to nitrification and denitrification (Granli and Bøckman, 1994; Bertora et al., 2008). PMN and PD
77 are used, respectively, as indicators of organic N (Curtin and McCallum, 2004; Russell et al., 2006;
78 Monaco et al., 2008) and C fractions (Bijay et al., 1988; Hill and Cardaci, 2004; Ullah and
79 Faulkner, 2006; Bertora et al., 2007) that can be mineralized easily by soil microorganisms.
80 Moreover, PMN and PD can be used to detect changes in soil organic N (SON) and SOC evolution
81 under differing treatments shortly after their establishment, before equilibrium is attained, as the
82 more labile C and N fractions are known to be more responsive (Campbell et al., 2001).
83 The magnitude of the emission rates and their relative importance among different
84 management practices can vary between soils with different pH. Gaseous emissions from acidic
85 soils have been widely assessed, but less information is available for calcareous soils (Stehfest and
86 Bouwman, 2006). Soil environment, pH in particular, can be crucial determinants of N_2O emissions
87 (Granli and Bøckman, 1994; Clough et al., 2004) and soil organic matter mineralization (Curtin et
88 al., 1998; Bertrand et al., 2007). The effect of compost application on gaseous emissions from
89 calcareous soil has already been evaluated in Mediterranean climate (Vallejo et al., 2006; López-
90 Fernández et al., 2007), but few data are available from temperate climates.
91 Nitrous oxide and CO_2 emissions from soil can potentially occur throughout the year. In
92 temperate climates, emissions are usually concentrated in spring and in summer. In spring, high
93 gaseous emissions occur due to fertilization combined with tillage, in presence of elevated soil
94 moisture following seasonal precipitation. In summer, gaseous fluxes are usually stimulated by
95 warmer temperatures and dry/rewetting cycles following irrigation or rainstorms (Velthof et al.,
96 2002; Jabro et al., 2008; Kavdir et al., 2008; Lee et al., 2009). In this period, plant presence and root

development strongly influence soil environmental conditions and microbial activity (Smith and Tiedje, 1979; Cheng et al., 2003). Though not exhaustive, N_2O and CO_2 emissions during spring and summer can be high and contribute to an important share of total year emissions. Carbon dioxide fluxes do not represent the treatment net effect on the C balance between soil and atmosphere, which corresponds to soil organic C (SOC) evolution over time (IPCC, 2007b), but CO_2 emissions contribute to and can help explain SOC changes (Alluvione et al., 2009).

The objective of this research was to give a preliminary evaluation of N_2O and CO_2 emissions from soil following corn fertilization with compost or leguminous green manure. We tested, using a calcareous soil in temperate climate, the hypothesis that fertilization of corn with compost or leguminous green manure, as alternatives to urea, could reduce the stimulation of CO_2 equivalents emitted as N_2O and CO_2 after fertilizer incorporation, by influencing its N and C cycles. We expected N_2O fluxes to be reduced by supplying organic N not readily available to soil microorganisms, as it must be mineralized prior to its availability for the nitrification and denitrification processes. We expected increased N_2O and CO_2 emissions with rising mineral N, PMN and PD values. Moreover, we expected PMN and PD to detect increases in the SON and SOC following different fertilization managements.

MATERIALS AND METHODS

Site description and experimental treatments

The field experiment was carried out at the agro-environment experimental station of the Department of Agronomy, Forest and Land Management of the University of Turin (44° 53’ 1’’ N; 7° 41’ 10’’ E; 232 m a.s.l.). The site is located in the western area of the Po plain (northwestern Italy) and is characterized by a silt-loam alluvial soil that is scarcely weathered (Typic Udifluvent). Selected chemical and physical properties of the plowed layer (0- to 30- cm) sampled in 2006 are reported in Table 1. The climate is temperate sub-continental, with a mean annual temperature of

11.9 °C and a mean annual precipitation of 734 mm, characterized by two main rainfall periods in spring (April-May) and autumn (September–November).

Our study considered the following treatments: 1) compost mixture of park, garden, and urban waste (COM), 2) winter leguminous green manure (LGM) of hairy vetch, 3) urea at a rate of 130 kg N ha^{-1} (UR), and 4) a non-fertilized check plot (CK). Fertilized treatments were designed to receive amounts of fertilizer N similar to urea. For compost the total N was considered as fertilizer N while for vetch only the fraction derived from biological N fixation was counted. Treated 48 m^2 plots were established in 2006 and organized in a completely randomized design with three replications. Silage corn was grown because it is a main regional crop. Table 2 reports key compost and vetch composition parameters. In the compost, we determined total C by wet oxidation with dichromate and total N by the Kjeldhal method. In the vetch, we analyzed tissue C and N content with a CHN elemental analyzer (NA 1500 Nitrogen Analyzer from Carlo Erba Instruments – Thermo Fisher Scientific Inc., Waltham, MA). We analyzed fiber fractions (neutral detergent fiber, NDF; acid detergent fiber, ADF; acid detergent lignin, ADL) according to the Robertson and Van Soest (1981) method. Soluble C was computed by the difference between total C and C in the NDF fraction (Thuriès et al., 2002; Gabrielle et al., 2004). Ash content was determined by loss on ignition at 550°C.

The hairy vetch was shredded with a stalk shredder a few days before soil incorporation (17 May 2007 and 13 May 2008). Next, equal amounts of triple super-phosphate (100 kg P_2O_5 ha^{-1}) and KCl (200 kg K_2O ha^{-1}) were applied to all plots. Then, a moldboard plow was used to till all plots (30 cm depth) and to incorporate the applied fertilizers. Corn was planted on the same date plots were fertilized and tilled (4 June 2007 and 19 May 2008). Post-emergence herbicides were used in all treatments, resulting in the plots being weed-free. A travelling gun sprinkler irrigation system applied 40 mm of irrigation water once during each growing season (in 2007: 31 July, in 2008: 28 August), following the common farmers' practice in the region. The low N content of the irrigation water corresponded to a 4 kg N ha^{-1} addition for each irrigation event.

148 Each year, we quantified the epigeic biomass of the vetch before its incorporation. Carbon
149 and N in below-ground residue were counted as well (Unkovich and Pate, 2000; Rochette and
150 Janzen, 2005). As we did not measure the biomass of root vetch, we estimated its C and N content.
151 We considered a shoot C: root C ratio of 2.1 to be reasonable (Michalk and Mueller, 2003; Hamer
152 et al., 2008) even though it has been shown to change during vetch development (Gregory et al.,
153 1995). The same ratio was used to estimate vetch root N (Unkovich and Pate, 2000; IPCC, 2007b).
154 The percentage of total N in the epigeic biomass derived from the atmosphere (Ndfa) has been
155 measured at 58% in like environment experimental plots (Zavattaro et al., 2003). We used the same
156 percentage for N in root biomass.
157
158 ***Gaseous fluxes***
159 Nitrous oxide and CO_2 emissions were monitored in 2007 and 2008. Each year, we
160 concentrated flux measurements in two periods of the corn growing season when we were expecting
161 high gaseous fluxes: one in spring and one in summer. Spring measurements took place in 2007 for
162 29 days, (from 1 June to 28 June), and in 2008 for 31 days (from 16 May to 16 June). Summer
163 campaigns took place in 2007 for seven days (from 30 July to 6 August), and in 2008 for 12 days
164 (from 16 July to 24 July). In spring, fluxes were monitored a few days before fertilization, and daily
165 measurements were performed in the first week to follow. In subsequent weeks, frequency
166 decreased progressively while gaseous emissions approached the basal flux (absent peaks from
167 tillage, precipitation or irrigation). In summer, about two months after fertilization, fluxes were
168 monitored daily during the first week and three times during the second week. In 2008, no irrigation
169 occurred during the summer campaign because of high late spring precipitation.
170 We used a non-steady state closed chamber technique (Livingston and Hutchinson, 1995) to
171 measure emissions. In each plot, three stainless-steel anchors, inserted 5 cm into the soil, were set in
172 the inter-row of three non-adjacent rows a week before pre-fertilization measurements started. They
173 were removed for tillage and re-inserted immediately after seeding where they remained until the

174 growing season ended. Every sampling day, a cylindrical PVC chamber (i.d.: 240 mm; height: 110
175 mm; wall thickness: 6.2 mm) was sealed to each anchor with a rubber O-ring. Internal chamber gas
176 concentrations were measured using an Innova 412 photo-acoustic infrared gas analyzer
177 (LumaSense Technologies A/S, Ballerup, Denmark) re-circulating the gas sample through two
178 Teflon tubes (each 12.5 m long) attached to two sampling ports installed atop the chamber.
179 Consistent with the findings of Flechard et al. (2005), we found the instrument to over-estimate
180 N_2O concentration due to high CO_2 and water vapor concentrations during chamber deployment
181 (data not shown). To solve this problem, N_2O and CO_2 were measured separately and N_2O was
182 measured by forcing the gas sample pumped from chamber to instrument through a soda lime filter
183 to eliminate almost all CO_2 (concentration below 40 $\mu l\ l^{-1}$). In order to equilibrate the water vapor
184 concentration identically across all N_2O samples, they were then bubbled in 400 ml of de-ionized
185 water maintained at 18°C . Nitrous oxide dissolution in water did not influence flux estimations
186 since we calculated the N_2O dissolved to be in the range of six orders of magnitude lower than that
187 in the chamber headspace. The absence of effect was verified with a lab test (data not shown). Gas
188 concentrations were adjusted to account for the mixing of the chamber headspace with the previous
189 sample still present in the internal volume of the photo-acoustic infrared gas analyzer plus the
190 sampling tubes (Bertora et al., 2007). Volumetric concentrations of N_2O and CO_2 measured by the
191 instrument were transformed to mass values using the ideal gas law and the recorded air
192 temperature (HOBO H08-031-08 from Onset Computer Corporation, Bourne, MA) and pressure
193 (measured by the Innova 1412) inside the chamber.
194 Fluxes of CO_2 and N_2O were estimated from using two gas concentration measurements
195 inside the chamber headspace—at the start and end of closing—and assuming a linear change in gas
196 concentration over time. This assumption was frequently checked for each treatment by continuous
197 measurement of the CO_2 and N_2O concentration increases for a period of about 40 min.
198 Consequently, closure time varied according to flux intensities from about 15 min during emission
199 peaks to about 30 min during low emission periods.

200

201 ***Soil temperature and moisture***

202 Soil temperature (7.5 cm depth) was monitored in one LGM plot during the entire corn
203 growing season using a temperature probe (HOBO H08-031-08 from Onset Computer Corporation,
204 Bourne, MA). Soil samples from 0-7.5 cm depth were collected on each gas sampling day using a
205 soil corer (OakField Apparatus Inc., Oakfield, WI), and gravimetric water content was determined
206 after drying at 105°C for 24 hours. Water-filled pore space (WFPS) (Linn and Doran, 1984a) was
207 calculated using the soil bulk density (Blake and Hartge, 1986) measured at 0- to 7.5- cm in fall
208 2008 after corn harvest and a particle density of 2.79 Mg m^{-3} (Zavattaro and Grignani, 2001).

209

210 ***Soil N and C availability indicators***

211 Soil mineral N content, PMN, and PD were measured in each plot during gaseous flux
212 monitoring days: five times in spring (one day before fertilization, the day following fertilization
213 and than weekly until the end of the measurement campaign) and two times in summer (at the
214 beginning and end of the measurement campaign). Measurements were concentrated in the 0- to
215 7.5- cm layer as a previous field experiment (Monaco et al., 2009) showed that the soil surface can
216 be used as a proxy for the whole plowed layer. We confirmed the homogeneity of the soil mineral N
217 content throughout the 0- to 15- cm layer by monitorings conducted during the two 2007 campaigns
218 (data not shown).

219 Soil mineral N was extracted by shaking 16 g of moist soil with 80 ml of 1 M KCl solution
220 for 1 h. Subsequently, samples were filtered through a Whatman No. 1 paper, then extracts were
221 frozen until they were analyzed for NO_3^--N and NH_4^+-N concentration by colorimetry with a
222 continuous flow analyzer (Evolution II, Alliance Analytical Inc., Menlo Park, CA).

223 Potentially mineralizable N in anaerobiosis was measured according to the method reported
224 by Monaco et al. (2008). Sixteen g of soil and 40 g of water were mixed in a gas tight syringe
225 equipped with a vacuum proof stopcock. Air was removed by adjusting the height of the piston and

226 samples were then incubated at 40 °C for seven days. At incubation start and end, inorganic N was
227 extracted with a final 1M KCl solution. Inorganic N was measured as described above.
228 Potential denitrification was measured following the method reported by Bertora et al.
229 (2008) in which undisturbed soil cores are incubated (at 25 °C) in water-saturated, oxygen-free, and
230 non-nitrate-limiting conditions in the presence of acetylene. The soil cores were sampled by
231 inserting a steel cylinder (i.d.: 5.8 cm) 7.5 cm into the soil (on average 257 g of dry soil were
232 sampled). Approximately 200 mg N kg^{-1} dry soil were applied using 40 ml of KNO_3 solution
233 distributed over the soil cores stored in 500 ml glass jars to avoid nitrate limitation to denitrification
234 processes (Limmer and Steele, 1982). Anaerobic conditions were created by replacing the jar
235 headspace with N_2 two times after air removal (final absolute pressure of 20 kPa). In order to inhibit
236 the transformation of N_2O to N_2 and to prevent nitrification, 10% of the headspace was replaced
237 with acetylene (Yoshinari et al., 1977; Davidson et al., 1986; Tiedje et al., 1989). After 24 h, and
238 again at 48 h, N_2O was sampled from the headspace (25 ml gas in 10 ml evacuated vial) and
239 quantified using a Trace GC Ultra gas-chromatograph (Thermo Fisher Scientific Inc., Waltham,
240 MA) equipped with a thermal conductivity detector. Denitrification rates were calculated assuming
241 a linear increase in N_2O concentration (Beek et al., 2004).
242
243 ***Percent N lost and CO_2 equivalent emitted***
244 In order to accurately compare N_2O fluxes from various fertilizer treatments regardless of
245 the amount of N added, cumulative N_2O emissions, net of the N_2O emitted from CK, were used to
246 calculate a N_2O emission factor of applied N (EF) (Table 4). Two EFs were calculated for vetch:
247 one accounting for only fixed N, as it represents the fertilizer N added to the soil (Kelner et al.,
248 1997; Drinkwater et al., 1998), and the other taking into account the full amount of N in the legume
249 biomass (IPCC, 2007b).
250 Cumulative N_2O and CO_2 emissions from fertilized treatments were used to calculate the
251 CO_2 equivalents emitted (Robertson et al., 2000) after discounting emissions from CK to estimate

252 the increase in emissions due to fertilization (ΔCO_2eq). We assumed that the increase in N_2O and
253 CO_2 emissions between fertilized treatments and CK was due only to the N addition. Carbon
254 dioxide equivalents of N_2O were calculated by multiplying the amount of N_2O for its global
255 warming potential (298 kg CO_2 equivalent per kg N_2O) as suggested by the IPCC (2007a). As
256 different amounts of N were distributed (Table 2), CO_2 equivalents were expressed on one kg of
257 applied fertilizer N basis (for LGM, the amount of N fixed was used).
258
259 ***Data analysis***
260 Cumulative CO_2 and N_2O emissions were calculated using a linear interpolation across
261 sampling days. Since the campaigns were slightly longer in 2008 than in 2007, values were
262 calculated on Day 24 after spring fertilization and Day 6 after the start of the summer campaign in
263 both years. The variability among same-plot chambers was found to be significantly lower than the
264 variability among same-treatment plots (data not shown). Therefore, triplicate gaseous flux
265 measurements and the respective CO_2 equivalent and N_2O EF inside each plot were averaged before
266 subsequent analysis.
267 Data normality was verified using the Kolmogorov–Smirnov test. Since N_2O fluxes were
268 log-distributed, the geometrical average of daily values and seasonal cumulates was calculated for
269 same-plot chambers. Daily and cumulative N_2O fluxes, average soil NO_3^--N content, and PD were
270 log-transformed (natural logarithm) before the ANOVA and the correlation analyses.
271 Analysis of variance was used to determine differences in cumulative CO_2 and N_2O fluxes,
272 cumulative ΔCO_2eq, average soil NO_3^--N content, PMN, and PD by treatment and year. Seasons
273 were analyzed separately as different phenomena were observed. When significant, means were
274 separated using a Sidak post-hoc test. The relationship between (i) daily CO_2 and N_2O fluxes, soil
275 temperature, and WFPS, and (ii) cumulative CO_2 and N_2O emissions, average soil NO_3^--N content,
276 PMN, and PD were analyzed using the Pearson bivariate correlation. Only data from the two spring

277 campaigns were used. All statistical comparisons were made at the $\alpha = 0.05$ probability.
278 Calculations were made using SPSS 16.0 (SPSS Inc., Chicago, IL).
279
280 **RESULTS**
281 ***Composition parameters of compost and vetch and amounts of added C and N***
282 Properties of applied compost and vetch, and amounts of N and C added to the soil varied
283 between years (Table 2). Utilized compost, produced in 2007, was deemed stable and fully mature
284 given its C:N ratio of less than 20 (Chefetz et al., 1996; Brinton, 2000; Silva et al., 2007).
285 Nonetheless, it continued to mature beyond the 2007 fertilization, yielding more stable compost for
286 the 2008 application. Consistent with Chefetz et al. (1996) results describing normal compost
287 evolution, organic matter, C:N ratio, and soluble C:total C ratio decreased while ash content and the
288 recalcitrant ADL fraction increased. Vetch was harvested at mid-flowering both years. In 2008,
289 however, soluble C:total C ratio was higher and ADL was lower. The colder and wetter climate in
290 the second year probably contributed to these differences, even though the vetch was harvested on
291 nearly the same date. Comparison of the two fertilizers, across both years, showed vetch to be
292 characterized by a higher C:N ratio and soluble C:total C ratio, with lower ADL and ash contents
293 than compost.
294 Based on the amount of compost and vetch biomass incorporated, the total C supplied to the
295 soil was similar between the 2007 treatments while it was 50 % higher in LGM in 2008 (Table 2).
296 The difference was even greater for soluble C (vetch to compost ratio was 2.3). While the amount
297 of total N supplied with compost was comparable to that given with urea (130 kg N ha^{-1}) in both
298 years, the amount of N supplied with vetch was higher (both the total N and the Ndfa).
299
300 ***Gaseous emissions***
301 Nitrous oxide emissions in spring were enhanced by fertilization and tillage; fluxes followed
302 a parabolic pattern (Fig. 1a) (Baggs et al., 2000). Fluxes increased within a day after fertilizer

303 incorporation while emissions peaked after precipitation events. In summer, fluxes were nearly
304 absent, even after the 2007 irrigation.
305 Applied N sources affected N_2O emissions in spring, but not in summer. In spring,
306 cumulative emissions, on average, were almost six times higher in UR and LGM than in COM and
307 CK. A shorter duration of the N_2O peak coupled with higher values resulted in no significant
308 difference in cumulative emissions between the springs of 2007 and 2008 (Table 3). In summer, no
309 treatment effect was found, and fluxes were lower in 2007 than in 2008. Nitrous oxide emission
310 factors of applied N calculated from spring emissions were on average 26 times higher in UR and
311 LGM than in COM. When EF was calculated considering the entire amount of N distributed with
312 the vetch in LGM (EF used by the IPCC (IPCC, 2007b)) rather than just the N fixed by the crop, the
313 EF of LGM did not vary from COM and was significantly lower than UR (Table 4).
314 Across all treatments, spring CO_2 emissions were enhanced by fertilization and tillage, with
315 fluxes following parabolic patterns similar to N_2O (Fig. 1b) (Chatskikh and Olesen, 2007;
316 Chatskikh et al., 2008). Fluxes increased immediately after fertilization and tillage, and peaked two
317 weeks later. Emissions retreated back to base level prior to tillage at the end of the third or fourth
318 week depending on the year. In 2007, CO_2 emissions increased after the summer irrigation.
319 Cumulative CO_2 emissions varied among treatments in spring, but not in summer. In spring,
320 cumulative fluxes rose significantly from CK, to UR, to COM, and to LGM (Table 3). A longer
321 duration of the CO_2 emission peak coupled with lower values, resulted in no significant difference
322 in cumulative emissions between the springs of 2007 and 2008. In summer no difference was found
323 among treatments; emissions were higher in 2007 than 2008.
324 Daily and cumulative N_2O fluxes were positively correlated with CO_2 emissions (daily
325 values: $R=0.65$, $P<0.01$; cumulative values: $R=0.56$, $P<0.01$).
326 When N_2O and CO_2 emissions were combined, ΔCO_2eq emitted in spring were no different
327 for LGM with respect to UR, and 49% lower in COM. In summer no treatment effect was found.
328

329 ***Soil temperature and moisture***

330 The average WFPS changed over time according to precipitation and irrigation events
331 during the different campaigns (Fig. 1). In 2007, the spring average WFPS was 0.62 (following 117
332 mm of rainfall during the monitored period); in the summer it increased from 0.34 pre-irrigation to
333 0.61 post-irrigation (40 mm of irrigation water and 30 mm of rainfall during the monitored period).
334 In 2008, precipitation was frequent; the average WFPS was 0.77 in spring (following 257 mm of
335 rainfall during the monitored period) and 0.49 in summer (21 mm of rainfall during the monitored
336 period). Treatments did not influence WFPS on any of the sampling dates (data not shown). Soil
337 moisture was significantly correlated to N_2O emission (correlation between daily N_2O-N emissions
338 and WFPS was significant, $R=0.43$; $P<0.01$), but not to CO_2 fluxes.

339 Soil temperature varied among seasons according to climatic conditions and crop presence.
340 The average soil temperature was 25.7 °C during spring 2007, which decreased from 24.3 °C pre-
341 irrigation to 21.3 °C post-irrigation in summer. During the wetter 2008, average soil temperature
342 was 20.2 °C in spring and 21.5 °C in summer. Because of corn shading, soil temperature in summer
343 was not much different from the spring temperature. Soil temperature was not correlated to daily
344 N_2O and CO_2 emissions.

345

346 ***Soil N and C availability indicators and their correlations with gaseous emissions***

347 Soil mineral N was composed almost exclusively of NO_3^--N; NH_4^+-N never exceeded 2 kg
348 ha^{-1} (data not shown). Fertilization increased soil NO_3^--N content in spring and in summer. In
349 spring, UR significantly increased (by 50%) the soil nitrate content relative to CK while COM and
350 LGM were intermediate (Table 5). A treatment effect was also present in summer when NO_3^--N
351 content in CK was 17% lower than in LGM and UR. Comparing the two years, soil NO_3^--N content
352 after fertilization was significantly higher in 2007 than in 2008 in both in spring and summer.

353 In spring, organic fertilizers (compost and vetch) increased the PMN by 20% with respect to
354 CK. Urea was lower than COM and not different from LGM (Table 5). In summer, PMN was still

355 16% higher in COM and LGM than in CK while UR was intermediate. Comparing the two years,
356 PMN in summer was significantly lower in 2007 than 2008.
357 Potential denitrification was significantly influenced by fertilizer addition. In spring, organic
358 fertilization (COM and LGM) increased the PD 50% with respect to UR while CK was
359 intermediate. In summer, UR was 2.5 times lower than the other treatments. Comparing the two
360 years, and similar to PMN, PD did not decrease in response to high 2008 precipitation. On the
361 contrary, PD in summer 2008 was 20% higher than in 2007.
362 No significant correlation was found between N_2O or CO_2 emissions and soil NO_3^--N
363 content or PMN. Average PD showed a positive correlation with cumulative CO_2 emissions
364 ($R=0.43$; $P=0.04$), but not with cumulative N_2O.
365
366 **DISCUSSION**
367 ***Gaseous emissions***
368 The availability of N to soil microorganisms was probably the driving factor in spring N_2O
369 emissions following fertilization. Losses were highest when N was supplied in a form readily
370 available for N_2O-producing microorganisms such as urea (Bertora et al., 2008), and were almost
371 null with compost. Surprisingly, fluxes following vetch incorporation were no different from those
372 of urea (Table 4), suggesting that vetch tissues were likely more prone to mineralization than the
373 stable organic matter of compost, which had a lower soluble C:total C ratio and higher ADL and ash
374 content. Moreover, compost contained almost no inorganic N ready for the nitrification or
375 denitrification processes (data not shown). Mineralization of legume residue is known to enhance
376 N_2O emissions (Rochette et al., 2004; Yang and Cai, 2005). Our results were similar to those of
377 López-Fernández (2007) and Meijide et al. (2007) who found a significantly lower N_2O emission
378 with organic fertilizers (compost in particular) as compared to urea. Beside N availability, soil
379 moisture fostered N_2O production, as showed by the significant correlation between daily N_2O
380 fluxes and WFPS.

Two considerations arise when evaluating calculated EFs. First, the EF calculated for LGM for which total N of the biomass is considered added N (recommended by IPCC (2007b)), was smaller than that calculated when only fixed N was considered added N. Compared with other N sources, fertilization with leguminous green manures suffers from the addition to the soil of N taken up by the crop together with the fixed N, thus increasing N_2O emissions. Second, the 1% default Tier 1 emission factor proposed by the IPCC (2007b) to estimate N_2O emissions from all N sources is high, relative to our results, for compost fertilization and low for vetch and urea. We remain cautious regarding the result, as it requires confirmation by measurements integrating year round N_2O measurements. Our findings indicate the need for source and site-specific coefficients for the compilation of national greenhouse gas inventories (Velthof et al., 2002; Flynn et al., 2005; Helgason et al., 2005; Bertora et al., 2008; Halvorson et al., 2009). Treatment comparison biases in our study could derive from variation in the percentage of N lost as N_2O with the N fertilization rate. Given that the percentage of N lost as N_2O remains constant or rises with increasing N fertilization (Granli and Bøckman, 1994; Mosier et al., 2006; Snyder et al., 2009), and since vetch in both years and compost in 2007 added higher amounts of N to the soil than urea, then the EF calculated for LGM and COM were either comparable or overestimated with respect to UR.

During summer, as opposed to spring, the year effect and absence of treatment effect suggest that N_2O emissions may be influenced more by climatic conditions than by the fertilizer applied in spring. We expected N_2O emission to increase after the 2007 summer irrigation, but no peak was observed. This contrasted with the findings of other authors (Ball et al., 1999; Velthof et al., 2002; Ruser et al., 2006; Werner et al., 2006). Result differences could be explained if N_2O emissions after dry-rewetting cycles occurred rapidly (possibly, within one day) before detection (Steenwerth et al., 2005; Yanai et al., 2007; Bertora et al., 2008). We excluded the possibility that, if denitrification was the main producer of N_2O, lower N_2O fluxes in summer 2007 than 2008 were caused by a lower N_2O:N_2 ratio (Granli and Bøckman, 1994) because higher WFPS was present in

406 summer 2007 than in that of 2008, but water drainage in the soil is swift enough to allow gases to
407 escape from the soil.
408 Organic fertilizers increased soil CO_2 fluxes in spring. Significantly higher emissions were
409 found in LGM than in CK while COM had an intermediate value. The results were likely partly
410 related to the amount of added C. We exclude the possibility that our experimental conditions could
411 have led to dissociation of soil carbonates and subsequent chemical CO_2 release since the soil pH
412 was above the threshold carbonate dissociation value (7.2). Frequent rainfall events during spring
413 2008 did not increase cumulative CO_2 emissions respect to spring 2007, probably because of the
414 consequent colder soil temperature.
415 Summer CO_2 emissions were influenced by climatic conditions and not by fertilizer applied
416 during the spring, as was the case with N_2O emissions. Soil rewetting via irrigation following a dry
417 period in 2007 promoted SOC mineralization characterized by increased CO_2 fluxes (Jabro et al.,
418 2008). The absence of an observed treatment effect in summer relative to spring was probably a
419 consequence of reduced SOC oxidation and increased root respiration. In both summers, basal
420 respiration was only about 10 to 30 kg CO_2-C ha^{-1} d^{-1} higher than when measured in spring. We
421 expected to observe higher differences due to the contribution of root respiration to CO_2 fluxes from
422 soil organic matter mineralization. Our fluxes are, however, in agreement with those of Rochette
423 and Flanagan (1997) and Rochette et al. (1999), who showed that increased CO_2 emissions from
424 summer root activity are offset by a decrease in SOC respiration. A similar increase in CO_2 fluxes
425 between spring and summer was found by Mosier et al. (2006) and Alluvione et al. (2009) in
426 conventionally-tilled continuous corn. If any difference in soil organic matter respiration was
427 present among the various treatments, then it is possible that the N fertilization effect was masked
428 by root activity and C rhizodeposition oxidation (Sainju et al., 2008; Alluvione et al., 2009).
429 We found a positive and significant correlation between CO_2 and N_2O emissions. This was
430 probably an outcome of the simultaneous interplay of mineralization and N_2O production or of

431 denitrification enhancement due to anaerobic sites where high amounts of CO_2 were produced
432 (Granli and Bøckman, 1994).
433 When comparing N_2O and CO_2 fluxes among treatments, it is important to consider that
434 observed differences were probably underestimated. Linear interpolation is known to be a less than
435 optimal model for estimating soil gas fluxes since it does not account for the inevitable immediate
436 depleting effect of chamber deployment on gas fluxes (Hutchinson and Mosier, 1981; Healy et al.,
437 1996). Despite this limitation, Venterea et al. (2009) suggest using linear interpolation when the
438 research goal is comparison between treatments as in our case. Thus, calculated EF and ΔCO_2eq
439 must be used only as indicators of the potential of tested fertilizers to reduce the impact of
440 fertilization on global warming while their quantitative information needs to be considered with
441 caution.
442
443 ***Soil N and C availability indicators and their correlations with gaseous emissions***
444 Soil N and C availability was significantly influenced by the different N sources and by the
445 tendency of added materials to be mineralized.
446 The addition of readily mineralizable N with urea (Bertora et al., 2008) tended to increase
447 nitrate content in spring relative to vetch and compost, which probably supplied mineral N to the
448 soil more gradually according to their respective organic matter mineralization rates.
449 Across the two years, soil NO_3^--N content after fertilization was significantly lower in 2008
450 than in 2007 in both spring and summer, probably due to the high frequency of precipitation events
451 that leached some N into deep soil layers.
452 Surprisingly, cumulative spring N_2O emissions were not correlated with average soil NO_3^--
453 N content. Though this result is not common in the literature, it is reasonable in our experiment for
454 a number of possible reasons. First, nitrate availability was probably higher and more different
455 among treatments in deep soil layers (beyond 7.5 cm). In fact, many studies show that materials
456 applied to the surface tend to move to the bottom of the ploughed layer by soil inversion from

457 tillage (Cousens and Moss, 1990; Swanton et al., 2000; Roger-Estrade et al., 2001). Second, a more
458 abundant facultative aerobe population and higher potential denitrification rates can be found below
459 the top soil layer in ploughed soils (Linn and Doran, 1984b; Groffman, 1985; Venterea et al., 2005),
460 suggesting that the N_2O we monitored was produced below 7.5 cm. A third possible reason for the
461 absence of nitrate and nitrous oxide flux correlation is the scale of phenomena. In fact, N_2O is
462 known to be produced in soil microsites, where local conditions fostering nitrification or
463 denitrification can vary radically from the surrounding soil (Parkin, 1987; Granli and Bøckman,
464 1994; Loecke and Robertson, 2009). It is possible that at a microsite scale, high NO_3^--N content
465 occurred and enhanced N_2O emissions through denitrification as opposed to dilution at the soil
466 sampling scale. This seems to be particularly true for LGM, where N_2O was likely produced around
467 decomposing vetch tissue aggregates. Loecke and Robertson (2009) found higher N_2O emissions
468 when residue was not uniformly distributed in the soil as a consequence of favorable local
469 conditions. Finally, it is also possible that N_2O was mainly produced by nitrification with NO_3^--N
470 content having no influence.

471 Potentially mineralizable N in anaerobiosis has been confirmed as a viable indicator to
472 monitor SON evolution under different treatments even in the brief period before equilibrium. In
473 fact, PMN measurements followed a pattern consistent with total SON evolution —which was
474 significantly higher in COM (1.22 mg N kg^{-1}) *versus* LGM (1.11 mg N kg^{-1}) and UR (1.08 mg N
475 kg^{-1}) in fall 2008. Increased SON and PMN after compost fertilization were consequential to very
476 low mineralization rates of added organic compounds (Debosz et al., 2002; Ginting et al., 2003;
477 Hargreaves et al., 2008). Work by both N'Dayegamiye and Tran (2001) and by Kuo and Jellum
478 (2002) have confirmed SON increases following legume incorporation, although the change can be
479 limited over time (Cherr et al., 2006b). The low PMN in CK was likely related to the lower amount
480 of N returned to the soil from crop residue, which in turn, supplied lower amounts of mineralized N.
481 Though root biomass was not directly measured, we surmise that a lower amount of N was returned
482 by crop root since the total N taken up by the aboveground biomass was not different among

483 fertilized treatments, but it was significantly lower in CK (two year averages for silage corn N
484 uptake: 146 kg N ha^{-1} in CK, 180 kg N ha^{-1}in COM, 201 kg N ha^{-1} in LGM, and 210 kg N ha^{-1} in
485 UR; treatment effect: P=0.01). The high PMN was not associated with CO_2 or N_2O emission.
486 The lower PD of urea with respect to organic fertilizers both in spring and in summer was
487 probably due to two phenomena: (1) a rapid mineralization of labile C compounds in soil organic
488 matter following urea application (Kuzyakov et al., 2000; Huang et al., 2004), and (2) a
489 simultaneous and opposite increase in the treatments receiving compost and vetch tissues. The
490 increase in soil labile C after compost and green manure additions agrees with findings from other
491 authors (Chantigny, 2003; Huang et al., 2004; Melero et al., 2007). A tendency for higher PD in CK
492 than in UR might be the consequence of a higher root exudate production by N-stressed corn in the
493 non-fertilized treatment (Rees et al., 2005), as shown by silage corn N uptake. We believe that this
494 difference is not linked to the amount of crop residue returned to the soil. Aboveground biomass
495 was low in CK and consistent among fertilized treatments (two year averages: 17.1 Mg d.m. ha^{-1} in
496 CK, 20.1 Mg d.m. ha^{-1} in COM, 21.6 Mg d.m. ha^{-1} in LGM, and 22.0 Mg d.m. ha^{-1} in UR;
497 treatment effect: P<0.01). In this paper, we report results from the second and third years after
498 treatment establishment. Therefore, it is likely that the difference in labile C amounts shown by PD
499 between mineral (UR) and organic (COM and LGM) fertilization was the result of a change that
500 occurred in the soil organic matter. Though total SOC measured in fall 2008 was not significantly
501 different among treatments (data not shown), PD was as expected with SOC increase when organic
502 fertilizers are applied.
503 Potential denitrification is an estimator of labile C available to soil microorganisms. In our
504 experiment, labile C was positively correlated with CO_2 emissions, but did not influence N_2O
505 fluxes. Our set-up did not allow identification of the relative contribution of denitrification and
506 nitrification to N_2O emissions. If denitrification was the driving process, then the amount of labile C
507 supplied by soil organic matter was probably sufficient to support N_2O production, regardless of the
508 treatment. On the contrary, if nitrification was the primary contributor, then the observed absence of

509 correlation between PD and N_2O would be expected since nitrifying bacteria are autotrophic. This
510 would also be consistent with the absence of correlation between soil nitrate content and N_2O
511 emissions that we found.
512
513 ***CO_2 equivalents emitted***
514 The calculated ΔCO_2eq values provided useful information on the combined contribution of
515 fertilizer induced N_2O and CO_2 emissions on global warming; however, they do not fully
516 characterize the effect of the studied fertilizers on global warming because soil C evolution, rather
517 than CO_2 fluxes, should be considered.
518 Compost application reduced ΔCO_2eq emissions following its use as a corn fertilizer while
519 vetch did not (Table 3). Though fertilization after compost incorporation in the soil seems to have a
520 very low impact on global warming, emissions associated with the collection of the organic
521 materials and the composting process need to be included for a more comprehensive evaluation.
522 Also, the advantage of compost utilization could be reduced in the farmers' practice when compost
523 is added on the basis of an estimated mineralizable N. Fertilization with leguminous green manure
524 seems to be a valuable substitute for mineral N fertilizer as its greenhouse gas impact on global
525 warming is similar to urea, and emissions during crop management can be lower than during
526 mineral fertilizer synthesis.
527
528 **CONCLUSIONS**
529 Compost and vetch significantly affected soil N_2O and CO_2 emissions. The amount and
530 quality of the organic matter they supplied regulated N and C mineralization rates and
531 consequently, the microbial processes producing N_2O and CO_2. During the first month in spring
532 after soil incorporation, the stable organic matter in compost mineralized slowly, thereby
533 significantly reducing N_2O emissions and without increasing CO_2 emissions compared to urea.
534 Compared to compost, fresh vetch tissues decomposed faster and sustained N_2O fluxes at a rate

535 similar to urea while increasing CO_2 emissions. When N_2O and CO_2 losses were combined, using
536 compost decreased CO_2 equivalents emitted per kg of fertilizer N applied by 49% relative to urea
537 while using vetch did not. In summer, there was no fertilizer effect detected on N_2O and CO_2 fluxes.
538 Soil mineral N content, potentially mineralizable N in anaerobiosis, and potential
539 denitrification all agreed with expected soil organic matter evolution. What did differ from our
540 expectation was that these measurements did not add substantial information about the processes
541 leading to N_2O and CO_2 emissions.
542 We conclude that compost proved to be efficient in reducing N_2O emissions and
543 maintaining CO_2 fluxes, thereby reducing the global warming impact of fertilization. Hairy vetch
544 did not, as no reduction of N_2O fluxes, relative to urea, was obtained. Further research is needed to
545 accurately compare tested fertilizers. Flux measurements need to be extended throughout the year
546 (to include off-peak and non-crop growing season emissions), and combined with the entire life
547 cycle of the fertilizer—be it compost (Hao et al., 2001) or urea production (Jenssen and Kongshaug,
548 2003) or hairy vetch field management. As GHG emissions during industrial N fixation are very
549 high, it is likely that both compost and vetch could be viable substitutes for urea. Furthermore, the
550 highly variable N emission factors we found, as compared to the default 1% value proposed by the
551 IPCC (2007b), confirm the need for site- and source-specific estimations.

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839 **List of Figures**

840 Fig. 1. Daily N_2O fluxes (a), daily CO_2 fluxes (b) and precipitation events (including irrigation), soil
841 temperature and water-filled pore space (WFPS) in the 0- to 7.5- cm soil layer (c) during
842 measurement campaigns.

843 Table 1. Selected soil chemical and physical properties in the 0- to 30- cm layer (plowed layer) at

844 the study site.

Soil texture	
Sand, %	36.0
Silt, %	56.5
Clay, %	7.5
Chemical properties	
pH (soil:water = 1:2.5)	8.1
Active $CaCO_3$, g kg^{-1}	24
Organic C, g kg^{-1}	10.2
Total N, g kg^{-1}	1.0
C.E.C., $cmol_c$ kg^{-1}	12.8
Exchangeable Ca, $cmol_c$ kg^{-1}	11.4
Exchangeable Mg, $cmol_c$ kg^{-1}	0.4
Exchangeable K, $cmol_c$ kg^{-1}	0.1
Olsen P, mg kg^{-1}	16

845

846 Table 2. Main properties of the compost and vetch incorporated into the soil in 2007 and 2008.

	Compost		Vetch	
	2007	2008	2007	2008
Dry matter, %	61.0	61.6	79.1	89.1
Organic Matter, % dm	51.1	45.4	89.5	88.2
Organic C, % dm	25.6	22.7	45.0	45.1
NDF‡, % dm	33.9	30.5	56.0	44.6
ADF‡, % dm	23.2	21.1	41.0	31.7
ADL‡, % dm	11.2	12.7	11.0	7.9
Ash, % dm	39.1	47.9	10.5	12.7
Total N, % dm	2.1	2.3	2.7	3.5
C:N	12.2	9.8	17.0	12.9
Soluble C:total C‡	0.34	0.33	0.37	0.49
Total supply, Mg d.w. ha^{-1} †	7.3	4.9	2.9	2.6
Total C supply, kg ha^{-1}‡	1877	1123	1910	1734
Soluble C supply, kg ha^{-1}‡	635	369	714	856
Total N supply, kg ha^{-1} ‡	154	110	260	298
Estimated N fixation, kg N ha^{-1}‡§			151	173

847

848 ‡ NDF: neutral detergent fiber, ADF: acid detergent fiber, ADL: acid detergent lignin according to

849 the Robertson and Van Soest (1981) method; soluble C was computed by the difference between

850 total C and C in the NDF fraction.

851 † Only the measured aboveground biomass of the vetch is reported.

852 ‡ Shoot : root C and N ratios were estimated to be 2.1

853 § N fixation was estimated to be 58% of total vetch N.

Table 3. Cumulative CO_2 and N_2O emissions and the corresponding increase in CO_2 equivalents above that of CK (ΔCO_2eq) for the different treatments during measurement campaigns (main effect differences are indicated by upper case letters). The following were the studied treatments: fertilization with compost (COM), leguminous green manure (LGM), urea (UR), and the absence of N fertilization (CK).

	Spring			Summer		
	CO_2 [kg CO_2-C ha^{-1}]	N_2O ǂ [g N_2O-N ha^{-1}]	ΔCO_2eq † [kg CO_2eq ha^{-1} kg^{-1} fertilizer N]	CO_2 [kg CO_2-C ha^{-1}]	N_2O ǂ [g N_2O-N ha^{-1}]	ΔCO_2eq † [kg CO_2eq ha^{-1} kg^{-1} fertilizer N]
COM	1499 AB‡	830 B	12.24 B	319	27	0.38
LGM	1828 A	3802 A	27.95 A	313	48	0.39
UR	1343 AB	4726 A	24.02 A	278	16	-0.70
CK	1058 B	637 B	/	302	26	/
2007	1533	1877	20.92	371	8	0.35
2008	1331	1651	21.89	236	67	-0.31
Average	1432	1760	21.40	303	28	0.02
Year	0.19	0.52	0.33	<0.01	<0.01	0.30
Fertilization	0.01	<0.01	0.02	0.29	0.29	0.32
Interaction	0.38	0.43	0.80	0.22	0.22	0.18
SD	359	2	12.95	47	2	1.30

ǂ Data normalization for the ANOVA was performed by log-transformation (natural logarithm). Back-transformed data are reported.

† Carbon dioxide equivalents emitted by the different treatments above the emissions from CK. To account for the amount of N distributed in the different treatments, values referred to one kg of fertilizer N applied per ha. For the vetch, only the fixed N was counted as added fertilizer. Carbon dioxide equivalents of N_2O were calculated using a global warming potential of 298 kg CO_2 equivalent per kg of N_2O (IPCC, 2007a). As reported N_2O emissions are geometrical averages, CO_2 equivalents do not correspond to the arithmetical sum of CO_2 and CO_2 equivalent N_2O values.

‡ When the effect of the factor was significant, letters were used to separate between means according to the results of the Sidak post-hoc test.

Table 4. Nitrous oxide emission factor of applied N (EF) in fertilized treatments during spring campaigns. In LGM, N_2O EFs were calculated taking into account only the N fixed (values calculated considering the total N in the vetch residue are reported in brackets). The following were the studied treatments: fertilization with compost (COM), leguminous green manure (LGM), or urea (UR).

	COM		LGM		UR		Avg
2007	0.19		1.95 (1.13)		4.05		2.06 (1.79)
2008	0.03		2.67 (1.55)		2.78		1.83 (1.45)
Avg.	0.11	A[†]	2.31 (1.34)	B(A)	3.42	B	1.95 (1.62)
Year	0.70		(0.50)				
Fertilization	<0.01		(<0.01)				
Interaction	0.40		(0.37)				
SD	1.22		(1.02)				

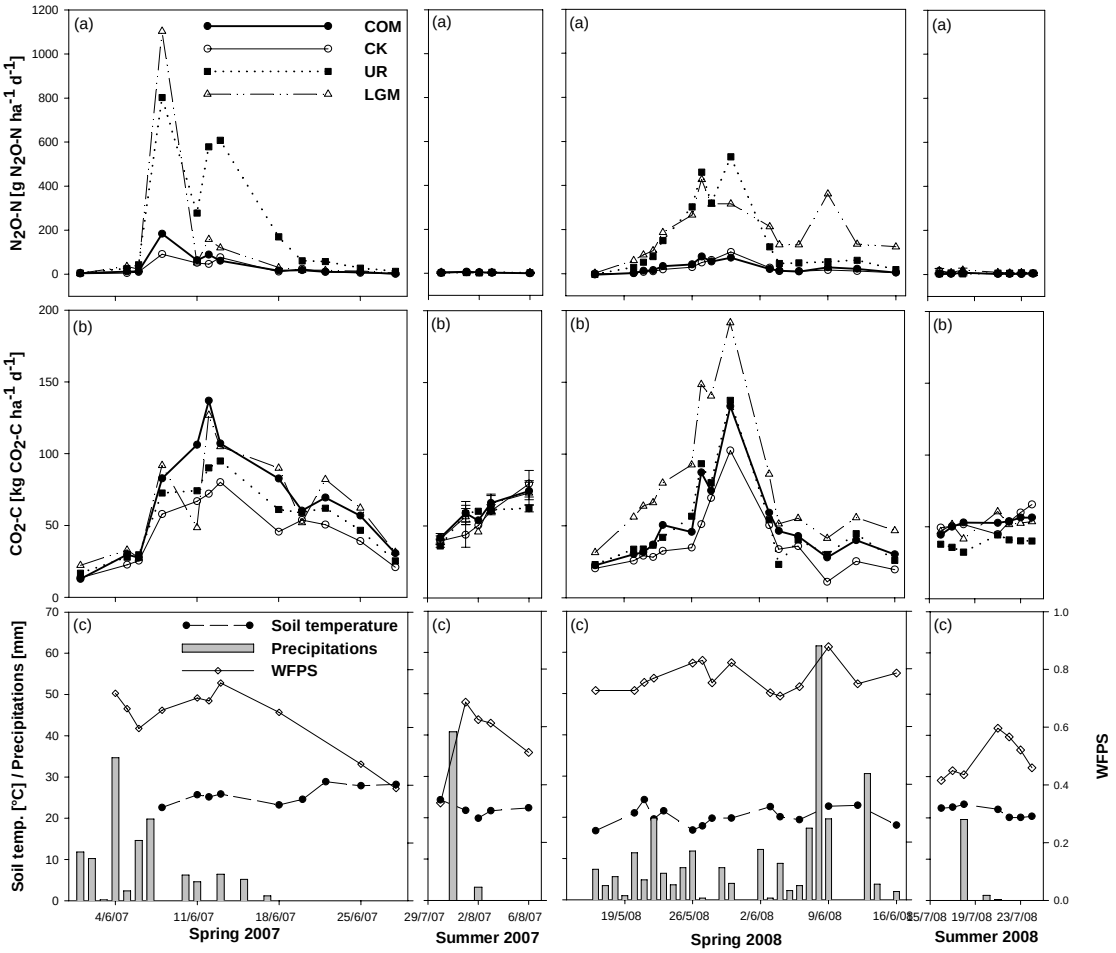
[†] When the effect of the factor was significant, letters were used to separate between means according to the results of the Sidak post-hoc test.

879 Table 5. Average soil NO_3^--N content, potentially mineralizable N (PMN), and potential
880 denitrification (PD) in the 0- to 7.5- cm layer in spring and summer. The following were the
881 studied treatments: fertilization with compost (COM), leguminous green manure (LGM), or
882 urea (UR).

	Spring			Summer		
	NO_3^--N[¶] [kg NO_3^--N ha^{-1})]	PMN [kg NH_4^+-N ha^{-1}]	PD[¶] [kg N_2O-N ha^{-1}]	NO_3^--N[¶] [kg NO_3^--N ha^{-1}]	PMN [kg NH_4^+-N ha^{-1}]	PD[¶] [kg N_2O-N ha^{-1}]
COM	3.94 ab[§]	10.21 c	0.66 b	1.45 ab	12.04 b	0.62 b
LGM	4.36 ab	9.74 bc	0.67 b	2.60 b	12.36 b	0.68 b
UR	5.14 b	8.94 ab	0.44 a	5.57 b	11.15 ab	0.28 a
CK	3.41 a	8.17 a	0.54 ab	0.27 a	10.44 a	0.76 b
2007	12.87	8.61	0.56	8.92	10.98	0.88
2008	1.35	9.92	0.60	0.26	12.00	1.06
Average	4.16	9.26	0.58	1.52	11.49	0.55
Year	<0.01	<0.01	0.47	<0.01	0.05	<0.01
Fertilization	0.01	<0.01	0.03	0.02	0.05	0.03
Interaction	0.52	0.18	0.55	0.71	0.44	0.12
SD	1	1	1	48	1	2

883

884 [¶] Data normalization for the ANOVA was performed by log-transformation (natural logarithm).
885 Back-transformed data are reported.
886 [§] When the effect of the factor was significant, letters were used to separate between means
887 according to the results of the Sidak post-hoc test.



888

889 Fig. 1. Daily N_2O fluxes (a), daily CO_2 fluxes (b), precipitation events (including irrigation), soil

890 temperature, and water-filled pore space (WFPS) in the 0- to 7.5- cm soil layer (c) during

891 measurement campaigns.

892