



N₂O emission from organic barley cultivation as affected by green manure management

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Abstract. Legumes are an important source of nitrogen in stockless organic cereal production. However, substantial amounts of N can be lost from legume-grass leys prior to or after incorporation as green manure (GM). Here we report N₂O emissions from a field experiment in SE Norway exploring different green manure management strategies: mulching versus removal of grass-clover herbage during a whole growing season and return as biogas residue to a subsequent barley crop. Grass-clover ley had small but significantly higher N₂O emissions as compared with a non-fertilised cereal reference during the year of green manure (GM) production in 2009. Mulching of herbage induced significantly more N₂O emission (+0.37 kg N₂O-N ha⁻¹) throughout the growing season than removing herbage. In spring 2010, all plots were ploughed (with and without GM) and sown with barley, resulting in generally higher N₂O emissions than during the previous year. Application of biogas residue (60 kg NH₄⁺-N + 50 kg organic N ha⁻¹) before sowing did not increase emissions neither when applied to previous ley plots nor when applied to previously unfertilised cereal plots. Ley management (mulching vs. removing biomass in 2009) had no effect on N₂O emissions during barley production in 2010. In general, GM ley (mulched or harvested) increased N₂O emissions relative to a cereal reference with low mineral N fertilisation (80 kg N ha⁻¹). Based on measurements covering the growing season 2010, organic cereal production emitted 95 g N₂O-N kg⁻¹ N yield in barley grain, which was substantially higher than in the cereal reference treatment with 80 kg mineral N fertilisation (47 g N₂O-N kg⁻¹ N yield in barley grain).

1 Introduction

Nitrous oxide (N₂O) is a potent greenhouse gas involved in ozone depletion and global warming (Ravishankara et al., 2009). During the past 250 yr, the atmospheric N₂O concentration has increased by 15 % (Artuso et al., 2010; Mosier, 1998). Agriculture is estimated to be responsible for 38 % of the N₂O emitted to the atmosphere (IPCC, 2007). Direct N₂O emissions from cultivated soils are often estimated as a constant fraction of applied N (Tier 1, IPCC, 2007) reflecting the importance of fertiliser induced emissions for annual N₂O budgets.

However, there is a broad consensus that cultivation history, crop rotation and local soil conditions modify fertiliser induced N₂O emissions and that management strategies have to be studied in a local context if regionalised emission factors are to be defined (Petersen et al., 2006). There is no conclusive evidence so far linking specific cultivation systems to specific emission factors, however. N₂O emissions from cultivated soils are notoriously variable both in space and time owing to the complexity of soil and cultivation driven controlling factors and their interactions (Olesen et al., 2006). Accordingly, agronomic systems cannot be studied independently of local environmental and economical settings and, therefore, cultivation-specific N₂O emission potentials must be determined locally.

One possible cultivation strategy in Norway is organic farming. A national aim of organic production has existed in Norway since 1999, and the current aim is 15 % organic production and consumption by 2020 (MAF Ministry of Agriculture and Food, Norway 2009). Organic farming is based

on the principle of internal nitrogen cycling, thus, reducing the use of extraneous nitrogen while maximising the crop yield and maintaining soil fertility (Watson et al., 2002; Chirinda et al., 2010). Traditionally, this goal is pursued by a diversified production with livestock densities well balanced with crop production. However, the areas with the most appropriate climate, machinery and infrastructure for arable farming in Norway are dominated by stockless farming. A similar trend of specialisation in organic farming has been reported for the whole of Europe with stockless arable farming gaining in relative importance (Stinner et al., 2008).

Stockless organic farming relies on leguminous green manures as source of external nitrogen, typically grown as whole-season grass-clover leys prior to an arable cash crop (Mueller and Thorup-Kristensen, 2001). The common advice in Norway is to mow 3–4 times during the growing season and to leave the herbage on the ground, hereafter termed “mulching” (Frøseth et al., 2008). The practice of leaving the herbage as mulch after repeated mowing increases N availability in the soil, due to large amounts of N accumulated in the green manure herbage. Although a large portion of N added with the green manure may be built into soil organic matter (Dahlin et al., 2011; Müller and Sundman, 1988), a substantial amount of N from herbage remains unaccounted for. Dahlin et al. (2011) suggested that 17 % of N in mulched herbage that could not be accounted for, was lost as NH₃ or NO_x. Flessa et al. (2002) found that 0.2–0.3 % of N returned with the grass mulch was emitted as N₂O during 7 weeks after application, whereas Larsson et al. (1998) observed that 0.1 to 1 % of N in grass and alfalfa mulch was lost as N₂O during three months depending of the N content of the mulch.

However, N₂O emitted directly from mulched foliage is likely to be only a small part of the total emissions. Large N losses may occur after ploughing mulched GM (Askegaard et al., 2005) because a large part of the nitrogen accumulated in foliage and stubbles becomes biologically available after ploughing (Askegaard et al., 2005). Only a small proportion (6–25 %) of the green manure N can be taken up by the subsequent crop (Müller and Sundman, 1988; Olesen et al., 2009). Ball et al. (2007) found that 75 % of the N₂O emission in an organic barley cropping system with a grazed grass-clover ley occurred shortly after ploughing. Excess mineral nitrogen (NO₃⁻, NH₄⁺) not readily taken up by the crop is a potential source for N₂O from denitrification or nitrification (Crutzen et al., 2008). Since fresh plant material also contains substantial amounts of readily degradable carbon, mulching may increase N₂O production by directly fuelling denitrification with carbon or by creating reductive zones by increased respiration (Flessa and Beese, 1995).

Increased N₂O losses from stockless organic systems would compromise the sustainability principle of organic farming unless mitigation strategies are sought and implemented. One option for increasing N recovery from green manures while reducing herbage-induced N₂O emissions could be to remove the aboveground biomass after mow-

ing and use it for biogas production (anaerobic digestion), with subsequent application of the biogas slurry back to the soil in spring next year. This could reduce the accumulation of mineral N during GM production, thus, reducing the risk for NO₃⁻ leaching and N₂O emissions (Moeller and Stinner, 2009). Due to the lower C/N ratio of the organic fraction and the prevalence of NH₄⁺ in the mineral fraction, biogas slurry may be less conducive to denitrification than fresh undigested biomass. Moreover, an increase of soil pH after biogas slurry application might decrease the N₂O/N₂ ratio of denitrification as shown in long-term liming experiments by Liu et al. (2010). Little is known about how the management of a whole-year GM ley affects N₂O emissions after ploughing. N₂O emissions during both GM production and after incorporation have to be studied when evaluating management options for grass-clover based organic cereal production with respect to N₂O emissions.

In the present study, we monitored N₂O emissions in year 2 and 3 of a stockless organic cereal production system consisting of grass-clover undersown in spring barley (year 1), a full-year grass-clover ley (year 2) followed by a spring barley crop (year 3). The objective of the study was to quantify N₂O emissions for different GM management strategies. We specifically studied the effect of mulching versus removing above-ground herbage during the year of GM production and the effect of returning harvested GM as anaerobically digested biogas residue to the cereal crop. We also evaluated the effect of grass-clover GM itself relative to a moderately fertilised (80 kg N ha⁻¹) reference treatment (continuous cereal) without green manure. Finally, we related cumulative N₂O emissions to dry matter yield and N content in the cereal crop to facilitate an assessment of the N₂O footprint of stockless organic cereal production.

2 Material and methods

2.1 Experimental site

The study was carried out on a plot experiment established in 2008 at the experimental farm “Østrevoll” in Ås (59°40' N, 10°47' E, 75 m a.s.l.), south east Norway. Long-term average (1961–1990) precipitation is 785 mm. The normal (1961–1990) mean annual temperature is 5.3 °C (Grimenes, 2009, 2010). The soil is a poorly drained, silty clay loam of marine origin (Typic endoaqualf). The field is artificially drained with pipes at one meter depth. Selected soil properties are given in Table 1.

The field had been managed organically since 1993 by a six-course arable crop rotation until 1997 with green fodder crops, leys, spring wheat or barley, row crops and oats (Bakken et al., 2006). From 1997 spring cereals were grown with undersown clover every year, leading to low yields and large populations of perennial weeds. For this reason a chemical weed control was necessary in spring 2008 before ploughing.

Table 1. Chemical and physical properties of the top soil at Østrevoll defined as naturally poorly drained silty clay loam. Standard deviations in parentheses where available. WFPS_{FC} = water-filled pore space at field capacity.

pH	6.2
Bulk Density (g cm ⁻³)	1.43 (0.13)
WFPS _{FC}	0.83
Organic C (%)	1.74 (0.19)
Total N (%)	0.21 (0.03)
C/N	8.5
P-AL(mg 100 g ⁻¹)*	5.4
K-AL (mg 100 g ⁻¹)*	9.4
Ca (mg 100 g ⁻¹)*	211.0
Na (mg 100 g ⁻¹)*	2.0

* From Bakken et al. (2006).

2.2 Treatments

The experimental design was a randomized complete block design (4 × 12 m plots) with 6 treatments arranged in 4 blocks. In 2008, a grass-clover mixture (2 g seed m⁻²) of 20 % red clover (*Trifolium pratense*, cultivar Nordia), 10 % timothy (*Phleum pratense*, cultivar Grindstad), 35 % meadow fescue (*Festuca pratensis*, cultivar Fure) and 35 % perennial rye grass (*Lolium perenne*, cultivar Napoleon) was sown together with barley (*Hordeum vulgare*, cultivar Sunnita, 16 g m⁻²), which was harvested after full ripening. This is the most common practice for establishing leys in Norway. In 2009, the ley was cut 3 times and either harvested or mulched. In spring 2010, the grass-clover was ploughed and barley was sown. Additional plots with cereals in all years (unfertilised in 2008 and 2009 and moderately fertilised with 80 kg N ha⁻¹ as NH₄NO₃ in NPK fertiliser in 2010) were used as a non-GM cereal reference. Treatments details are given in Table 2.

Three management strategies were established in the green manure ley during 2009: all 3 cuts mulched (G-3M), all 3 cuts harvested and removed (G-0M) and 2 cuts removed and the 3rd mulched (G-1M). All treatment plots were ploughed in spring 2010 before sowing of barley and biogas residue from fermentation of grass-clover herbage of similar composition was applied to treatments where biomass had been removed in the previous year (G-0M(B)) and to the unfertilised cereal reference (C-(B); Table 2). The application of biogas residue added 110 kg total N ha⁻¹ (containing 60 kg NH₄-N ha⁻¹), which is about 55 % of the total N removed with the herbage. The reference plots with oat were weeded manually at the end of April 2009 to remove clover which had been growing into the reference plots. Grain dry matter yields and N contents were recorded by standard methods (Frøseth et al., unpublished data).

2.3 N₂O flux sampling and calculation

Nitrous oxide (N₂O) emissions were measured by manually operated static chambers (Rochette and Eriksen-Hamel, 2008). In spring 2009, aluminium frames (51 × 51 × 20 cm) were pushed 10–15 cm into the soil. The metal frames had a 3 × 3 cm groove on top which was filled with water prior to deploying a chamber to ensure air tight connection (Maljanen et al., 2003). Aluminium chambers (19.5 cm high) equipped with a sampling port with a 3-way stop cock and a 3 mm diameter pressure equilibrium tube were used. The frame height from soil to the base of the groove was measured twice a year to calculate the exact volume of frames along with the known volume of the chambers. Frames were installed in duplicate in two replicate plots for each treatment in two of the available four blocks, giving a total of 24 (6 treatments × 2 replicate plots × 2 replicate frames). N₂O flux measurements were carried out by deploying the chambers on the frames for 45 min. Gas samples (15 ml) were taken from the chamber volume with 20 ml polypropylene syringes at regular intervals (0, 15, 30 and 45 min). Before sampling, the air was mixed by pulling and pushing the plunger of the syringe 3–4 times to obtain a well-mixed sample from the chamber atmosphere. The sample was transferred to a pre-evacuated 12 ml glass vial crimped with a butyl rubber septum. An extra collar (30 cm high) was used when crop height was above 20 cm to avoid damaging the crop. Gas sampling was done weekly or bi-weekly during the growing seasons of 2009 and 2010. High snow packs in winter 2009/2010 prevented sampling during winter. Frames were removed for agricultural operations like sowing and harvesting and placed back shortly after. Gas samples were analysed by gas chromatography (Model 7890A, Agilent, Santa Clara, CA, US) using a 30 m wide-bore Poraplot Q (0.53 mm) column at 38 °C with back flushing and He as a carrier gas. The electron capture detector (ECD) conditions were 375 °C with 17 ml min⁻¹ ArCH₄ (90/10 vol %) as makeup gas. The GC was connected to an autosampler via a peristaltic pump (Gilson minipuls 3, Middleton, WI, USA), pumping approximately 2.5 ml gas into a 250 µl sampling loop maintained at 1 Atm pressure. The injection system was back-flushed by He 6.0 before each sampling to minimise memory effects. Details of the autosampler and injection system are given in Molstad et al. (2007). N₂O emission rates were calculated from the linear or quadratic change in chamber gas concentration using the equation

$$F_{N_2O} = d_{N_2O}/dt \times V_c/A \times M_n/V_m \times 60 \quad (1)$$

where F_{N_2O} is the emission flux of N₂O in µg N₂O-N m⁻² h⁻¹, d_{N_2O}/dt the rate of change in N₂O concentration (ppmv min⁻¹) in the chamber headspace, V_c the volume of chamber (l), A the area covered by the chamber (m²), M_n the molecular mass of N in N₂O (g mol⁻¹) and V_m the molecular volume of gas at chamber temperature (l mol⁻¹). Chamber temperature was calculated as the average of measured

Table 2. Treatments in the 3-yr crop rotation. N₂O emissions were measured during the vegetation periods of 2009 and 2010. The field was spring ploughed and sown in 2008. Cereal reference plots were also ploughed in spring 2009. All plots were ploughed on 23 April and sown on 14 May 2010 with barley. Biogas residue and mineral fertiliser was applied on 12 May 2010.

Treatment	2008	2009	2010
G-3M	Barley <i>w</i> / GM	GM 3 cuts mulched	Barley
G-0M	Barley <i>w</i> / GM	GM 3 cuts removed	Barley
G-0M(B)	Barley <i>w</i> / GM	GM 3 cuts removed	Barley + biogas residue (110 kg N ha ⁻¹)
G-1M	Barley <i>w</i> / GM	GM 2 cuts removed, last mulched	Barley
C-(B)	Barley	Oats	Barley + biogas residue (110 kg N ha ⁻¹)
C-(M)	Barley	Oats	Barley + fertiliser (80 kg N ha ⁻¹)

G-3M = grass-clover herbage mulched, G-0M = grass-clover herbage removed, G-0M(B) = grass-clover herbage removed in 2009 and partly applied as biogas slurry in 2010, G-1M = first two cuts of grass-clover herbage removed, last cut mulched, C-(B) = cereal reference with biogas residue application in 2010, C-(M) = cereal reference with mineral fertiliser application in 2010.

chamber temperature at the beginning and end of deployment. Cumulative N₂O emissions (kg N₂O-N ha⁻¹ period⁻¹) were calculated assuming average flux between two measurements.

2.4 Soil sampling

Soil samples (0–20 cm) were taken at every date of gas sampling by a 30 mm diameter soil corer to determine soil moisture and mineral nitrogen (NH₄⁺ and NO₃⁻) content. In both years, 8 soil cores were sampled per treatment and pooled to one composite soil sample. In 2009, soil samples from GM treatments, where herbage was removed, were pooled (G-0M, G-0M(B), G-1M until 3rd cut) as were the cereal reference treatments C-(B) and C-(M), because there was no difference in treatment. In 2009, two soil depths (0–10 and 10–20 cm) were sampled in all the ley treatments and one depth (0–20 cm) in the cereal reference plots. Differences in mineral N content between the two soil depths in the ley treatments were found to be small and, therefore, weighted averages for 0–20 cm are presented. In 2010, only one depth (0–20 cm) was sampled in all the treatments, representing the plough layer. Soil moisture was determined in 2009 by drying 20 g fresh soil at 105 °C until constant weight. Water filled pore space (WFPS) was calculated as (gravimetric soil moisture × bulk density) / soil porosity assuming a soil density of 2.65 g cm⁻³. Bulk density was determined from volumetric samples (100 cm³) taken from 2–7 and 10–15 cm depth in the ley treatments and cereal reference plots in autumn 2009. Soil mineral N (NH₄⁺, NO₃⁻) was determined by

extracting 45 g fresh soil with 50 ml 2M KCL solution. Extracts were stored at –20 °C until analysed by a flow injection analyser (FIA Star 5010 analyser Tecator, Sweden) for NH₄⁺ and NO₃⁻. Soil pH was determined in spring 2010 in 1 : 5 *w* / *w* slurries with deionised water.

2.5 Field monitoring

Air temperature and precipitation data were obtained from a nearby weather station located on the University campus in Ås (Grimenes, 2009, 2010). Data loggers (Decagon EM50, Pullman, WA, USA) were installed in selected treatment plots end of 2009 to monitor soil moisture and temperatures (ECH₂O sensors, Decagon) on an hourly basis throughout 2010. Soil air probes (SAP) were installed end of 2009 in selected plots (G-3M, G-0M(B) and C-(B)) to monitor soil air concentration along with surface flux of N₂O. SAPs were constructed from small porous cups (height 3.5 cm, volume 6 cm³) made of sintered polyethylene (pore diameter 100 μm, type G1/4-5PU, Bosch Rexroth, Germany) and narrow Teflon tubes (length 1.5 m, inner diameter 0.97 mm) equipped with a 3-way stop cock at the sampling end. The cup was glued on top of a 1.5 m long 3.3 cm diameter PVC tube to protect the Teflon tube. The probes were installed in pre-augered (3.2 cm diameter) holes in a 60° angle to prevent water from running along the PVC tube to the porous cup. Two replicate sets of SAPs per treatment were installed at 15, 25 and 35 cm depth. Soil air samples were taken at each sampling date by carefully drawing 15 ml soil air by a polypropylene syringe and injecting the gas sample into

a pre-evacuated 12 ml vial after discarding the first 1.5 ml. CH₄, CO₂ and N₂O concentrations were analysed by GC as described above.

2.6 Statistical data analysis

Differences in cumulative N₂O emissions, barley dry matter yield and grain N content between the various treatments were tested using a general linear model (ANOVA). Two-way ANOVA was used to test for a combined effect of block and treatment on cumulative N₂O emissions. Multiple regression and Pearson correlation analysis was performed to find correlation between N₂O flux and ancillary variables. Differences in N₂O emission rates between the treatments on a daily basis were analysed by one way ANOVA. All statistical analyses were performed by using software Minitab 15 (Minitab Inc., State College, PA, USA).

3 Results

3.1 Soil conditions and climate

In 2009, the mean daily air temperature for the study period May to December was 0.8 °C warmer than the long-term (1961–1990) average of 9.7 °C for this period (Fig. 1d). The cumulative precipitation (for May to December) was 145 mm higher than the long-term average (785 mm for this period) with highest precipitation in July and August (Fig. 1d). In 2010, the mean daily temperature was 0.5 °C colder than the long-term average during the period April to November and the annual precipitation 69 mm higher than the long-term average (Fig. 2d). Spring came earlier in 2009 than in 2010. In 2010, there was a long cold period in early spring resulting in a 1.3 °C lower annual mean temperature as compared with 2009. In 2009, we observed higher WFPS in mulched GM than in removed GM and cereal reference, except shortly after rain events (Fig. 1d). In 2010, WFPS was similar in all treatments; therefore, average values are presented for 2010 (Fig. 2d).

3.2 Soil mineral N dynamics

In 2009, ammonium concentrations were quite stable during most of the growing season of the grass-clover ley, fluctuating between 2 and 5 µg NH₄⁺-N g dw soil⁻¹ (Fig. 1b). NO₃⁻ concentrations were lower (< 2 µg N g dw soil⁻¹), except for the cereal reference which had much higher concentrations than the leys in May–June 2009, presumably because plant uptake by the oats was still low (Fig. 1c). Small increase of NO₃⁻ content were observed after the 1st and 2nd cut in ley treatments, but comparison with NO₃⁻ dynamics in the cereal reference plots revealed that this could not be attributed directly to the cuts. NO₃⁻ concentrations showed an increase towards the end of the vegetation period in all treatments, when soil temperatures dropped in early October.

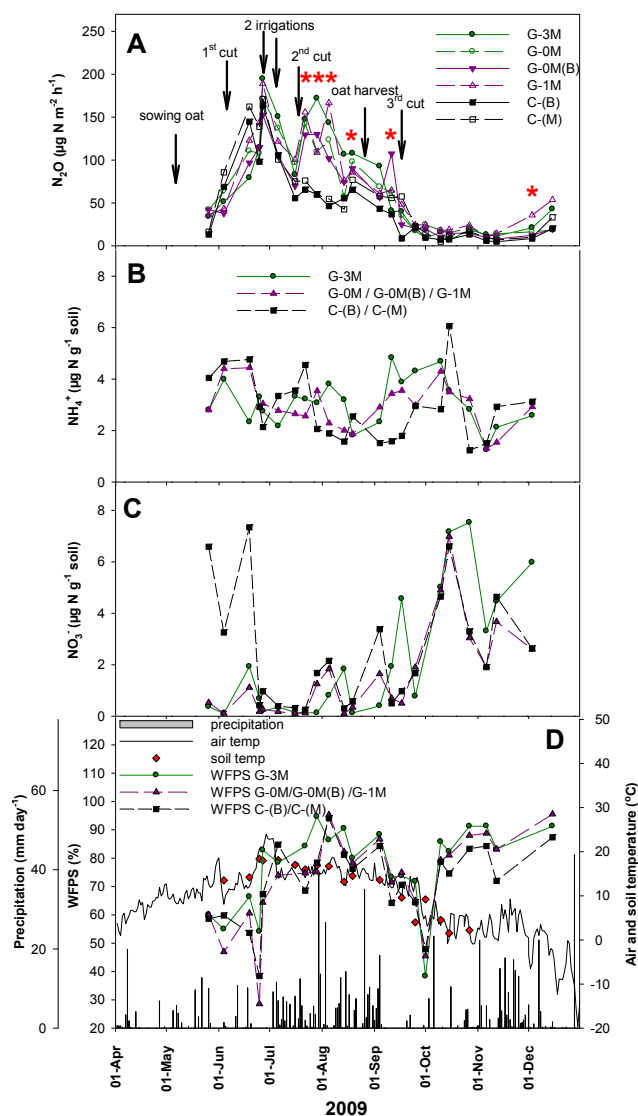


Fig. 1. (A) Average N₂O fluxes (µg N m⁻² h⁻¹) during the year of GM management (2009). Standard deviations are omitted to increase readability. Dates with significant treatment difference in N₂O emissions are marked by asterisk. (B) Soil NH₄⁺ (µg N g⁻¹ soil). (C) Soil NO₃⁻ (µg N g⁻¹ soil). Mineral N concentrations are from one composite sample (0–20 cm) per treatment on each sampling date. (D) Daily precipitation, air temperature, soil temperature and % WFPS at the time of flux measurement. Soil samples from G-0M, G-0M(B) and G-1M (until 3rd cut) as well as C-(B) and C-(M) were pooled, since these plots were treated equally in 2009. See Table 2 for treatment description.

In 2010, high NH₄⁺ and NO₃⁻ contents were observed 5 weeks after ploughing (Fig. 2b and c). At that point, NH₄⁺ was highest in treatments in which green manure had been ploughed under either as mulched grass-clover ley or as biogas residue. NO₃⁻ showed the opposite trend with higher concentration in the cereal reference treatment due to

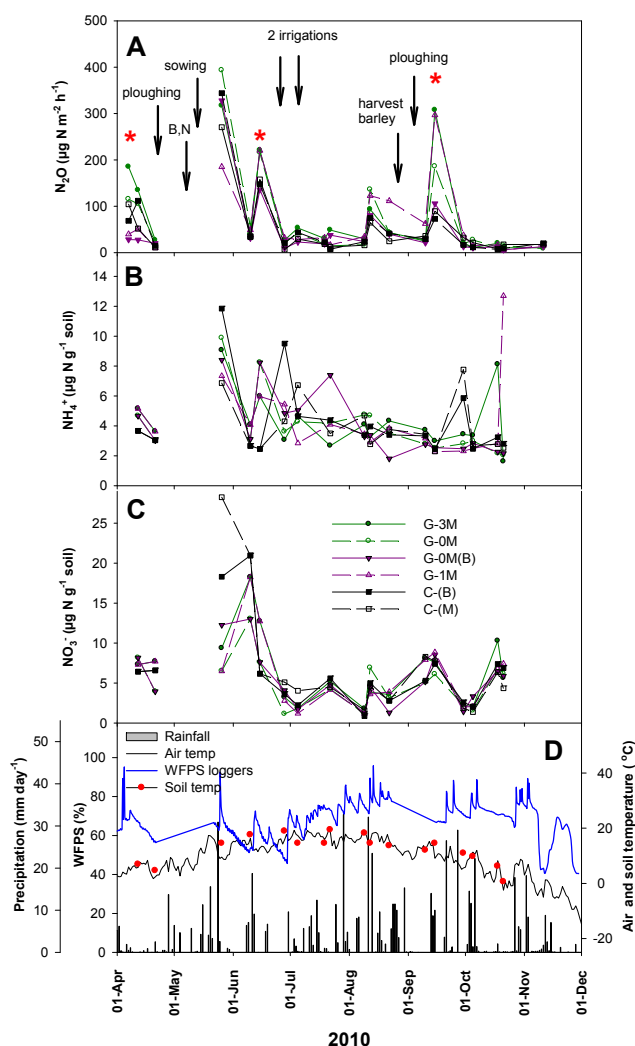


Fig. 2. (A) Average N₂O fluxes ($\mu\text{g N m}^{-2} \text{h}^{-1}$) during the year of barley production (2010). Standard deviations are omitted to increase readability. Dates with significant treatment difference in N₂O emissions are marked by asterisk. (B) Soil NH₄⁺ ($\mu\text{g N g}^{-1}$ soil). (C) Soil NO₃⁻ ($\mu\text{g N g}^{-1}$ soil). Mineral N concentrations are from one composite sample (0–20 cm) per treatment on each sampling date. (D) Daily precipitation, air temperature and % WFPS and soil temperature at the time of flux measurement. WFPS is calculated from weighted mean of hourly volumetric soil moisture readings at 5 and 20 cm depth. See Table 2 for treatment description. B = biogas residue, N = mineral N application.

mineral fertiliser application. Differences in NO₃⁻ contents between the treatments observed 5 weeks after ploughing converged rapidly during early summer and fluctuated uniformly throughout the remainder of the growing season irrespective of treatment, whereas fluctuations in NH₄⁺ concentrations were less uniform. In general, mineral N values were higher in 2010 than in 2009.

3.3 N₂O emission patterns

3.3.1 Green manure ley (2009)

In 2009, N₂O emissions showed a seasonal pattern roughly following the vegetation period with highest emission rates in the warm and wet period in June and July (Fig. 1a) and gradually decreasing emission rates in August and September. The N₂O emissions were in the range of 4–188 $\mu\text{g N}_2\text{O-N m}^{-2} \text{h}^{-1}$ with peak emissions in June–August and lowest emissions in October. N₂O emissions peaks were observed after cuts in the ley treatments, irrigation (in June after drought) and after ample rainfall in July and early August. In the cereal reference treatment, N₂O emissions ranged from 4.8 to 171 $\mu\text{g N}_2\text{O-N m}^{-2} \text{h}^{-1}$ showing two peaks, 27 June (171 $\mu\text{g N}_2\text{O-N m}^{-2} \text{h}^{-1}$) and 19 August (76 $\mu\text{g N}_2\text{O-N m}^{-2} \text{h}^{-1}$) which were associated with irrigation and rainfall events, respectively. N₂O peak emission after the 2nd cut was significantly higher in GM treatments than in the cereal reference, while no such difference was seen after the 3rd cut. A sudden drop in N₂O fluxes at the end of September was observed upon an 8 °C drop in temperature (14 to 6 °C) and a drop in WFPS from 65 to 47%. Stepwise regression included soil temperature and WFPS as the only significant factors in 2009 ($p = 0.035$) explaining 10% of the variation in N₂O emission rates.

3.3.2 Barley (2010)

After spring thaw 2010, significantly higher N₂O emission was observed in the G-3M treatment and lowest in GM removed (Fig. 2a). Elevated N₂O emissions were roughly proportional to N₂O accumulation in the soil atmosphere (Fig. 3a). During the growing season, N₂O emission showed an event-driven emission pattern characterised by short-lived peaks (Fig. 2a). High emissions were observed in all treatments when measurements were resumed 5 weeks after ploughing, while mineral N and soil air N₂O concentrations were still high (Figs. 2b, c and 3). Peaking N₂O emissions in mid-June were observed upon rapid increase in WFPS in all treatments resulting in fluxes up to 200 $\mu\text{g N}_2\text{O-N m}^{-2} \text{h}^{-1}$, without showing any build up of N₂O in the soil atmosphere (Fig. 3). A second N₂O emission peak associated with drying-rewetting was recorded on 12 August at high WFPS (80%), resulting in a marked accumulation of N₂O in the soil atmosphere (Fig. 3). Finally, all treatments responded with elevated N₂O emissions to autumn ploughing on the 15th of September with mulched GM treatments showing significantly higher emission rates than the other treatments. Emission rates levelled off to low values ($< 30 \mu\text{g N}_2\text{O-N m}^{-2} \text{h}^{-1}$) in October. Reference plots had a tendency for lower emission fluxes than GM treatment plots, particularly towards the end of the growing season. Biogas residue application to cereal reference plots (C-(B)) or GM treatment (G-0M(B)) did not result in significantly

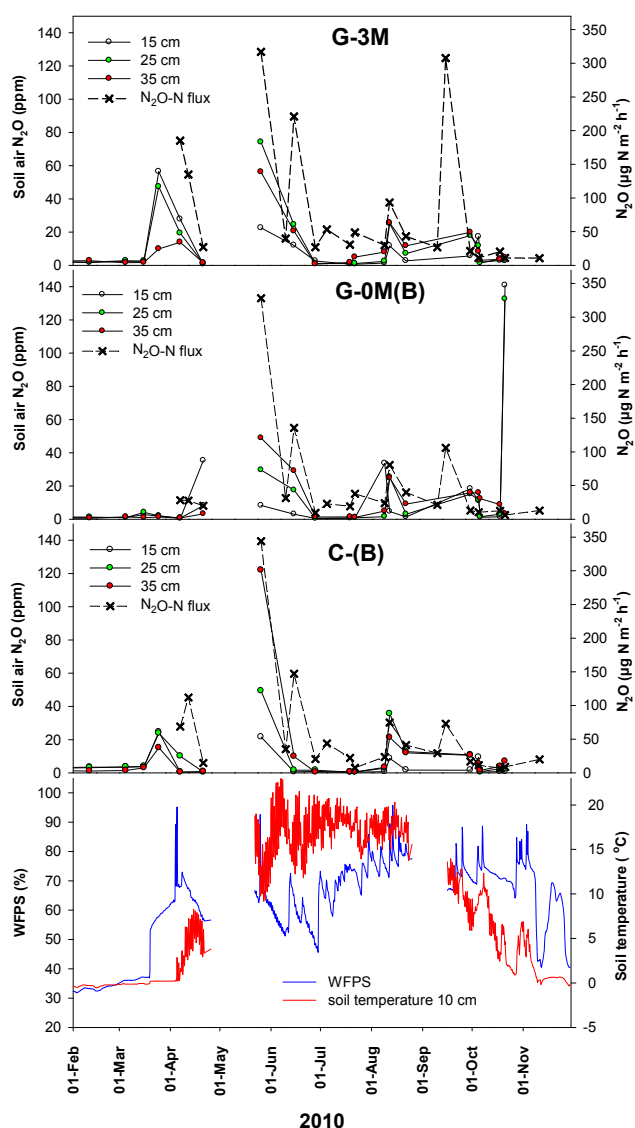


Fig. 3. N₂O concentration (ppmv) in soil atmosphere at three different depths (left y-axis) and N₂O emission rates (right y-axis) as affected by different green manure management strategies in 2010. The lowest panel shows the % WFPS (left y-axis) and soil temperature (right y-axis) in 10 cm depth. Soil air N₂O concentrations are average values from two replicate sets of soil air probes (3 depths). N₂O emission rates are the average of four replicates. Soil temperature was calculated as a weighted mean of soil temperature readings in 5 and 20 cm depth.

different fluxes as compared to their counterparts C-(M) and G-1M, respectively. On the contrary, G-1M (last cut mulched back in 2009) showed higher emissions in August to October than G-0M(B) which had received N in form of biogas residue in spring 2010 even though this effect was not statistically significant. In general, N₂O fluxes were higher in 2010 than in 2009. In 2010, NH₄⁺ and NO₃⁻ were the only factors included in a stepwise regression, explaining 25 % of

variation in emission rates. There was no significant correlation between N₂O emission and WFPS or soil temperature. In 2011, one year after the incorporation of GM and after ploughing under barley residues in the previous autumn, we observed high N₂O fluxes (150–393 µg N₂O-N m⁻² h⁻¹) in early spring after snow melt (Fig. 4). Soil air N₂O concentrations were up to 2000 times higher than ambient under the receding snow pack with a great variability in magnitude between replicates, showing no clear relationship to previous treatment.

3.4 Cumulative N₂O losses

Cumulative N₂O losses for the measurement periods in 2009 and 2010 (204 and 218 days, respectively) are given in Table 3. The values presented are averages of the four replicates (±SD) since no significant effect of block on cumulative N₂O emission was found in neither year. In 2009, some treatments were identical (as there was no application of biogas residue; see Table 2). Combining these treatments, cumulative N₂O-N losses were highest in the mulched treatments, intermediate in the leys with herbage removal, and lowest in the cereal reference plots with oats (Table 4).

In 2010, the year of barley cropping, overall cumulative N₂O losses were higher than in 2009, ranging from 3.15 to 4.70 kg N₂O-N ha⁻¹ 218 d⁻¹. The highest N₂O losses were observed in the G-3M and G-0M treatments (4.70 and 4.67 kg N₂O-N ha⁻¹ 218 d⁻¹, respectively), both of which were significantly higher than in the minerally fertilised treatment C-(M). Treatments which had received only one GM harvest either as mulch in 2009 (G-1M) or as biogas residue in 2010 (G-0M(B) and C-(B)) were not statistically different from each other. The lowest N₂O-N loss (3.15 kg N₂O-N ha⁻¹ 218 d⁻¹) was observed in the minerally fertilised cereal reference (C-(M)). In this treatment, 3.9 % of the applied N (80 kg N ha⁻¹) was lost as N₂O (including background emission), whereas 2.4 % of the total N returned with the mulched aboveground biomass (195 kg N ha⁻¹; Table 6) was lost in the G-3M treatment throughout the barley growing season.

3.5 Emission intensity

Dry matter and N yield of barley in 2010 were significantly lower in green manure treatments as compared to the moderately fertilised cereal reference treatment except for dry matter yield in G-0M(B) (Table 5), while the opposite was the case for N₂O-N loss. Together, this resulted in clearly higher relative N₂O loss (g N₂O-N kg⁻¹ N yield) for the organic treatments than for the mineral reference treatment except for G-0M(B) (Table 5).

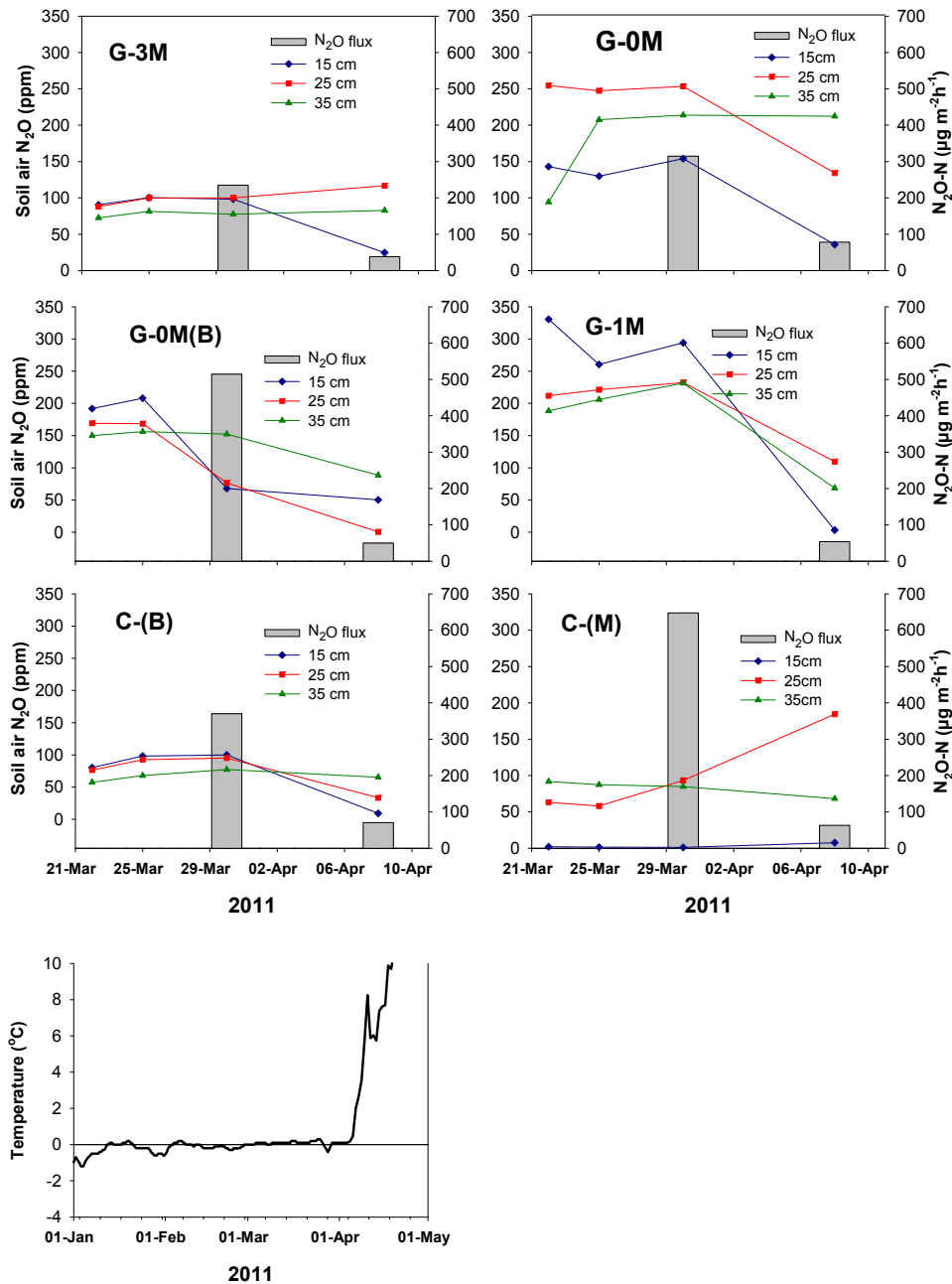


Fig. 4. N₂O concentration (ppmv) in soil atmosphere at three different depths (left y-axis) and N₂O emission rates (right y-axis) during snow melt in 2011, as affected by different green manure management strategies in 2009 and 2010. Soil air N₂O concentrations are average values from two replicate sets of soil air probes (3 depths). N₂O emission rates are the average of four replicates. Soil temperature was calculated as a weighted mean of soil temperature readings in 5 and 20 cm depth.

4 Discussion

In stockless organic farming, involving full-year clover leys, copious amounts of organic nitrogen are accumulated in aboveground biomass, raising the question as to how N losses can be minimised. In the present study, we evaluated different GM management strategies (mulching versus remov-

ing herbage and returning it as biogas residue) with respect to N₂O emissions throughout a two year crop rotation consisting of a full-year grass-clover ley followed by a year with cereal production. N₂O emission fluxes were measured during the snow-free period only and there were no flux measurements during several weeks after ploughing the GM in the second year (2010) because lengthy agronomic field

Table 3. Cumulative N₂O losses (N₂O-N kg ha⁻¹) over 204 and 218 days in the years 2009 (GM ley) and 2010 (barley), respectively, and sum over the two measurement periods (2009 + 2010). Given values are means of 4 replicate microplots ± standard deviation. Treatments not sharing the same letters are significantly different from each other within each year at the 95 % confidence interval.

Treatment	2009 N ₂ O-N (kg ha ⁻¹ 204 d ⁻¹)	2010 N ₂ O-N (kg ha ⁻¹ 218 d ⁻¹)	Sum of N ₂ O-N (kg ha ⁻¹ 422 d ⁻¹)
G-3M	3.26 ^a ±0.13	4.70 ^a ±1.2	7.96 ^a ±1.15
G-0M	3.05 ^{ab} ±0.36	4.67 ^a ±1.32	7.72 ^{ab} ±1.63
G-0M(B)	2.73 ^{bc} ±0.25	3.49 ^{ab} ±0.67	6.22 ^{ab} ±0.88
G-1M	3.31 ^a ±0.49	3.89 ^{ab} ±0.94	7.20 ^{ab} ±0.97
C-(B)	2.19 ^c ±0.11	3.73 ^{ab} ±1.17	5.92 ^b ±1.09
C-(M)	2.63 ^{bc} ±0.59	3.15 ^b ±0.42	5.78 ^b ±1.86

Table 4. Cumulative N₂O-N losses (kg ha⁻¹) over 204 days in 2009 in grass-clover treatments and cereal reference plots (treatments having no differences in 2009 were combined).

2009 combined		
Treatment	N ₂ O-N (kg ha ⁻¹)	St. dev. (no. of replicates)
G-3M	3.26 ^a	0.13(4)
G-0M,G-0M(B)	2.89 ^b	0.33(8)
G-1M	3.31 ^a	0.49(4)
C-(B), C-(M)	2.41 ^c	0.46(8)

operations prevented us from installing the flux measurement equipment. This means that cumulative fluxes given here fall short to estimate annual emissions as flux rates through snowpack (Maljanen et al., 2007; Sommerfeld et al., 1993) and presumably high emission rates directly after ploughing (Baggs et al., 2000) were not captured. We, therefore, limit the discussion of N₂O flux dynamics, N input and environmental factors (soil moisture, temperature and mineral N content) to the remainder of the seasons and compare growth season cumulative fluxes in 2010 to N yields.

Temporal dynamics of N₂O emissions during the year of GM establishment (2009) and the year of cereal production (2010) differed considerably (Figs. 1 and 2), indicating fundamental differences in N cycling between the two years and the two crops. In 2009, N₂O emission rates showed a pronounced seasonality with highest emissions during summer which was warmer and wetter than normal. N₂O emission rates were positively correlated to the seasonal course of soil temperature ($r = 0.29$, $P = 0.007$) and dropped upon an early cold period in October 2009. During the GM year, no fertiliser was applied and mineral N concentrations were generally low during the period of vigorous plant growth. Mulching of aboveground biomass had no measurable effect on mineral N contents in 0–20 cm. Despite the obvious competition by plants for mineral N, N₂O emissions were greatest during the period of plant growth. Mineral N con-

Table 5. Average dry matter yield (tons ha⁻¹), grain N yield (kg ha⁻¹) and N₂O emission intensity (g N₂O kg⁻¹ grain N yield) of barley during 2010. Values given are means of 4 replicate treatment plots ± standard deviation. Treatments not sharing the same letters are significantly different from each other at the 95 % confidence interval.

Treatment	DM yield* (t ha ⁻¹)	N yield* (kg ha ⁻¹)	N ₂ O-N/N yield (g N ₂ O-N kg N ⁻¹)
G-3M	2.54 ^c ±0.36	49.5 ^b ±8.5	95
G-0M	2.41 ^c ±0.25	44.2 ^b ±5.5	106
G-0M(B)	3.13 ^{ab} ±0.48	54.0 ^b ±10.1	65
G-1M	2.52 ^c ±0.49	48.1 ^b ±7.2	81
C-(B)	2.79 ^{bc} ±0.24	47.7 ^b ±5.9	78
C-(M)	3.56 ^a ±0.36	67.4 ^a ±11.6	47

* Data from R. Frøseth, unpublished.

tents markedly increased upon the onset of the first cold period in October, but then N₂O emission rates dropped to low values despite of high WFPS (Fig. 1). This suggests that N₂O emissions during summer were controlled by temperature and moisture dependent N mineralisation rather than by pool sizes of mineral N. As an exception, we observed a mineral N effect on N₂O emissions early during 2009 before and after the 1st ley harvest when N₂O emissions and mineral N contents were lower in the ley treatments than in the cereal reference (Fig. 1), apparently because the young cereal crop could not take up mineral N as efficiently as the well-established ley. This illustrates that grass-clover leys act as sink for mineral N during plant growth early in the year and, thereby, reduce N₂O emissions. Higher N₂O emissions from the grass-clover leys than the cereal reference were recorded after irrigation during the second growth period when WFPS values were high (60–80 %) and the soil was warm. Higher N₂O emissions in the grass-clover ley as compared to the cereal reference later during the growth period could be the result of net N release from mulched herbage and/or below-ground root turnover (Rochette and Janzen, 2005) and optimal conditions for simultaneous nitrification

and denitrification at WFPS values between 60 and 80% (Ciarlo et al., 2007; Dobbie and Smith, 2003). Overall, cumulative N₂O losses in 2009 showed a small but significant surplus of N₂O emission in ley treatments as compared to the non-fertilised cereal reference (Table 3). Legume plants have been discussed to directly contribute to N₂O emissions during the process of N-fixation in pure culture lab experiments (Bedmar et al., 2005), but more importantly have a potential to increase N₂O emissions by accumulating additional N in the root zone (Jensen and Hauggaard-Nielsen, 2003; Mayer et al., 2003). However, the present study was not designed to delineate direct or indirect legume associated N₂O emissions.

Mulching of grass-clover herbage (G-3M) resulted in a small but significant difference in seasonal N₂O emissions (+370 g N₂O-N) relative to non-mulching (cf. G-3M and G-0M/G-0M(B) in Table 4), likely reflecting the input of N from the decomposing mulch and/or more anaerobic conditions due to higher WFPS values in the mulched leys (Fig. 2). Flessa et al. (2002) found that 0.3% of the N applied in the form of a grass mulch was emitted as N₂O, whereas Larsson et al. (1998) found as much as 1.0% relative losses of applied N in alfalfa and N rich grass mulch. The N₂O surplus in our study (G-3M minus G-0M) corresponded to ca. 0.2% of the average harvested above-ground total N, suggesting that mulching itself was not a major source of N₂O during the year of GM production.

In 2010, N₂O emission dynamics showed no seasonality, but appeared to be strongly influenced by the two ploughing events, the first of which (end of April) incorporated the organic material from the GM or was followed by application of biogas residue or mineral N whereas the second (end of September) incorporated the barley residues while redistributing residual GM. Interestingly, we found significantly higher N₂O emissions in G-3M than G-0M before ploughing in early spring 2010. This may point towards an effect of mulching during the GM year on subsequent spring emissions with mulched leys (G-3M) contributing more decomposable organic matter to microbial N turnover during winter and spring thaw (Chantigny et al., 2002). Higher nitrogen release in G-3M during winter was also indicated by substantial N₂O accumulation in the thawing soil which was higher in G-3M than in G-0M or the cereal reference (Fig. 3). N₂O accumulation in soil under snow in spring 2011 showed no such clear effect of mulching (Fig. 4), probably because most of the easily available substrates in mulched biomass were decomposed during 2010.

Lengthy field operations by the farmer in spring 2010 prevented us from reinstalling flux measurement equipment and we are missing N₂O measurements directly after ploughing. When resuming measurements 5 weeks after ploughing, N₂O emissions were among the highest recorded in 2010, irrespective of treatment (Fig. 2). While high N₂O emission after ploughing arable soils seems to be short-lived (Kessavalou et al., 1998), contrasting effects on N₂O emis-

Table 6. Cumulative N yield (kg ha⁻¹) in harvested biomass during the study period in 2009 (GM-year). Values given are the mean of 4 replicates plots ± standard deviation. In 2009, G-0M, G-0M(B) and G-1M were treated identically until the last harvest; therefore, composite samples for N analysis were taken from these treatments. In 2010, only biogas residue (containing 60 kg NH₄⁺-N and 50 kg organic N ha⁻¹) and mineral fertiliser was applied in corresponding treatments and it is assumed that all N applied as GM mulch during the previous year to G-3M and G-1M was carried over to the cereal cropping year.

Treatment	2009 N yield (kg ha ⁻¹)	2010 N ha ⁻¹ application
G-3M	195.4±1.9	195.4
G-0M	177.1±2.8	–
G-0M(B)	177.1±2.8	110.0
G-1M	177.1±2.8	62.1
C-(B)	32.2±5.3	110.0
C-(M)	31.8±9.2	80.0

sions have been reported for incorporating GM in spring. Ball et al. (2007) found no immediate N₂O response to ploughing of grass-clover in January and February in Scotland, which they attributed to low temperatures (< 7 °C), whereas ploughing later during the year induced N₂O emissions similar to those observed after mineral fertiliser application. Davies et al. (2001) found a short-lived N₂O emission peak in a UK grass-clover ley ploughed under in June. Soil management (harrowing and sowing) in spring 2010 was delayed because of abundant rainfalls and low temperatures after ploughing (Fig. 2). These conditions (cold and wet soils) may have delayed N mineralisation from ploughed GM in our experiment. In contrast, addition of mineral fertiliser to treatment C-(M) on 12 May (i.e., 20 days after ploughing) may have induced direct N₂O emissions which were not captured in our study. Thus, cumulative N₂O emission in the minerally fertilised cereal reference may be an underestimation, rendering the N₂O budget of this treatment inherently uncertain. There was no significant treatment effect on N₂O emissions observed five weeks after ploughing, despite markedly higher NO₃⁻ contents in the minerally fertilised treatment (Fig. 2). This underpins the notion that N₂O emissions in organically fertilised soils can be as high as in minerally fertilised soils (Ball et al., 2007). For the remainder of the year, N₂O emission peaks appeared to be triggered by drying-rewetting. Also here no treatment effects were observed, suggesting that the perturbation effect exerted by drying-rewetting (Rudaz et al., 1991) overrode treatment related differences in organic matter quality (mulch vs. biogas residue vs. mineral N). This is plausible because drying-rewetting likely only affected the uppermost soil layer. As an exception, a treatment effect was observed during the rewetting event on 15 June, when N₂O emissions were significantly higher in previously mulched GM treatments than in

other treatments (Fig. 2a). N availability from the decomposing GM is in principal controlled by temperature with lower temperatures favouring immobilization of N during the early phase of decomposition (McKenney et al., 1993; Larsson et al., 1998). The relatively cold weather in 2010 with frequent dry-wet cycles may, thus, have prevented a stronger GM effect on N₂O emissions. Surprisingly, ploughing in mid-September 2010 (after harvest of barley) resulted in an N₂O emission peak similar in magnitude as observed 5 weeks after ploughing in spring. At that point, N₂O emissions were higher in previously mulched GM treatments, suggesting that some organic matter from green manure persisted throughout the growing season, fuelling N₂O production in autumn.

Overall, different GM management strategies during the previous year had no significant effect on cumulative N₂O emissions during barley production (Table 3). Despite the higher N yield in harvested grass-clover biomass of the G-3M treatment in 2009 (and therewith higher N input in 2010), there was no significant difference in N-yields of barley (2010) between the G-3M and G-0M treatments (Table 5). Substituting the aboveground biomass N removed in 2009 by biogas residue in 2010 (G-0M(B)) had a positive effect on barley DM and N yields, while slightly, but not significantly, reducing the N₂O emissions as compared with other GM treatments. Similar findings with increased DM yield and reduced N₂O emission (38 %) from digested biogas residue application (to the same crop) as compared to mulching and incorporation of GM were reported by Moeller and Stinner (2009). In contrast, biogas residue application to the non-fertilised cereal reference (C-(B)) did not lead to increased DM and N yields of barley as compared to the treatment with 80 kg mineral N alone (C-(M)), while N₂O emissions were slightly, but not significantly, increased. Apparently, the amount of N supplied with the biogas residue (60 kg NH₄⁺-N + 50 kg organic N ha⁻¹) was less sufficient in supporting crop yields than the mineral fertiliser applied (80 kg N ha⁻¹). Biogas residue application had no significant effect when comparing cumulative N₂O emissions in GM (G-0M(B) vs. G-0M) or cereal reference (C-(M) vs. C-(B); Table 3). Overall, GM fertilised barley had higher N₂O emissions as compared with the moderate inorganic fertiliser N treatment.

Organic farming has been reported to reduce NO₃⁻ leaching (Hansen et al., 2000). On the other hand, nitrogen release from organic fertilisers is not necessarily well-synchronized with plant uptake, which may lead to potentially significant N losses and N₂O emissions (Stinner et al., 2008; Berentsen et al., 1998). In our study, it was not possible to quantify off season losses because of high snow packs (ca. 1 m) in winter 2009/2010. However, there might have been substantial N₂O losses through the snow pack (Sommerfeld et al., 1993; Maljanen et al., 2007) caused by subnivean microbial activity and it is likely that these emissions would have been higher in mulched GM leys than in non-ploughed cereal reference plots, resulting in an underestimation of total N₂O losses

from mulched leys. Based on the N application (as mulch in 2009) in our experiment, 2.4 % of the mulched aboveground biomass total N was emitted as N₂O in 2010 (including background emissions). In the moderately fertilised (80 kg ha⁻¹) cereal reference C-(M), the fraction of applied N lost as N₂O was 3.9 %, which is substantially higher than the IPCC Tier 1 emission factor of 1 % (IPCC, 2007). The fraction of N₂O calculated for our cereal reference (3.9 %) may be an underestimation because we are missing emission data right after fertiliser application in spring 2010. The generally high emission factors found in our study indicate that this poorly-drained, clayey soil with a history of heavy trafficking and soil compaction is prone to high N₂O emission levels irrespective of fertilisation type.

In many agricultural systems, direct N₂O emissions from soils and CH₄ emissions from ruminants are the prime components of the GWP of food production, while CO₂ emissions from changing soil C stocks play a minor role (Janzen, 2005). The CH₄ uptake of arable soils is generally small relative to other greenhouse gases. Organic farming in general is considered to sustain soil C stocks through increased organic matter input (Mäder et al., 2002). This makes N₂O emission the most important component for “atmospheric warming” in stockless organic farming (followed by indirect N₂O emissions from volatilized NH₃ and leached NO₃⁻), pinpointing its importance when evaluating environmental friendliness of this cultivation system in terms of GWP. To arrive at a sound appraisal, we estimated N₂O emission for the year of cereal production and expressed it in terms of N₂O-N per unit commodity (here kg N yield in grain). Our data suggest that moderate mineral N application results in less N₂O emission per unit N yield of barley (47 g N₂O-N kg⁻¹ N yield) as compared with GM treatments (78–106 g N₂O-N kg⁻¹ N yield) (Table 5). Our calculation is based on cumulative N₂O emissions and N yields measured during the year of cereal production (2010) and neglects N₂O emissions related to ley establishment and management in 2008 and 2009, which would increase the difference in relative N₂O emissions because of the omission of one cereal harvest every second year in the organic system. However, in terms of CO₂ equivalents, the higher relative GHG footprint of our stockless production system would be somewhat reduced if taking into account the CO₂ and CH₄ saved by biogas production. Our results should not be generalised, since the experiment was not designed as a comparative study between two cultivations systems. The reference plots were established as a non-leguminous control with moderate inorganic N input for the purpose of parameterizing a soil-crop model and do not represent conventional farming practices in Norway, which operates with much higher mineral fertiliser amounts (120 kg ha⁻¹ yr⁻¹). Moreover, CO₂ eq costs of fertiliser production and transportation in conventional farming should be taken into account when evaluating the overall life cycle emissions of stockless organic farming.

5 Conclusions

Our data show that the use of GM may stimulate N₂O emissions in cereal production relative to moderate mineral fertilisation in a heavy clay soil. Mulching may have an additional stimulating effect on N₂O emissions particularly during warm and moist summers as observed in 2009. Harvesting of GM and replacement as biogas slurry may be one option to circumvent mulching induced N₂O emissions. However, addition of biogas slurry to continuous cereal cropping does not seem to supply enough nitrogen to the crop (as compared to mineral fertiliser) under the climatic conditions in SE Norway, so that yield-scaled N₂O emissions would increase.

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