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Discussions**

N$_2$O 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 N$_2$O 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 replacement as biogas residue to a subsequent barley crop. Grass-clover ley had small but significantly higher N$_2$O emissions as compared with a non-fertilized cereal reference during the year of green manure (GM) production in 2009. Mulching of herbage induced significantly more N$_2$O emission ($+0.37$ kg N$_2$O-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 N$_2$O emissions than during the previous year. Application of biogas residue (110 kg N ha^{-1}) before sowing did not increase emissions neither when applied to previous ley plots nor when applied to previously unfertilized cereal plots. Ley management (mulching vs. removing biomass in 2009) had no effect on N$_2$O emissions during barley production in 2010. In general, GM ley (mulched or harvested) increased N$_2$O emissions relative to a cereal reference with low mineral N fertilisation (80 kg N ha^{-1}). Organic cereal production emitted 95 g N$_2$O-N kg^{-1} N yield in barley grain, which was substantially higher than in the cereal reference treatment with 80 kg mineral N fertilization in 2010 (47 g N$_2$O-N kg^{-1} N yield in barley grain).

1 Introduction

Nitrous oxide (N$_2$O) is a potent green house gas involved in ozone depletion and global warming (Ravishankara et al., 2009). During the past 250 yr, the atmospheric N$_2$O concentration has increased by 15 % (Mosier, 1998). Agriculture is estimated to be responsible for 38 % of the N$_2$O emitted to the atmosphere (IPCC, 2007). Direct N$_2$O

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 N$_2$O budgets.

However, there is a broad consensus that cultivation history, crop rotation and local soil conditions modify fertiliser induced N$_2$O emissions and that management strategies have to be studied in a local context if regionalised emission factors were to be defined (Petersen et al., 2006). There is no conclusive evidence so far linking specific cultivation systems to specific emission factors, however. N$_2$O 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 N$_2$O 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 nutrient cycling, thus reducing the use of extraneous nitrogen while maximizing 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 main 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 NH_3 or NO_x. Flessa et al. (2002) found that 0.2–0.3 % of applied N in grass mulch was emitted as N_2O 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 N_2O during three months depending of the N content of the mulch.

However, N_2O 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 portion (6–25 %) of the green manure N can be taken up by the subsequent crop (Müller and Sundman, 1988). Ball et al. (2007) found that 75 % of the N_2O emission in an organic barley cropping system with a grazed grass-clover ley occurred shortly after ploughing. Excess mineral nitrogen (NO_3^-, NH_4^+) not readily taken up by the crop is a potential source for N_2O from denitrification or nitrification (Crutzen et al., 2008). Since fresh plant material also contains substantial amounts of readily degradable carbon, mulching may increase N_2O production by directly fuelling denitrification with carbon or by creating reductive zones by increased oxygen respiration (Flessa and Beese, 1995).

Increased N_2O 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 N_2O emissions could be to remove the aboveground biomass after mowing 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 NO_3^-

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leaching and N$_2$O emissions (Moeller and Stinner, 2009). Due to the lower C/N ratio of the organic fraction and the prevalence of NH$_4^+$ in the mineral fraction, biogas slurry may be less conductive to denitrification than fresh undigested biomass. Moreover, an increase of soil pH after biogas slurry application might decrease the N$_2$O/N$_2$ ratio of

5   denitrification as shown in long-term liming experiments by Liu et al. (2010). Nothing is known about how the management of a whole-year GM ley affects N$_2$O emissions after ploughing. N$_2$O 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 N$_2$O emissions.

10   In the present study we monitored N$_2$O 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 N$_2$O emissions for different GM management strategies. We specifically studied the effect of mulching versus removing above

15   ground herbage during the year of GM production and the effect of returning harvested GM as anaerobically digested biorest to the cereal crop. We also evaluated the effect of grass-clover GM itself relative to a moderately fertilized (80 kg N ha^{-1}) reference treatment (continuous cereal) without green manure. Finally, we related cumulative N$_2$O emissions to dry matter yield and N content in the cereal crop to facilitate an

20   assessment of the N$_2$O 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

25   average (1961–1990) precipitation is 785 mm. The normal (1961–1990) mean annual temperature is 5.3 °C (Thue-Hansen and Grimenes, 2009, 2010). The soil is a poorly

drained, silty clay loam of marine origin (Typic endoaqualf) with a WFPS of 83% at field capacity (Sveistrup et al., 1997). 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.

2.2 Treatments

The field experiment was one of four identical field trials at different locations in Norway designed to study the effect of different GM treatments on barley yields and nitrogen efficiency and to improve the prediction of crop N uptake. 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 vulgar*, 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 (unfertilized in 2008 and 2009 and moderately fertilized with 80 kg N ha^{-1} as NH$_4$NO$_3$ in NPK fertilizer 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 unfertilized cereal reference C-(B). The application of biorest added 110 kg total N ha^{-1} (containing 60 kg NH$_4$-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   N$_2$O flux sampling and calculation

Nitrous oxide (N$_2$O) 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 giving a total of 24 (6 treatments × 2 replicate plots × 2 replicate frames). N$_2$O 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 biweekly during the growing seasons of 2009 and 2010. High snow packs in winter 2009/10 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, Agilant, Santa Clara,

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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} ArCH$_4$ (10/90 vol %) as makeup gas. The GC was connected to an autosampler via a peristaltic pump (Gilson minipuls 3, Middleton, W1, 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 minimize memory effects. Details of the autosampler and injection system are given in Molstad et al. (2007). N$_2$O emission rates were calculated from the linear or quadratic change in chamber gas concentration using the equation

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

where F_{N_2O} is the emission flux of N$_2$O in µg N$_2$O-N m^{-2} h^{-1}, d_{N_2O} the rate of change in N$_2$O concentration (ppmv min^{-1}) in the chamber headspace, V_c the volume of chamber (litre), A the area covered by the chamber (m^2), M_n the molecular mass of N in N$_2$O (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 chamber temperature at the beginning and end of deployment. Cumulative N$_2$O emissions (kg N$_2$O-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 (NH$_4^+$ and NO$_3^-$) 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 ($100\,cm^3$) taken from 2–7 and 10–15 cm depth in the ley treatments and cereal reference plots in autumn 2009. Soil mineral N (NH_4^+, NO_3^-) was determined by extracting 45 g fresh soil with 50 ml 2 M KCL solution. Extracts were stored at –20 °C until analysed by a flow injection analyser (FIA Star 5010 analyser Tecator, Sweden) for NH_4^+ and NO_3^-. Soil pH was determined in spring 2010 in 1:5 w/w slurries with deionised water.

2.5 Field monitoring

Air and precipitation data were obtained from a nearby weather station located on the University campus in Ås (Thue-Hansen and 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 (ECH_2O 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 N_2O. SAPs were constructed from small porous cups (height 3.5 cm, volume $6\,cm^3$) made of sintered polyethylene (pore diameter 100 μm) 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-augured (3.2 cm diameter) holes in a 60° angel 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. CH_4, CO_2 and N_2O concentrations were analysed by GC as described above.

2.6   Statistical data analysis

Statistical analysis was performed by Minitab software version 15 (Minitab Inc., State College, PA, USA), using a general linear model and Tukey's test for pair wise comparison of cumulative N_2O flux between the treatments within each year. Regression analysis was performed to find correlations between N_2O flux and ancillary variables.

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 NH_4^+-N g dw soil^{-1} (Fig. 1b). NO_3^- concentrations were lower (<2 µg N g dw soil^{-1}), except for the cereal reference
5 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 excursions of NO_3^- content were observed after the 1st and 2nd cut in ley treatments, but comparison with NO_3^- dynamics in the cereal reference plots revealed that this could not be attributed directly to the cuts. NO_3^- concentrations showed an increase towards the end of the
10 vegetation period in all treatments, when soil temperatures dropped in early October.

In 2010, high NH_4^+ and NO_3^- contents were observed 5 weeks after ploughing (Fig. 2b, c). At that point, NH_4^- was highest in treatments in which green manure had been ploughed under either as mulched grass-clover ley or as biogas residue. NO_3^- showed the opposite trend with higher concentration in the cereal reference treatment
15 due to mineral fertilizer application. Differences in mineral N 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 NH_4^+ concentrations were somewhat more erratic. In general, mineral N values were higher in 2010 than in 2009.

20 ## 3.3 N$_2$O emission patterns

3.3.1 Green manure ley (2009)

In 2009, N$_2$O 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 N$_2$O
25 emissions were in the range of 4–188 µg N$_2$O-N m^{-2} h^{-1} with peak emissions in June–August and lowest emissions in October. The N$_2$O emissions peaked after cuts in the

ley treatments, after irrigation (in June after drought) and after ample rainfall in July and early August. In the cereal reference treatment (oats), N_2O fluxes ranged from 4.8 to $171 \, \mu g \, N \, m^{-2} \, h^{-1}$ showing two peaks, 27 June ($171 \, \mu g \, N_2O$-N $m^{-2} \, h^{-1}$) and 19 August ($76 \, \mu g \, N_2O$-N $m^{-2} \, h^{-1}$) which were associated with irrigation and rainfall events, respectively. N_2O peak emission after irrigation and after the 2nd cut was higher in G-3M than in other GM treatments and the cereal reference, while no such difference was seen after the 3rd cut. A sudden drop in N_2O fluxes at the end of September was observed upon a $8 \, ^{\circ}C$ drop in temperature (14 to $6 \, ^{\circ}C$) and a drop in WFPS from 65 to 47 % WFPS.

3.3.2 Barley (2010)

After spring thaw 2010, N_2O emissions were highest in GM mulched treatment and lowest in GM removed (Fig. 2a). Elevated N_2O emissions were roughly proportional to N_2O accumulation in the soil atmosphere (Fig. 3a). After ploughing, N_2O emission showed an event-driven emission pattern characterized 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 N_2O concentrations were still high (Figs. 2b, c and 3). Peaking N_2O emissions in June were observed upon rapid increase in WFPS in all treatments resulting in fluxes around $200 \, \mu g \, N_2O$-N $m^{-2} \, h^{-1}$ without showing any build up of N_2O in the soil atmosphere (Fig. 3). A second N_2O emission peak associated with drying-rewetting was recorded on 12 August at substantially higher WFPS values of around 80 %, resulting in a marked accumulation of N_2O in the soil atmosphere (Fig. 3). Finally, all treatments responded with elevated N_2O emissions to autumn ploughing on 15 September with GM treatments showing higher emission rates than cereal reference plots. Emission rates levelled off to low values ($<30 \, \mu g \, N_2O$-N $m^{-2} \, h^{-1}$) 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 different fluxes as compared to their counterparts C-(M) and G-1M, respectively. To 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 biorest in spring 2010. In general, N_2O fluxes were higher in 2010 than in 2009. In spring 2011, one year after the incorporation of GM and after ploughing under barley residues in the previous autumn, we observed high N_2O fluxes (150–393 $\mu g\,N_2O$-N m^{-2} h^{-1}) in early spring after snow melt (Fig. 4). Soil air N_2O 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 N_2O losses

Cumulative N_2O losses for the measurement periods in 2009 and 2010 (204 and 218 days, respectively) are given in Table 3. In 2009 some treatments were identical (as there was no application of biogas residue). Combining these treatments, cumulative N_2O-N losses where 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 N_2O losses were higher than in 2009, ranging from 3.15 to 4.70 kg N_2O-N ha^{-1} 218 d^{-1}. The highest N_2O losses were observed in the G-3M and G-0M treatments (4.70 and 4.67 kg N_2O-N ha^{-1} 218 d^{-1}, respectively), both of which were significantly higher than in C-(M) with mineral fertilizer. 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 N_2O-N loss (3.15 kg N_2O-N ha^{-1} 218 d^{-1}) was observed in the cereal reference with mineral fertilization (80 kg N ha^{-1}; C-(M)). In this treatment, 3.9 % of the applied N was lost as N_2O (including background emission), whereas 2.4 % of the N applied as mulched aboveground biomass in 2009 was lost in the G-3M treatment.

3.5 Emission intensity

Dry matter and N yield of barley in 2010 were statistically 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 N_2O-N loss.
5 Together, this resulted in clearly higher relative N_2O loss (g N_2O-N kg^{-1} N yield) for organic treatments than for the mineral reference treatment (Table 5).

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
10 losses can be minimized. In the present study we evaluated different GM management strategies (mulching versus removing herbage and replacing it as biogas residue) with respect to N_2O emissions throughout a two year crop rotation consisting of a full-year grass-clover ley followed by a year with cereal production. N_2O emission fluxes were measured during the snow-free period only and there were no flux measurements dur-
15 ing several weeks after ploughing the GM in the second year (2010) because lengthy agronomic field 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
20 not captured. We therefore limit the discussion of N_2O flux dynamics, N input and environmental factors (soil moisture, temperature and mineral N content) to the remainder of the years and compare growth season cumulative fluxes in 2010 to N yields.

   Temporal dynamics of N_2O emissions during the year of GM establishment (2009) and the year of cereal production (2010) differed considerably (Figs. 1 and 2), indicat-
25 ing fundamental differences in N cycling between the two years and the two crops. In

2009, N$_2$O emission rates showed a pronounced seasonality with highest emissions during summer which was warmer and wetter than normal. N$_2$O emission rates appeared to follow the seasonal course of soil temperature 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, N$_2$O emissions were greatest during the period of plant growth. Mineral N contents markedly increased upon the onset of the first cold period in October, but then N$_2$O emission rates dropped to low values despite of high WFPS (Fig. 1), resulting in a counterintuitive negative relationship between mineral N and N$_2$O emission rates over the entire measurement period in 2009. Together this suggests that N$_2$O emissions were controlled by nitrogen turnover rather than pool sizes of mineral N. As an exception, we observed a mineral N effect on N$_2$O emissions early during 2009 before and after the 1st ley harvest when N$_2$O 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 can be a sink for mineral N during aggressive plant growth early in the year and thereby reduce N$_2$O emissions. Higher N$_2$O 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. WFPS between 60 and 80 % allows for simultaneous nitrification and denitrification (Ciarlo et al., 2007; Dobbie and Smith, 2003), the extent of which was likely greater in the grass-clover ley fuelled by net N release from mulched herbage and/or below-ground root turnover as concluded by Rochette and Janzen (2005). At above 90 % WFPS, N$_2$O emissions appeared to be lower than at 60–80 % WFPS, probably because of restricted diffusion of N$_2$O through the soil, resulting in more complete denitrification (lower N$_2$O/N$_2$ ratio) as suggested by Ciarlo et al. (2007). Overall, cumulative N$_2$O losses in 2009 showed a small but significant surplus of N$_2$O emission in ley treatments as compared to the non

fertilized cereal reference (Table 3). Legume plants have been discussed to directly contribute to N_2O emissions during the process of N-fixation in pure culture lab experiments (Bedmar et al., 2005) but more certainly are a source for additional mineral N through root exudates (Jensen and Hauggaard-Nielsen, 2003; Mayer et al., 2003). The
5 present study was not designed to delineate direct or indirect legume associated N_2O emissions, however.

Mulching of grass-clover herbage resulted in a small, non-significant surplus of 370 g N_2O-N in 2009 as compared with harvesting (Table 3), likely reflecting the overall higher N input in mulched leys. Flessa et al. (2002) found that 0.3 % of applied N in the
10 form of a grass mulch was emitted as N_2O whereas Larsson et al. (1998) found much higher (1.0 %) relative losses. The N_2O surplus in our study (G-3M minus G-0M) corresponded to ca. 0.2 % of the average harvested above-ground biomass N, suggesting that mulching did not contribute strongly to N_2O emissions in our study.

In 2010, N_2O emission dynamics showed no seasonality, but appeared to be strongly
15 influenced by the two ploughing events, the first of which (end of April) incorporated the GM and the second (end of September) the barley residues while redistributing residual GM. Interestingly, we found higher N_2O emissions in G-3M than G-0M during two measurement dates before ploughing in spring 2010. This may point towards an effect of mulching during the GM year on subsequent spring emissions with mulched leys
20 (G-3M) contributing more decomposable organic matter to microbial N turnover during winter and spring thaw (Chantigny et al., 2002). Higher nutrient release in G-3M during winter was also indicated by substantial N_2O accumulation in the thawing soil which was higher in G-3M than in G-0M or the cereal reference (Fig. 3). N_2O accumulation in soil under snow in spring 2011 showed no such clear effect of mulching
25 (Fig. 4), probably because most of the easily available substrates in mulched biomass were decomposed during 2010. Incorporation of crop residues by ploughing resulted in overall higher cumulative N_2O emissions than measured during the previous year in the GM ley, which is in accordance with findings of Kaiser et al. (1998). Ploughing increases the decomposability of crop residues (including root material and mulched

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aboveground biomass) and alters the gas diffusivity due to the physical disturbance of the soil structure (Baggs et al., 2000). Elevated N_2O emissions after ploughing of crop residues have been reported previously but this effect tends to be short-lived and depends on the quality and C:N ratio of the crop residues (Aulakh et al., 2001; Kaiser et al., 1998), as well as on soil moisture and temperature after ploughing (Stott et al., 1986). In the present study we were not able to capture N_2O emissions directly after ploughing GM or applying biogas residue or mineral N because of lengthy spring cultivation by the farmer. However, there was no clear treatment effect on the magnitude of N_2O emissions observed five weeks after ploughing, suggesting that all treatments responded with elevated N_2O emission to the incorporation of fresh organic matter (GM or stubble). The same was true for N_2O emission peaks triggered by drying-rewetting during summer 2010 which were very similar in magnitude among the treatments. Obviously, the perturbation effect exerted by drying-rewetting (Rudaz et al., 1991) overrode treatment related differences in organic matter quality (mulch vs. biorest vs. mineral N), which is plausible if only the uppermost soil layer is affected by drying-rewetting. An effect of GM incorporation became first obvious 7 weeks after ploughing when N_2O emissions were slightly higher in GM treatments than in the minerally fertilized cereal reference (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 N_2O emissions. N_2O emission response to ploughing in mid September 2010 (after harvest of barley) was somewhat lower than to spring ploughing, likely because much of the fresh organic matter added by spring ploughing had been mineralised and/or immobilised during summer. Overall, different GM management strategies during the previous year had no significant effect on cumulative N_2O emissions during barley production (Table 3). Despite the higher N yield in harvested biomass of G-3M treatments 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 reducing the N$_2$O emissions as compared with other GM treatments. Similar findings with increased DM yield and reduced N$_2$O emission (38 %) from digested biogas residues application (to the same crop) as compared to mulching and incorporation of GM were reported by Moeller and Stinner (2009). However, biogas residue application to the non-fertilized cereal reference did not manage to increase the DM and N yields of barley as compared to 80 kg N applied as compound fertilizer, while N$_2$O emissions were increased relative to the minerally fertilized cereal reference. Little is known about how the N status of a soil affects the mineralisation kinetics of biogas residues, but our data suggest that the net effect on N$_2$O emissions depends on the N status and possibly also C status in terms of readily decomposable crop residues. Overall, GM fertilized barley had higher N$_2$O emissions as compared with the moderate inorganic fertilizer N treatment.

Organic farming has been reported to reduce NO$_3^-$ leaching (Hansen et al., 2000). On the other hand, nutrient release from organic fertilizers is not necessarily well-synchronized with plant uptake, which may lead to potentially significant N losses and N$_2$O emissions (Stinner et al., 2008; Berentsen et al., 1998). For example, Thomsen et al. (1993) observed high off-season N$_2$O emissions after the last cut in a GM ley. 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 N$_2$O 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 N$_2$O losses from mulched leys. Based on the N application (as mulch in 2009) in our experiment, 2.4 % of the mulched aboveground biomass N was emitted as N$_2$O in 2010 (including background emissions). However, in the moderately fertilized (80 kg ha^{-1}) cereal reference C-(M), the fraction of applied N lost as N$_2$O was even higher (3.9 %), which is substantially higher than the IPCC Tier 1 emission factor

of 1 % (IPCC, 2007). These results indicate that this poorly-drained, clayey soil with a history of heavy trafficking and soil compaction supported high N$_2$O emission levels irrespective of fertilisation type.

In many agricultural systems, direct N$_2$O emissions from soils and CH$_4$ emissions from ruminants are the prime components of the GWP of food production, while CO$_2$ emissions from changing soil C stocks play a minor role (Janzen, 2005). In stockless organic farming, CH$_4$ uptake by soil may exhibit a small cooling effect and short-term CH$_4$ emissions from wet mulch can be offset by fermenting above ground herbage. Organic farming in general is considered to sustain soil C stocks through increased organic matter input (Mäder et al., 2002). This makes N$_2$O emission the most important component for "atmospheric warming" (followed by indirect N$_2$O emissions from volatilized NH$_3$ and leached NO$_3^-$) in stockless organic farming, pinpointing its importance when evaluating environmental friendliness of this cultivation system in terms of GWP. To arrive at a sound appraisal, we estimated N$_2$O emission for the year of cereal production and expressed it in terms of N$_2$O-N per unit commodity (here kg N yield in grain). Our data suggest that moderate mineral N application results in less N$_2$O emission per unit N yield of barley (47 g N$_2$O-N kg^{-1} N yield) as compared with GM treatments (78–106 g N$_2$O-N kg^{-1} N yield) (Table 5). Our calculation is based on the cumulative N$_2$O emissions and N yields in cereals during 2010 and neglects N$_2$O emissions related to ley establishment and management in 2008 and 2009, respectively, which would increase the difference in relative N$_2$O emissions because of the omission of one cereal harvest every second year in the organic system. However, in terms of CO$_2$ equivalents, the higher relative GHG footprint of our stockless production system would be somewhat reduced if taking into account the CO$_2$ and CH$_4$ saved by biogas production. Our results should not be generalized, 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 modelling purposes and do not represent conventional farming praxis in Norway, which operates with much higher mineral fertilizer amounts (120 kg ha^{-1} yr^{-1}). Moreover, CO$_2$ costs

of fertilizer production and transportation in conventional farming should be taken into account when evaluating the life cycle of stockless organic farming.

5 Conclusions

In summary our data show that the use of GM may stimulate N_2O emissions in cereal production relative to moderate mineral fertilisation in a heavy soil like used in our study. The common practice of mulching may be partly responsible for this 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 N_2O emissions whereas addition of biogas slurry to continuous cereal cropping does not seem to supply enough nutrients to the crop (as compared to mineral fertilizer), thereby increasing N_2O emissions per unit N in grain.

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Table 1. Chemical and physical properties of the top soil at Østrevoll defined as naturally poorly drained silty clay loam. Standard deviations in parentheses when 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/100g)[*]	5.4
K-AL (mg/100g)[*]	9.4
Ca (mg/100g)[*]	211.0
Na (mg/100g)[*]	2.0

[*] From Bakken et al. (2006).

Table 2. Treatments in the 3-yr crop rotation. N$_2$O 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. Biorest and mineral fertilizer was applied on 12 May 2010.

Treatment	2008	2009	2010
G-3M	Barley w/GM	GM 3 cuts mulched	Barley
G-0M	Barley w/GM	GM 3 cuts removed	Barley
G-0M(B)	Barley w/GM	GM 3 cuts removed	Barley + biorest (110 kg N ha^{-1})
G-1M	Barley w/GM	GM 2 cuts removed, last mulched	Barley
C-(B)	Barley	Oats	Barley + biorest (110 kg N ha^{-1})
C-(M)	Barley	Oats	Barley + fertilizer (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 biorest application in 2010.
C-(M) = cereal reference with mineral fertilizer application in 2010.

Table 3. Cumulative N$_2$O losses (N$_2$O-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 (a, b and c) 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 N$_2$O-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). Treatments not sharing the same letters (a, b and c) are significantly different from each other at the 95 % confidence interval.

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(8)
C-(B), C-(M)	2.41[c]	0.46(8)

Table 5. Average dry matter yield (tons ha^{-1}), grain N yield (kg ha^{-1}) and N$_2$O emission intensity (g N$_2$O 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	49.47^b ± 8.5	95
G-0M	2.41^c ± 0.25	44.22^b ± 5.5	106
G-0M(B)	3.13^{ab} ± 0.48	53.99^b ± 10.1	65
G-1M	2.52^c ± 0.49	48.07^b ± 7.2	81
C-(B)	2.79^{bc} ± 0.24	47.72^b ± 5.9	78
C-(M)	3.56^a ± 0.36	67.38^a ± 11.6	47

[*] Data from R. Frøseth (unpublished data).
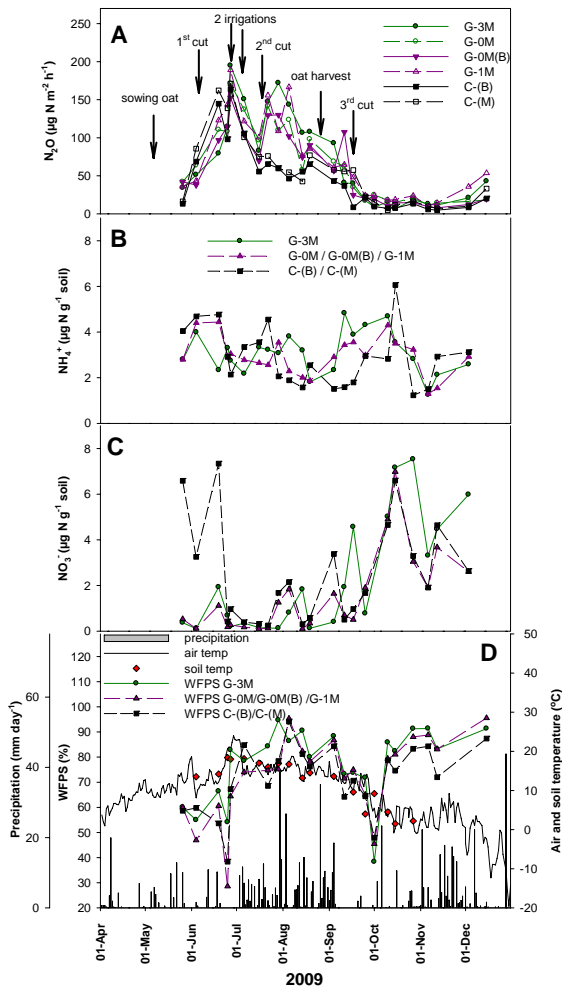
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Fig. 1. (A) Average N$_2$O fluxes (μg N m^{-2} h^{-1}) during the year of GM management (2009). Standard deviations are omitted to increase readability. **(B)** Soil NH$_4^+$ (μg N g^{-1} soil). **(C)** Soil NO$_3^-$ (μg N g^{-1} soil). Mineral N concentrations 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.
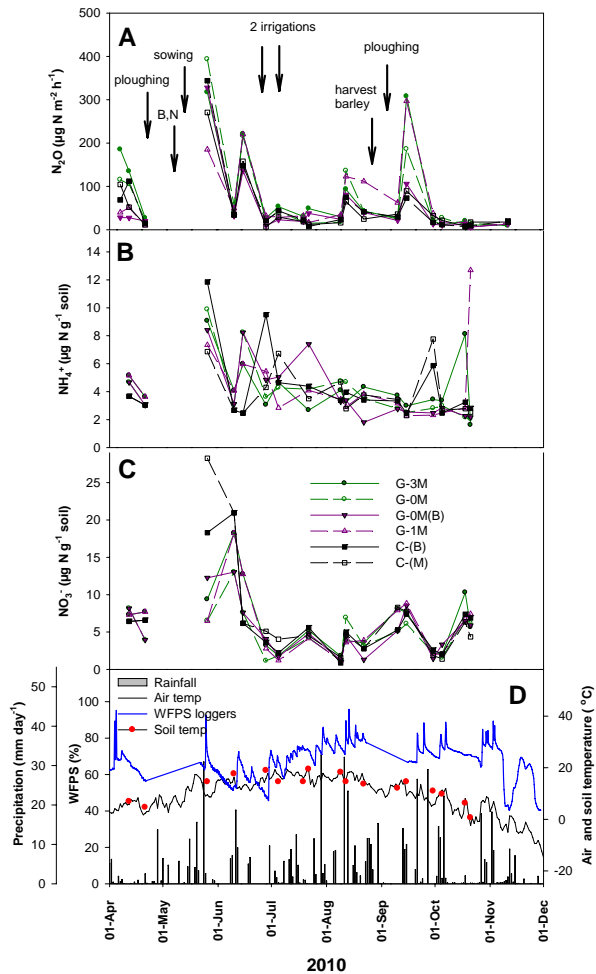
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Fig. 2. (A) Average N$_2$O fluxes (μg N m^{-2} h^{-1}) during the year of barley production (2010). Standard deviations are omitted to increase readability. **(B)** Soil NH$_4^+$ (μg N g^{-1} soil). **(C)** Soil NO$_3^-$ (μg N g^{-1} soil). Mineral N concentrations are from one composite sample (0–20 cm) per treatment on each sampling date. **(D)** Daily precipitation, air temperature 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 = biorest, N = mineral N application.

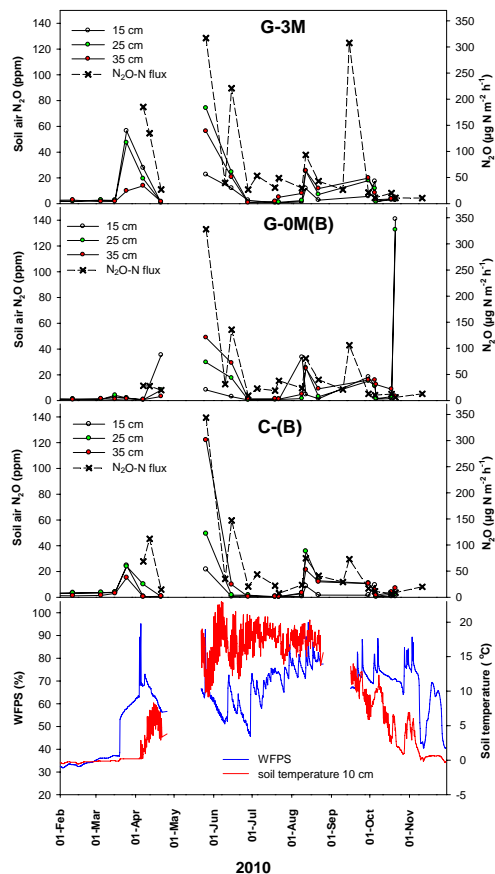**Fig. 3.** N$_2$O concentration (ppmv) in soil atmosphere at three different depths (left y-axis) and N$_2$O emission rates (right y-axis) as affected by different green manure management strategies. The lowest panel shows the % WFPS (left y-axis) and soil temperature (right y-axis) in 10 cm depth throughout the study period in 2010.
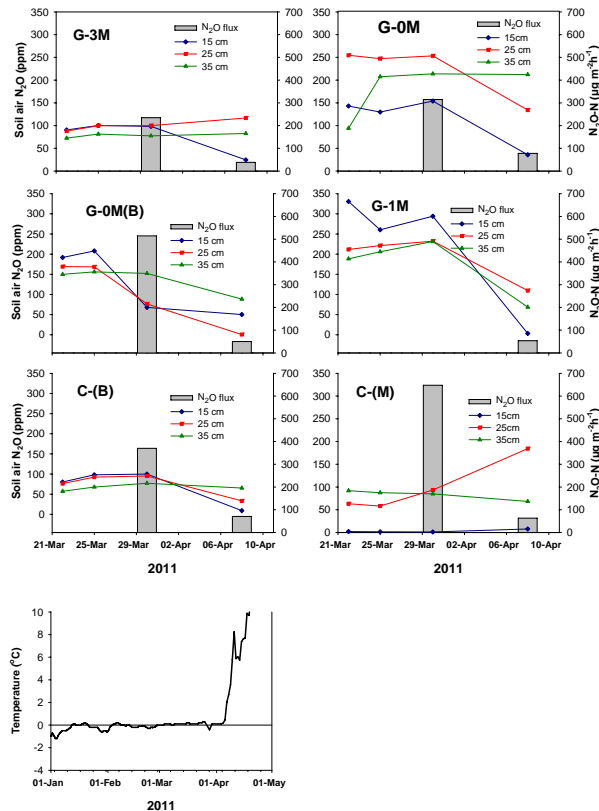
Fig. 4. N_2O concentration (ppmv) in soil air atmosphere at three different depths (left y-axis) and N_2O emission rates (right y-axis) during snow melt in 2011, as affected by different green manure management strategies in 2009 and 2010. Soil air N_2O concentrations are average values from two replicate sets of soil air probes while N_2O emission rates are the average of four replicates. Soil temperature is calculated from weighted mean of 5 and 20 cm depth soil temperature readings by loggers.