Application Technique and Slurry Co-Fermentation Effects on Ammonia, Nitrous Oxide, and Methane Emissions after Spreading: II. Greenhouse Gas Emissions

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ABSTRACT

The aim of this study was to investigate the effect of different application techniques on greenhouse gas emission from co-fermented slurry. Ammonia (NH_3), nitrous oxide (N_2O), and methane (CH_4) emissions were measured in two field experiments with four different application techniques on arable and grassland sites. To gather information about fermentation effects, unfermented slurry was also tested, but with trail hose application only. Co-fermented slurry was applied in April at a rate of 30 m^3 ha^{-1}. Measurements were made every 4 h on the first day after application and were continued for 6 wk with gradually decreasing sampling frequency. Methane emissions were <150 g C ha^{-1} from co-fermentation products and seemed to result from dissolved CH_4. Only in the grassland experiment were emissions from unfermented slurry significantly higher, with wetter weather conditions probably promoting CH_4 production. Nitrous oxide emission was significantly increased by injection on arable and grassland sites two- and threefold, respectively. Ammonia emissions were smallest after injection or trail shoe application and are discussed in the preceding paper. We evaluated the climatic relevance of the measured gas emissions from the different application techniques based on the comparison of CO_2 equivalents. It was evident that NH_3 emission reduction, which can be achieved by injection, is at least compensated by increased N_2O emissions. Our results indicate that on arable land, trail hose application with immediate shallow incorporation, and on grassland, trail shoe application, bear the smallest risks of high greenhouse gas emissions when fertilizing with co-fermented slurry.

In addition to slurry, organic wastes from household and food processing industries are increasingly used as fertilizers in agricultural systems. Of increasing relevance in this context is the combined anaerobic fermentation of organic wastes with slurry in biogas plants. The addition of organic wastes increases the profitability of biogas plants, as CH_4 production can be increased and farmers get paid for waste recycling (Kuhn, 1998). At the same time, nutrient input into the farming systems is increased, and care should be taken that these nutrients are used efficiently and detrimental environmental effects are minimized. Among these are gaseous emissions of ammonia (NH_3), nitrous oxide (N_2O), and methane (CH_4), which directly or indirectly contribute to the greenhouse effect. Various studies have already shown that for slurry the best way of reducing NH_3 volatilization is to reduce the air contact of the slurry by incorporation or injection (van der Molen et al., 1990; Mannheim et al., 1995; Rubaek et al., 1996; Malgeryd, 1998; Reitz et al., 1999). Although fermentation products differ in their physical and chemical properties from slurry (higher pH, higher NH_4^+ content, lower dry matter content), this is also the case for co-fermentation products (Wulf et al., 2002).

Greater contact of slurry with soil, for example, after incorporation, can on the other hand induce conditions favorable for N_2O or CH_4 emissions. Petersen et al. (1996) showed in laboratory studies that anoxic conditions are produced at the soil–slurry interface most probably through mineralization of carbon abundant in slurry. In laboratory experiments, Flessa and Beese (2000) found that N_2O emissions after injection into 10-cm soil depth were increased 13-fold compared with surface-applied slurry, and also CH_4 emissions were increased. On the other hand, Dendooven et al. (1998) and Sommer et al. (1996) could not find evidence for increased CH_4 or N_2O emissions after injecting slurry 5 to 10 cm deep into the soil in incubation experiments. Also, the results of the few field experiments conducted so far are contradictory. Weslien et al. (1998) reports slightly, but not significantly higher emissions after band-spreading followed by harrowing compared with band-spreading, trenching, and shallow injection (6 cm), whereas Ellis et al. (1998) and Dosch and Gutser (1996) conducted experiments on grassland and arable land, respectively, and found denitrification N losses after injection (10–15 cm) to be higher than after trail hose application. Nitrous oxide was not measured separately without acetylene inhibition in these experiments, but higher total N losses indicate that also higher N_2O losses might have taken place. On the other hand, no effects of application technique on N_2O emissions are reported by Velthof et al. (1997) for grassland and Clemens et al. (1997) for arable land with injection depths of 5 and 10 cm. All these studies were conducted with unfermented cattle slurries, some also with separated and thus C-reduced slurries (Dosch and Gutser, 1996; Clemens et al., 1997). Only a few studies include trace gas emissions after application of fermented slurries. Rubaek et al. (1996) studied NH_3 and denitrification N losses from fermented and unfermented slurries after trail hose application and injection. Petersen (1999) measured N_2O emissions after surface application of unfermented and co-fermented slurry. In both studies, N_2O emissions tend to be reduced when fermented substrates are applied. However, the effect of various application techniques for fermented substrates on the emissions of NH_3, CH_4, and N_2O has not been studied in the same field experiment yet.

Due to the increasing production of fermented slurry, our objective was to evaluate application techniques for co-fermentation products on arable and grassland sites, and to compare co-fermented and unfermented slurry after trail hose application. Fermented substrates differ from slurry in some of their chemical and physical parameters that might influence greenhouse gas emissions.

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Published in J. Environ. Qual. 31:1795–1801 (2002).

Among these are an increased NH_4^+ content as well as a reduced viscosity and carbon and dry matter content, which might make them comparable with separated slurries. As the agricultural use of fermentation products is increasing in importance, we want to provide information on effective and environmentally reasonable spreading techniques. To compare the emissions of the different trace gases including NH_3, we used CO_2 equivalents.

MATERIALS AND METHODS

Organic Fertilizer

The digested slurry used was a co-fermentation product. This substrate was produced through combined anaerobic fermentation of 70% dairy cow slurry together with 30% organic household waste (per weight). Mean fermentation duration was approximately 40 d under mesophilic temperature conditions (42°C).

Unfermented slurry was used as a reference with trail hose application technique only. The slurry was taken from the fermenter input material. Both slurries were stored for 90 d before field application. Slurry and the fermentation products differed in their chemical parameters (Table 1). During fermentation, methanogenic microorganisms producing CO_2 and CH_4 digest organic compounds. Nitrogen from this organic pool is mineralized to inorganic nitrogen during this process. So the proportion of NH_4^+–N of the total nitrogen increases and constituents that can be oxidized by chemical (COD) or biological processes (BOD_5) as well as dry matter content decrease.

Site Description and Experimental Setup

The experiments were conducted in the western part of Germany near the city of Bonn. The grassland experiment was started on 7 Apr. 1999 on a poorly drained Stagno–Gleyic Luvisol (FAO classification) with rather wet and cool weather at the beginning of the investigation. The trial on arable land was started three weeks later on a well-drained Luvisol (FAO classification) when warmer and drier weather prevailed. The C and N contents of the grassland topsoil (0–0.15 m) were 1.9 and 0.19% respectively, with soluble organic carbon (extracted with 0.05 *M* K_2SO_4) of 136 mg kg^{-1} dry soil. The contents of the arable land topsoil were 0.9 and 0.12% for C and N and 38 mg kg^{-1} for soluble organic carbon.

For both experiments, a completely randomized design with four replications for each treatment was chosen. Co-fermented and unfermented slurry were applied to plots of 9 m^2 with an application rate of 114 kg N ha^{-1}, corresponding to 30 m^3 and 67 kg NH_4^+–N ha^{-1} for co-ferment and 27 m^3 and 43 kg NH_4^+–N ha^{-1} for slurry. The number of emission measurements per day was gradually reduced from six to four during the first 4 d. Measurements were then performed daily for two weeks, later once per week until 6 wk after application.

Table 1. Chemical and physical properties of the co-fermented slurry and unfermented cattle slurry used in the experiments.

	Co-ferment	Slurry
Dry matter, g kg^{-1}	4.8	8.1
Total N, g kg^{-1}	3.8	4.3
NH_4^+–N, g kg^{-1}	2.2	1.6
COD†, g O_2 kg^{-1}	41	111
BOD_5‡, g O_2 kg^{-1}	3	21
pH	8.9	7.6

† Constituents that can be oxidized by chemical processes.
‡ Constituents that can be oxidized by biological processes.

Application Techniques

The co-fermentation product was spread with four different application techniques on each site. For injection a tractor-drawn device was used, which applied the substrate into 10-cm-deep V-shaped slots with injection tines being 30 cm apart. Due to the small plots (9 m^2), the other application techniques where simulated by hand. Defined amounts of substrate corresponding to 30 m^3 ha^{-1} were added to the plots with watering cans modified to simulate splash plates, trail hoses, and trail shoes, with slurry bands being 30 cm apart. On arable land one of the trail hose treatments was immediately followed by simulated shallow incorporation with a garden harrow. Application of substrate on a single plot took about 10 min and gas flux measurement started immediately thereafter. The substrates were applied to all plots within 90 min.

Sampling and Analytical Methods

Nitrous oxide and CH_4 emissions were determined with one closed chamber per plot (Hutchinson and Mosier, 1981) covering a surface area of 0.25 m^2 with a volume of 96 dm^3. Gas samples were taken from the chambers with evacuated headspace vials (0.02 dm^3) through a butyl septum. During the first week, samples were taken 0, 30, 60, and 90 min after placing the chambers airtight onto installation rings that were permanently inserted 10 cm deep into the soil surface. In the following weeks the sampling intervals were 0, 45, 90, and 135 min, because emission rates were expected to decrease. Longer accumulation periods resulted in higher gas concentration in the samples and a higher accuracy in analysis. Gas analysis was performed with a gas chromatograph (SRI [Torrance, CA] 8610C) with a backflush system to eliminate water vapor, electron capture detector (ECD) for N_2O, and flame ionization detector (FID) for CH_4. The gas chromatograph was operated at a column temperature of 40°C, an ECD temperature of 320°C, and gas flow rates (nitrogen 5.0) set at 35 mL min^{-1} for carrier gas and 6 mL min^{-1} for the ECD makeup gas. Because chamber size and closing time were chosen in such a way that no saturation effects in the headspace occurred, emission rates were calculated from the concentration increase in the chambers by linear regression.

Calculation of Carbon Dioxide Equivalents

To consider the treatment effects on greenhouse gas emissions as a whole, emissions were converted to CO_2 equivalents with Intergovernmental Panel on Climate Change conversion factors (Houghton et al., 1996). These factors are calculated from absorption efficiency of long-wave radiation and mean residence time in the atmosphere. They amount to 310 g g^{-1} for N_2O and 21 g g^{-1} for CH_4. Ammonia is not considered a direct greenhouse gas because of its short lifetime in the atmosphere, but its deposition induces N_2O formation elsewhere. It is postulated that 1% of deposited NH_3–N is reemitted as N_2O-N (Intergovernmental Panel on Climate Change, 1997). The data on NH_3 emissions from the different treatments are taken from Wulf et al. (2002), which were measured in the same experiments. To compare fertilizer-induced emissions only, emissions from the control plots (no fertilization) were subtracted from overall emissions.

Statistical Analysis

Results are presented as arithmetic means of four replicates. Although emission measurements are often not normally distributed, this estimator was preferred to geometric or lognormal means, because according to Velthof and Oenema (1995), it is less biased and more robust than any other estimator for small numbers of replicates. For comparison of treatments, nonparametric tests were used. Kruskall–Wallis followed by Mann–Whitney *U* tests were performed with SPSS (2000) software.

RESULTS AND DISCUSSION

Methane

Highest emission rates took place immediately after spreading on both sites and were hardly detectable during the following measurements for the most treatments (Fig. 1). But, after injection and application of unfermented slurry, emissions continued for 24 h on arable land and for 4 d on the grassland site. There are two possible sources of CH_4 emissions. Emissions that occur immediately after application can be assigned to the release of dissolved CH_4 being produced prior to application during storage of the substrate, as already postulated by Sommer et al. (1996) and Chadwick et al. (2000). In the arable land experiment this seems to be the dominating process, with some of the CH_4 being physically entrapped in the soil after injection, which might retard the emission process. The second possible source is via methanogenesis in the soil, as anaerobic conditions might be promoted through this application technique causing CH_4 production (van den Pol-van Dasselaar et al., 1999; Flessa and Beese, 2000), resulting in higher overall emissions compared with surface-applied substrate (Fig. 2a).

On grassland, emissions after injection of co-fermented slurry continued for up to 4 d after application, resulting in highest cumulated emissions, significantly higher ($p < 0.05$) only compared with splash plate application (Fig. 2b). This continued emission over a longer period might also result from the degradation of volatile fatty acids (VFA) by methanogene bacteria. Kirchmann and Lundvall (1993) and Sommer et al. (1996) observed a rapid decrease of VFA content of the soil within 24 to 48 h after application of slurry to soil in laboratory experiments, which coincided with the decrease of CH_4 emissions. Prolonged emissions in our experiment might be explained by especially high soil water contents prevailing during the first weeks of the grassland experiment, promoting anaerobic conditions especially in the injection slots.

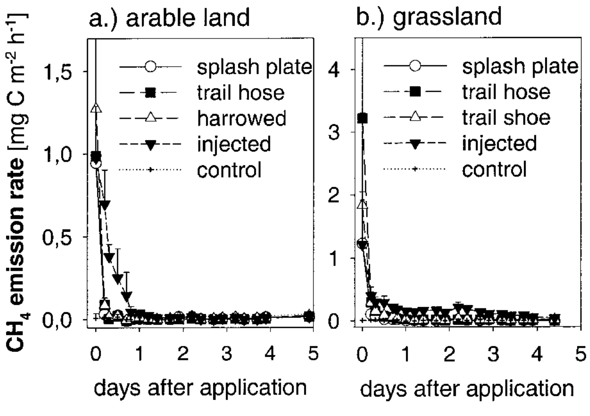

Fig. 1. Methane emission rates after spreading co-fermented slurry with different application techniques on arable (*a*) and grassland (*b*) sites. Data represent means and standard deviation (n = 4).

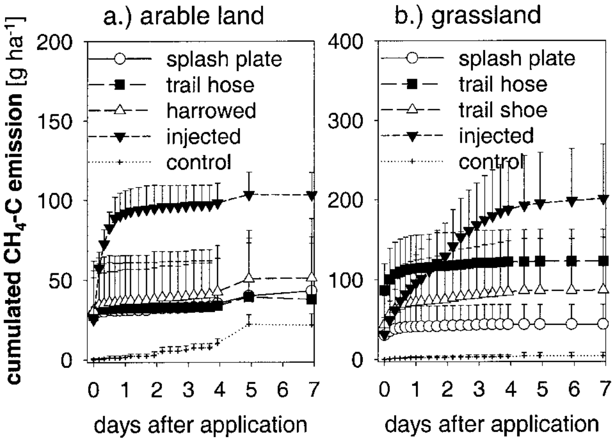

Fig. 2. Cumulated CH_4 emission after spreading co-fermented slurry with different application techniques on arable (*a*) and grassland (*b*) sites. Data represent means and standard deviation (n = 4).

Climatic conditions during the first days of the experiment might also be responsible for higher CH_4 emissions from trail hose–applied slurry compared with co-ferment, being much more obvious on the grassland than on the arable site (Fig. 3).

The higher CH_4 emissions from trail hose–applied unfermented slurry compared with emissions from co-fermented slurry can be assigned to the physical and chemical properties of the substrates. The band of unfermented slurry does not disperse as fast as the less viscous co-fermented substrate, preserves its moisture and thus retains dissolved CH_4, and conserves anaerobic conditions over a longer period. Also, the content of total carbon measured as COD and easily degradable carbon measured as BOD_5 is much higher in the raw slurry than in the co-fermented slurry. In cattle slurry analyzed by Paul and Beauchamp (1989), 50% of soluble carbon were volatile fatty acids. So it can be assumed that also VFA content in the slurry used in our experiment must have been much higher than in the co-ferment. Continued emission for several days from the surface-applied

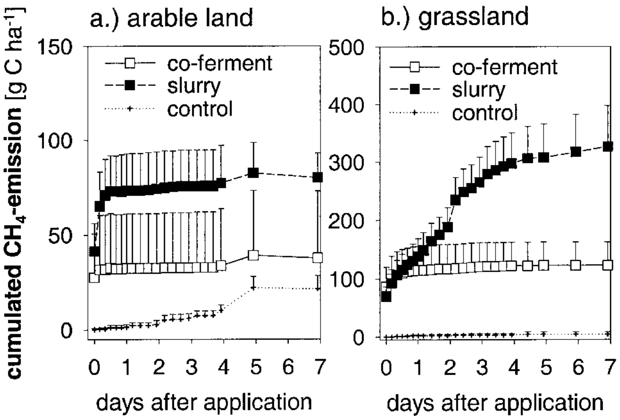

Fig. 3. Cumulated CH_4 emission after trail hose application of co-fermented slurry and unfermented cattle slurry on arable (*a*) and grassland (*b*) sites. Data represent means and standard deviation (n = 4).

unfermented slurry in the grassland experiment indicates that under wet conditions anaerobic sites can develop in the slurry band and CH_4 might be produced. On arable land dry conditions prohibited CH_4 formation.

Nitrous Oxide

Nitrous oxide emissions varied clearly with time and space. The temporal pattern of emissions strongly depended on environmental conditions during the experiment (Fig. 4 and 5). On both sites, increased emission rates could be observed in the first week after fertilization. Further emission peaks occurred in response to changing environmental conditions in the course of the experiments, with the most distinct peaks on both sites being observed after injection of the co-fermented slurry.

Spatial variability of N_2O emissions ranged from 10 to 200% on both sites, but mean variability was much lower on arable land, with 60%, than on the grassland site, with 145%. The explanation might be that on bare arable land, homogeneous spreading of the substrates was easier to achieve than on grassland and soil properties are much more uniform on regularly cultivated arable land than on permanent grassland.

Apart from these general observations, emission patterns were different on the two sites. In the grassland experiment, emissions from the fertilized plots started to increase above background rates about 24 h after application and continued for one week (Fig. 4). Highest emission rates in this period occurred after splash plate spreading with a maximum of 128 μg m^{-2} h^{-1}, but due to high variability were not significantly different from the other application techniques. Emission peaks after fertilization with organic fertilizers are often observed (Dosch and Gutser, 1996; Clemens et al., 1997; Flessa and Beese, 2000; Whalen, 2000) and attributed to transformation and depletion of readily available nitrogen and carbon. The explanation for the decrease of emissions to background levels after one week in the grassland experiment might not only be the depletion of easily degradable carbon pools and thus reduced denitrification activity, but also the decrease of soil temperature at a 10-cm depth to 6°C. Goodroad and Keeney (1984), Kliewer and Gilliam (1995), and Clayton et al. (1997) all describe a strong positive relationship of soil temperature and N_2O emissions. Furthermore, precipitation increased the water content of the already wet soil and may have favored the reduction of N_2O to N_2 (Firestone and Davidson, 1989). In the third and fourth week of our grassland experiment, emissions after injection peaked significantly following a sharp increase of soil temperature and decreasing precipitation (Fig. 4).

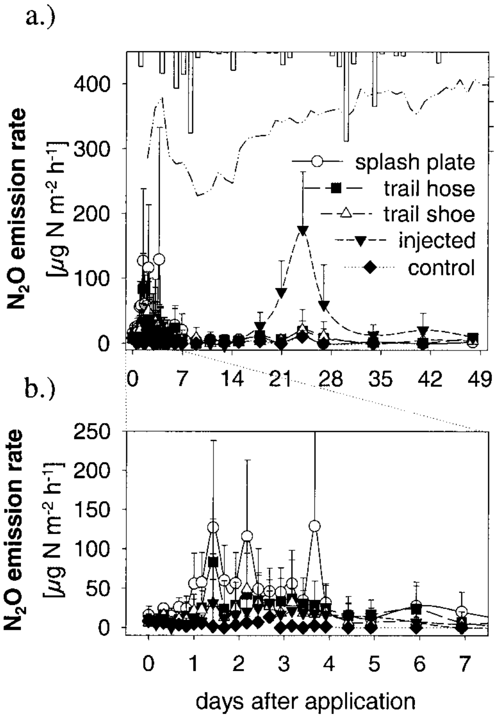

Fig. 4. Nitrous oxide emission rate and meteorological data after spreading co-fermented slurry with different application techniques on grassland. (*a*) Whole experimental period. (*b*) First week of the experiment. Data represent means and standard deviation for emission rates (n = 4).

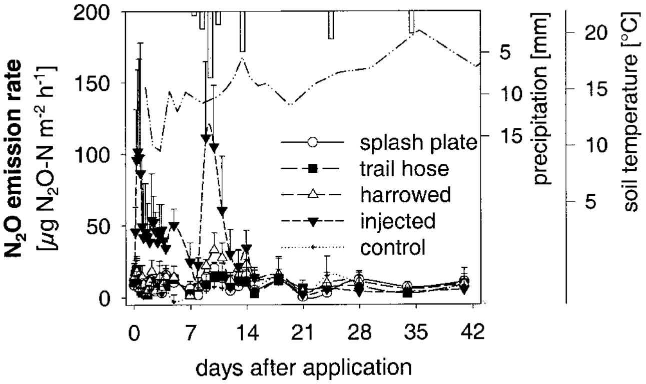

Fig. 5. Nitrous oxide emission rate and meteorological data after spreading co-fermented slurry with different application techniques on arable land. Data represent means and standard deviation for emission rates (n = 4).

The experiment on arable land (Fig. 5) started three weeks later than on grassland, under drier and warmer conditions. Again, increased emissions took place in the first week after fertilization, but were significant only after injection. Under these dry conditions, the input of water can have an effect on emissions as well as the carbon and nitrogen inputs. Through injection the moisture input is confined to a much smaller soil volume and moisture loss through evaporation is minimized compared with surface application. Decreasing soil moisture might be the reason for the decrease of emissions after one week. This is confirmed by a further emission peak in the second week, following precipitation, which could be observed in all treatments but was most pronounced after injection.

Comparing cumulated emissions over the experimental period, the effect of application technique on N_2O emissions becomes evident (Fig. 6). Although being different in emission pattern, injection of the co-fermented slurry resulted in total emissions of 216 g N_2O-N ha^{-1} on arable land and 359 g N_2O-N ha^{-1} on the grassland, equivalent to 0.32 and 0.54% of added NH_4^+–N or 0.17

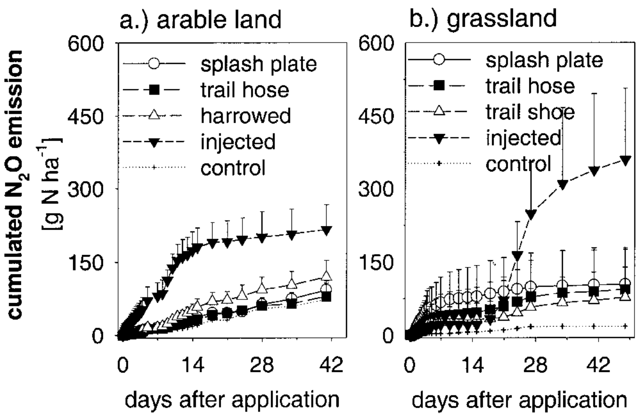

Fig. 6. Cumulated N_2O emission after spreading co-fermented slurry with different application techniques on arable (*a*) and grassland (*b*) sites. Data represent means and standard deviation ($n = 4$).

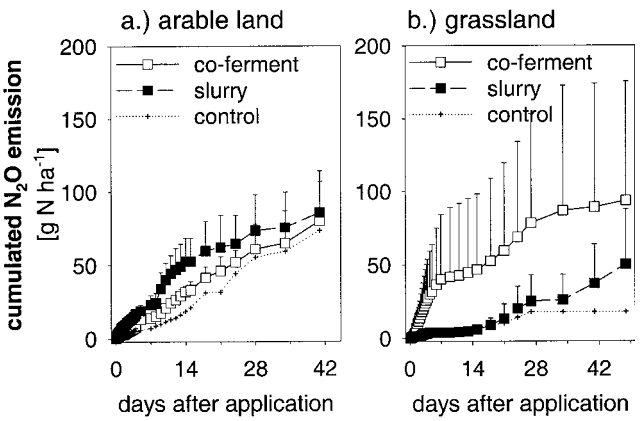

Fig. 7. Cumulated N_2O emission after trail hose application of co-fermented slurry and unfermented cattle slurry on arable (*a*) and grassland (*b*) sites. Data represent means and standard deviation ($n = 4$).

and 0.30% of total N, being two- or even threefold higher than emissions from the other treatments. The creation of anaerobic conditions might be promoted after injection, as air contact is minimized and filtration of the substrate from the slit area into the surrounding soil is decreased as soil pores might be blocked through the smearing effect of the injector tines. Thus, oxygen depletion due to mineralization of the added organic compounds can effectively create microsites of high denitrification activity (Parkin, 1987). Under the drier and warmer environmental conditions during the arable land experiment, this resulted in higher emissions also in the first week after slurry application, whereas in the grassland experiment increased emissions from slurry injection were restricted to the third and fourth week. In the first week, excess water on the grassland site might have promoted further reduction of N_2O to N_2 and low soil temperature might have reduced overall microbial activity. In this period on grassland, slightly higher emissions (not significant $p < 0.1$) occurred after splash plate spreading, probably because of better diffusion conditions for N_2O and warmer daytime temperature at the soil–slurry interface.

The effect of slurry fermentation on N_2O emissions after application on both sites differed distinctly (Fig. 7). On arable land, cumulated emissions up to the second week after application were significantly higher ($p < 0.05$) from plots fertilized with unfermented slurry than from those fertilized with co-fermented slurry. On grassland it was vice versa, with emissions after application of co-fermented slurry being higher than after unfermented slurry application. This difference was also only significant ($p < 0.05$) up to the second week because of increasing variability in emissions, especially from co-fermented slurry.

One possible explanation for higher emissions after spreading of unfermented slurry in the beginning of the experiment on arable land is the reduction of degradable carbon pools during fermentation. Thus, more carbon is available for denitrification processes after application of the unfermented slurry compared with the co-fermented substrate. Dosch and Gutser (1996) observed a distinctly lower denitrification activity after spreading separated slurry on arable land compared with raw slurry and attributed this to a reduction of degradable carbon. Also, Paul and Beauchamp (1989) describe a positive relationship of water-soluble carbon in liquid manure and N_2O emissions after application. In our experiment, the effect of carbon input with the slurry on arable land was of only short duration, pointing out that readily degradable carbon applied with the unfermented slurry seems to be decomposed within a few days, as was shown by Kirchmann and Lundvall (1993) and Sommer et al. (1996) for volatile fatty acids in soil after slurry application.

On grassland we observed not a reduction but an increase of N_2O emissions through fermentation. The reason for this might be that on grassland carbon availability is less limiting for denitrification processes (Granli and Bockman, 1994). Soil dissolved organic C contents at the beginning of our experiment were 137 mg kg^{-1} dry soil on the grassland compared with 38 mg kg^{-1} on the arable site. Of greater relevance on this site might be the contact of unfermented and co-fermented slurry with the soil, which was different for the trail hose–applied substrates because of their divergent physical properties. The viscous band of unfermented slurry was applied onto the grass and was gradually washed through the grass and into the soil by precipitation. During this process carbon decomposition might have already taken place. The more liquid co-fermented slurry, however, passed through the grass layer and infiltrated into the soil much faster. Thus, soil microbial processes might have been induced faster by the co-fermented slurry than by the unfermented slurry, if applied with trail hoses on grassland.

Global Warming Potential

The emissions of the different trace gases are affected in different manners by the fermentation and application techniques. The influence of co-fermentation on N_2O and CH_4 emission was only small and of short duration. Application technique had a much stronger effect, with injection reducing NH_3 volatilization (Wulf

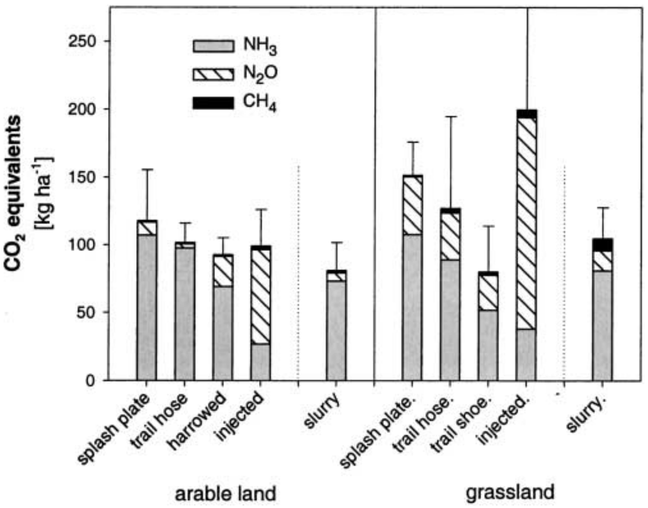

Fig. 8. Climatic warming potential of the different application techniques for co-fermented slurry and trail hose–applied unfermented slurry expressed as CO_2 equivalents calculated from the emission of the single trace gases. Data represent means and standard deviation (n = 4).

et al., 2002), but increasing N_2O emissions two- to threefold. To consider these contrasting effects and to evaluate the potential of co-fermentation and the tested application techniques for the reduction of greenhouse gas emissions as a whole, emissions of the different trace gases were converted into CO_2 equivalents (Fig. 8).

The contribution of NH_3 and N_2O emissions to global warming is evident, while CH_4 emissions after field application are of minor relevance, as CH_4 emissions from enteric fermentation and during slurry storage are much more important (Chadwick et al., 2000). Improved application techniques are designed to reduce NH_3 emissions. Incorporation, injection, and trail shoe application can be very effective in these terms, but bear the risk of increased N_2O emissions. In terms of CO_2 equivalents the increase in N_2O emissions after injection might be as high as the reduction of NH_3 losses or, as in the case of injection on grassland, might even increase overall greenhouse gas emissions. Nevertheless, it should be considered that detrimental effects of NH_3 also include acidification and eutrophication (Fangmeier et al., 1994). These effects are not the subject of this paper, but should be taken into account when choosing certain application techniques.

Comparing the emissions of CO_2 equivalents in our experiments, trail shoe application on grassland could clearly be identified as the most effective measure to reduce greenhouse gas emissions (Fig. 8). For the experiment on arable land, results were not significant, but it should be considered that weather conditions during this experiment were very dry and warm. Under wetter conditions, the potential for NH_3 losses would have been less (Sommer et al., 1991; Vandré et al., 1997) and conditions for denitrification more favorable (Clemens et al., 1997; Maag and Vinther, 1999), which might have reduced NH_3 losses from surface-applied co-fermented slurry and promoted N_2O emissions after injection. Therefore, trail hose application with immediate shallow incorporation seems to be the best way of minimizing the risk of trace gas emissions on arable land.

CONCLUSIONS

Nitrous oxide and CH_4 emissions were affected by application technique, environmental conditions, and fermentation.

The effect of application technique was predominant in our experiments, increasing N_2O emissions two- to threefold after injection of co-fermented slurry on both sites. The promotion of anaerobic sites through carbon input and diffusion constraints through injection seem to be the most probable reasons for this effect. Weather conditions influenced the time course of N_2O emissions but not the effect of application technique on cumulated emissions.

The effect of soil moisture on overall CH_4 emissions was strong, as at the drier arable site emissions seemed to result only from CH_4 dissolved in the substrate, whereas at the wetter grassland site production of CH_4 after application seemed to take place at least after injection of co-fermented slurry and trail hose application of unfermented slurry.

The higher input of organic carbon with unfermented slurry compared with fermented substrates can result in elevated CH_4 emissions if soil moisture is high. Fermentation effect on N_2O emissions seemed to be small and dependant on site characteristics. On arable land, with low soil dissolved organic C contents, a slight transient increase of N_2O emissions was observed after trail hose application. However, on grassland, with higher soil dissolved organic C content, the physical properties of the substrate seemed to be more important, resulting in higher emissions from co-fermented slurry, which passes through the grass much faster than the more viscous unfermented slurry.

Our experiments showed that indirect N_2O production from emitted NH_3 might contribute a great proportion to greenhouse gas emissions from organic fertilization. Therefore, NH_3 measurements should be included in experiments designed to evaluate emissions of greenhouse gases. For spreading co-fermented slurry on grassland, trail shoe application seemed to be the best way of minimizing trace gas emissions. On arable land, trail hose application with immediate harrowing seems to be recommendable, as in addition to the mentioned sources of greenhouse gases, injection of slurry causes higher fuel consumption with negative effects on greenhouse gas budgets. To assess environmental effects on economic feasibility of slurry management practices, further studies are needed that integrate additional aspects such as applicability to certain crops, site exposition, soil structure, and cost of machinery.

ACKNOWLEDGMENTS

We are greatly indebted to Mr. B. Passmann for allowing us to set up the grassland experiment on his farm and Mr. H. Scheb-Wetzel (Institut für Betriebslehre) for providing the arable land site. Dr. M. Thelen (Institute of Agricultural Engineering) helped us very much in organizing and applying the injection treatment. Above all we would like to thank Mrs. W. Schlüter for her valuable support on the field and in the laboratory. This project was funded by the Deutsche Bundesstiftung Umwelt (DBU).

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