

Contents lists available at ScienceDirect

Agriculture, Ecosystems and Environment

journal homepage: www.elsevier.com/locate/agee

Green waste compost reduces nitrous oxide emissions from feedlot manure applied to soil

Ram C. Dalal^{a,b,*}, Iain Gibson^a, Diane E. Allen^a, Neal W. Menzies^b

^a Department of Environment and Resource Management, Indooroopilly, Australia

^b School of Land, Crop and Food Sciences, University of Queensland, Brisbane, Australia

ARTICLE INFO

Article history:

Received 15 March 2009

Received in revised form 8 June 2009

Accepted 15 June 2009

Available online 10 July 2009

Keywords:

Nitrous oxide

Green waste compost

Feedlot manure

Vertisol

Subtropics

ABSTRACT

Australia produces in excess of 1 million tonnes of feedlot manure (FLM) annually. Application of FLM to grain cropping and grazing soils could provide a valuable nutrient resource. However, because of high nutrient concentration, especially of N (>2%), FLM has the potential for environmental pollution, for example, N pollution to the water bodies and N₂O emission to the atmosphere. Therefore, controlling N supply from FLM is essential for the judicious utilisation of FLM in the field as well as reducing N₂O emission to the atmosphere. We utilised the low N concentration green waste compost (GWC, about 3 million tonnes produced annually) as a potential management tool to assess its effectiveness in regulating N release from FLM and controlling the rates of N₂O emission from field application when both FLM and GWC were applied together to sorghum (*Sorghum bicolor* Moench) grown on a Vertisol. We measured N₂O emission rates during the sorghum crop and clean fallowing over one-year period in the field. Annual soil N₂O emissions were 5.0 kg N₂O ha⁻¹ from urea applied at 150 kg N ha⁻¹, 5.1 and 5.5 kg N₂O ha⁻¹ from FLM applied at 10 and 20 t ha⁻¹ respectively, 2.2 kg N₂O ha⁻¹ from GWC applied at 10 t ha⁻¹, 4.3 kg N₂O ha⁻¹ from FLM and GWC applied together at 10 t ha⁻¹ each, and 3.3 kg N₂O ha⁻¹ from the unamended soil. Thus, we found that GWC application reduced N₂O emissions below those from an unamended soil while annual emission rate from FLM approached that from fertiliser N application (~0.7% N₂O emission factor). A mixture of FLM + GWC applied at 10 t ha⁻¹ each reduced N₂O emission factor by 64% (the emission factor was 0.22%), most likely by reducing the amount of mineral N in the soil because soil NH₄-N and NO₃-N and the rate of N₂O emission were significantly correlated in this soil. Since the global warming potential of N₂O is 298 times that of CO₂, even a small reduction in N₂O emission from GWC application has a significant and positive impact on reducing global warming.

© 2009 Elsevier B.V. All rights reserved.

1. Introduction

In Australia over 1 million tonnes of feedlot manure (FLM) is produced each year (ALFA, 2007), containing about 25,000 tonnes of nitrogen, and thus its application to soil could supply significant amount of N to crops. However, N supply from FLM depends primarily on N mineralisation, which is influenced by a number of variables including the nature and composition of FLM, soil moisture, temperature, and microbial activity (Castellanos and Pratt, 1981; Beauchamp et al., 1986; Eghball, 2000). Therefore, it is extremely difficult to develop a generalised prediction of miner-

alisation rates across all conditions and this effect is reflected in the large variability in N mineralisation rates from manures reported in the literature (Castellanos and Pratt, 1981; Paul and Beauchamp, 1994; Gilmour et al., 2003; Kliese et al., 2005a,b).

Green waste compost (GWC) is produced from urban garden waste in excess of 3 million tonnes each year. Compared to FLM, which usually has C:N ratio of 10:1 to 12:1, GWC has much higher C:N ratio, usually >15:1. Since C:N ratio is one of the variables used to predict manure N mineralisation (Janssen, 1996; Qian and Schoenau, 2002), it is expected that N mineralisation rates of FLM are likely to exceed those of GWC. Thus, a combination of FLM and GWC could be used to regulate N mineralisation rates from these organic amendments.

Addition of degradable organic matter such as animal manures to soil increases N₂O production by increasing mineral N (NH₄-N and NO₃-N) as well as C substrate concentration in soil (Bremner and Blackmer, 1981). Also, in subtropical environments, with seasonal high temperatures and uneven rainfall distribution,

* Corresponding author at: Natural Resource Sciences, Environment and Resource Sciences, 80 Meiers Road, Indooroopilly, Qld 4068, Australia. Tel.: +61 7 3896 9895; fax: +61 7 3896 9591.

E-mail addresses: Ram.Dalal@derm.qld.gov.au, Ram.Dalal@nrw.qld.gov.au (R.C. Dalal).

Table 1
Soil properties at the field site.

Depth (m)	Sand (%)	Silt (%)	Clay (%)	BD (Mg m ⁻³)	pH	Organic C (%)	Total N (%)
0–0.1	16	29	52	0.95	7.7	1.78	0.15
0.1–0.3	20	29	55	1.21	8.7	1.08	0.12

events of both high temperatures and high water-filled pore space (WFPS) could favour relatively high N₂O production provided sufficient NO₃ concentration is present in soil (Davidson, 1992; Weier et al., 1993; Castaldi, 2000).

Provided other conditions are favourable for denitrification (temperature, soil moisture, organic carbon, and pH), N₂O production from soil is related to mineral N produced (Dalal et al., 2003). Denitrification is probably the main source of N₂O production in soil although N₂O production by nitrification may sometimes be equally important (Granli and Bockman, 1994). N₂O emission as a proportion of total denitrification increases with increasing NO₃-N concentration in soil (Mosier et al., 1983). Furthermore, since N₂O production also occurs during the nitrification process (Dalal et al., 2003), reducing mineral N in soil could reduce N₂O emissions from soil. Therefore, application of GWC could be used as a management tool to reduce N₂O emission from soil by reducing mineral N in soil.

The objective of our study was to assess the effect of GWC in reducing N₂O emissions from soil when applied either alone or in combination with FLM. This was done by measuring N₂O emissions from soil when organic amendments, FLM and GWC were applied alone or in combination under variable temperature and moisture conditions in the field and then relate the N₂O emission rates to biophysical regulators such as temperature, soil moisture (water-filled pore space, WFPS) and soil NH₄-N and NO₃-N concentrations to identify the regulators of N₂O emission from soil.

2. Materials and methods

2.1. Site description

The field site was located on a flat alluvial plain on the Lockyer Creek in Gatton, South East Queensland, Australia (27°33'56"S and 152°19'33"E). Climate of the region is subtropical, with mean annual temperature of 20.5 °C and cumulative annual rainfall of 780 mm; almost 60% of rainfall is received during summer months (November to March inclusive) (Bureau of Meteorology, Australia). The site was cleared of native bluegum (*Eucalyptus tereticornis*) and

Table 2
Composition of feedlot manure (FLM) and green waste compost (GWC) expressed on an oven-dried weight basis.

Property	FLM	GWC
H ₂ O (%)	28.2	37.9
Organic C (%)	27.8	24.6
Total N (%)	2.40	1.29
C:N ratio	11.6	19.0
Lignin (%)	17.3	21.9
Lignin:N ratio	7.2	17.0
NH ₄ -N (mg/kg)	1092	48
NO ₃ -N (mg/kg)	0	3
Ash (%)	20.9	43.1
Total P (%)	0.77	0.34
K (%)	2.54	0.41

Moreton bay ash (*Corymbia tessellaris*) in 1897 for agriculture. Irrigation and cropping commenced at the site in 1946. Prior to this study, mostly sorghum crops were grown at the field site, generally fertiliser N applied at 100–150 kg N ha⁻¹ and with 1–3 irrigations during the growing season, followed by 6 months of weed-free fallow.

Soil at the field site is classified as a Vertisol (Udic Hapluster, USDA, 1975). The soil contains a high proportion of smectite clay (50–59%). Plant available water capacity of the soil is estimated to be 250 mm (Field Capacity–Wilting Point), with effective rooting depth of 1.2 m. Soil properties including texture, pH, organic carbon and total nitrogen (analytical procedures described in Rayment and Higginson, 1992) are given in Table 1. Monthly rainfall and mean monthly temperatures for the duration of the study are shown in Fig. 1.

2.2. Sample collection

Screened FLM (25 mm) and GWC (10 mm) samples were collected using a front-end loader from nearby FLM stockpiles and a commercial GWC composting facility. FLM was produced from grain-fed beef animals and stockpiled for 6–12 months while GWC was collected from composting facility of trees and shrubs pruning from urban areas and was composted for 12 months before field applications. FLM and GWC samples were stored separately at ambient air temperature in drums. All samples for chemical analysis were ground to pass a 1 mm sieve. Chemical properties of FLM and GWC are given in Table 2. FLM contained 2.4% N and 17% lignin, whereas GWC contained 1.29% N and 22% lignin. The respective C:N ratios of the organic materials were 12 and 19, and lignin:N ratios were 7 and 17 (Table 2).

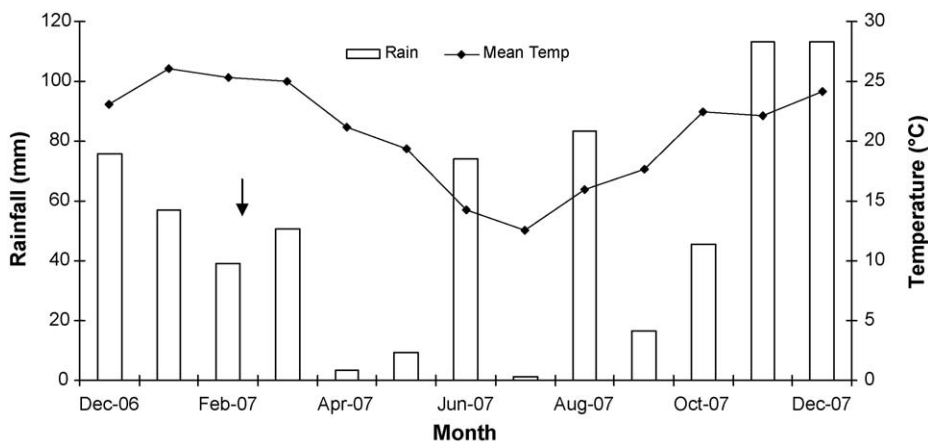


Fig. 1. Monthly rainfall (mm) and mean monthly air temperature (°C) from December 2006 to 2007 at the field site. Sorghum was grown from December 2006 to April 2007, with remaining period clean (weed-free) fallow. The arrow indicates when 50 mm of irrigation was applied to the sorghum crop.

2.3. Field experiment

The field experiment consisted of 6 treatments, arranged in a complete randomised block design with 3 replications. Plot size was 10 m width \times 20 m length. The treatments were: (1) control, (2) 10 t ha⁻¹ FLM (FLM10), (3) 20 t ha⁻¹ FLM (FLM20), (4) 10 t ha⁻¹ GWC (GWC10), (5) 10 t ha⁻¹ of FLM and 10 t ha⁻¹ GWC (FLM10 + GWC10), and (6) 150 kg N ha⁻¹ as urea (Urea150).

The application of FLM and GWC was made on 5 December 2006 to the sorghum crop. The organic amendments were spread manually and immediately rotovated to mix with soil to approximately 0.1 m depth. For urea N treatment, granular urea was evenly dropped on the soil surface and immediately rotovated to approximately 0.1 m soil depth. Sorghum (*S. bicolor* Moench, cv. Pioneer 86G87) was sown on 21 December 2006, at the rate of about 5 kg/ha (150,000 seeds ha⁻¹) at 40 mm depth and 750 mm row spacing. Sorghum crop was harvested on 26 April 2007. The crop was irrigated once (50 mm) using a single jet style winch cannon that travelled down the centre line of the length of the trial in an East-West orientation. Weeds were controlled during fallow by application of glyphosphate. Glyphosphate as Roundup360 was applied at 2 L ha⁻¹ using a standard fan pattern nozzle on a 12 m boom spray.

2.4. Soil sampling

Soil profile sampling was undertaken at the beginning and end of sorghum crop down to 1.5 m depth. Two cores were collected from each plot by a hydraulically operated core sampler with a 50-mm diameter steel tube. Cores were sectioned into 6 depth intervals, 0–0.1, 0.1–0.3, 0.3–0.6, 0.6–0.9, 0.9–1.2, and 1.2–1.5 m depths and the samples from respective depth intervals were combined. Four additional samples in proximity to each deep core were randomly taken from the 0–0.1 m depths using a manual core sampler and bulked with the other 0–0.1 m composite sample. The soil samples were sealed in polyethylene bags in the field and stored at 4 °C until further processing. A sub-sample was dried at 105 °C for 48 h for measuring moisture content of the field soil sample. The soil samples were dried at 40 °C for about 7 days, and the soil was ground to <2 mm size. A sub-sample of the dried (40 °C) soil was further dried at 105 °C for 48 h to determine moisture content of the air-dried soil. The remaining dried (40 °C) soil was stored in sealed plastic containers at room temperature until analysis.

Soil samples from the top 0.1 m depth were taken during the sorghum crop growth and the following fallow period, from December 2006 to 2007. Four samples were collected at random throughout each plot length using a 50 mm diameter core sampler for soil moisture, and NH₄-N and NO₃-N determinations to correlate with N₂O measurements during this period.

Nitrate- and NH₄-N concentrations in the soil samples were determined in soil extracts using 10 g of ground (<2 mm) soil sample and 50 mL of 2 M KCl (Bremner, 1965), and analysed by an automated procedure (Crooke and Simpson, 1971). Soil profile NO₃-N, NH₄-N and mineral N were calculated on an oven-dry weight basis (kg ha⁻¹) using soil moisture content and bulk density for each layer.

2.5. Nitrous oxide sampling

Nitrous oxide gas samples were collected from static closed chambers ($n = 2$ per plot; 6 treatments, 3 replications, 36 chambers total) positioned 4–5 m apart (~ 10 m between plots). Chamber design has been described in Weier (1999) and calculation of gas flux was done according to the procedure of Saggar et al. (2004). In brief, 250 mm diameter and 200 mm high PVC chamber bases were inserted approximated 50–100 mm into the soil to act as ‘collars’

that were permanently positioned along the transect throughout the sampling campaign and removed temporarily only during harvest. Periodically throughout the sampling period, chamber heights were measured to ensure accurate determination of chamber volume.

Gas samples were collected between 9.00 and 11.00 h, with gas samples collected around the same time of day. Preliminary studies showed that gas flux during this period approximated mean daily gas flux. At the time of gas sampling, lids were placed on the chambers to form a gas-tight seal. During the closure period (~ 1 h, see below), 25 mL gas samples were taken intermittently from a sampling tube fitted with a 3-way stopcock and transferred to 12 mL pre-evacuated exetainers (Labco, UK) and transported to the laboratory for analysis.

Gas samples were analysed as soon as possible after collection by gas chromatography (3800, Varian, Netherlands) as described by Allen et al. (2007), using commercial standards (BOC Gases, Australia) with $\pm 1\%$ or better calibration accuracy to calculate sample gas concentrations. Standards were injected every 10 samples to monitor instrument precision. N₂O gas fluxes were calculated using linear increase over time, with values of $R^2 > 0.95$ accepted.

Linearity tests on gas fluxes were performed on a subset of sampling occasions during a range of seasons in all treatment plots. Samples were initially taken upon chamber closure and then each 30 min interval over a 1–1.5 h closure period to test for linearity of gas fluxes. The fluxes of N₂O were linear ($R^2 > 0.95$) over the measurement periods of up to 1.5 h, indicating that linear gas fluxes over the sampling period were preserved.

Nitrous oxide samples were collected from a total of 38 sampling events over the course of the year-long measurement period. Except for the intensive sampling immediately following the application of the amendments and urea to the soil, and during the summer period (December–February), other sampling times were based on rainfall events since N₂O emissions during winter and prolonged dry periods were low. Cumulative N₂O emissions were calculated by linearly interpolating the natural log (Ln) transformed N₂O emission rates between the two measurements over the year-long measurement period.

2.6. Statistical analysis

Significant differences ($P < 0.05$) between treatments (6 treatments and 3 replications) for the various measurements were determined by standard analyses of variance (Snedecor and Cochran, 1967), using untransformed and log-transformed data. In some cases, a log-transformation of data improved the distribution of residuals. The relationships between measured parameters were examined using linear regression analysis and multiple regression analysis (Statistica software, Carver, Brooks/Cole, Canada).

3. Results

The amounts of NO₃-N in the soil profile (0–1.5 m depth) in April 2007 were similar in FLM (FLM10, 194 kg N ha⁻¹ and FLM20, 204 kg N ha⁻¹) and urea N treatments (191 kg N ha⁻¹). However, the GWC treatment (GWC10, 72 kg N ha⁻¹) contained significantly lower amounts of NO₃-N than the FLM10 and FLM20 treatments. The FLM10 + GWC10 treatment (59 kg N ha⁻¹) had similar amount of NO₃-N to that in the control soil (43 kg N ha⁻¹) and the GWC10 treatment, but much lower than the FLM10 and FLM20 treatments (Table 3). Similar trends were observed for soil mineral N since NO₃-N and mineral N were closely correlated at all sampling periods ($R^2 > 0.95$).

Sorghum grain yields showed no significant response to the organic amendments except for a small response to fertiliser N

Table 3
Nitrogen applied, NO₃-N (0–1.5 m) in April 2007, and annual N₂O emissions and emission factors of feedlot manure (FLM), green waste compost (GWC) and urea applied to a Vertisol during the sorghum-fallow period.

Treatment	N applied (kg N ha ⁻¹)	NO ₃ -N (kg N ha ⁻¹)	Annual N ₂ O emissions (kg ha ⁻¹)	Emission factor (%)
Control	0	43.4 ± 8.1	3.29 ± 0.12	0
Urea (150 kg N ha ⁻¹)	150.0	190.8 ± 50.2	4.96 ± 0.26	0.71 ± 0.05
FLM (10 t ha ⁻¹)	186.7	194.4 ± 73.7	5.09 ± 0.26	0.61 ± 0.05
FLM (20 t ha ⁻¹)	373.4	204.1 ± 68.6	5.52 ± 0.35	0.38 ± 0.05
GWC (10 t ha ⁻¹)	80.1	71.8 ± 47.3	2.20 ± 0.13	-0.74 ± 0.04
FLM + GWC (10 t ha ⁻¹ each)	266.8	59.3 ± 28.0	4.25 ± 0.12	0.22 ± 0.03

application (Fig. 2a). However, sorghum N uptake was significantly correlated with the amount of NO₃-N (and mineral N) in the soil at the sowing of the sorghum crop (Fig. 2b). On average, only 29% of the mineral N present at sowing was utilised by the sorghum crop. Thus, substantial amount of mineral N or residual N remained in the soil.

The amount of NO₃-N remaining in the deepest layer (1.2–1.5 m depths) of the soil sampled at the end of the sorghum crop showed the highest amounts of NO₃-N in the FLM10, FLM20, and Urea150 treatments, while GWC10 and FLM10 + GWC10 had similar amounts of NO₃-N to that of the control treatment (Fig. 3). Thus, the potential NO₃-N leaching beyond the root zone from FLM application was significantly reduced by GWC either applied alone or applied together with FLM application.

3.1. Nitrous oxide emission rate

We measured N₂O emission rates for one year after FLM and GWC application to the sorghum crop in December 2006. The N₂O

emission rates peaked soon after rainfall although the peak heights generally decreased over the course of the year-long measurement period. In general, N₂O emission rates were lowest from GWC and even lower than the unamended soil, especially during the weed-free fallow period, but highest from FLM applied at 10 and 20 t ha⁻¹ and fertiliser N applied at 150 kg ha⁻¹ (Fig. 4). In general, the order of treatments for most individual N₂O emission rates was: FLM20 ≥ Urea150 ≥ FLM10 > FLM10 + GWC10 > control > GWC10, although during the cropping phase N₂O emissions from GWC10 were similar to those from the control treatment.

The cumulative N₂O emission over one year was 3.29 kg ha⁻¹ from the unamended soil, 4.96 kg ha⁻¹ from fertiliser N applied at 150 kg N ha⁻¹, 5.09 kg ha⁻¹ from FLM applied at 10 t ha⁻¹, 5.52 kg ha⁻¹ from FLM applied at 20 t ha⁻¹, 2.20 kg ha⁻¹ from GWC applied at 10 t ha⁻¹, and 4.25 kg ha⁻¹ from FLM + GWC applied at 10 t ha⁻¹ each (Table 3). Thus, annual N₂O emissions from the FLM treatment were the highest and that from the GWC treatment were the lowest during the 2006–2007 periods (Table 3).

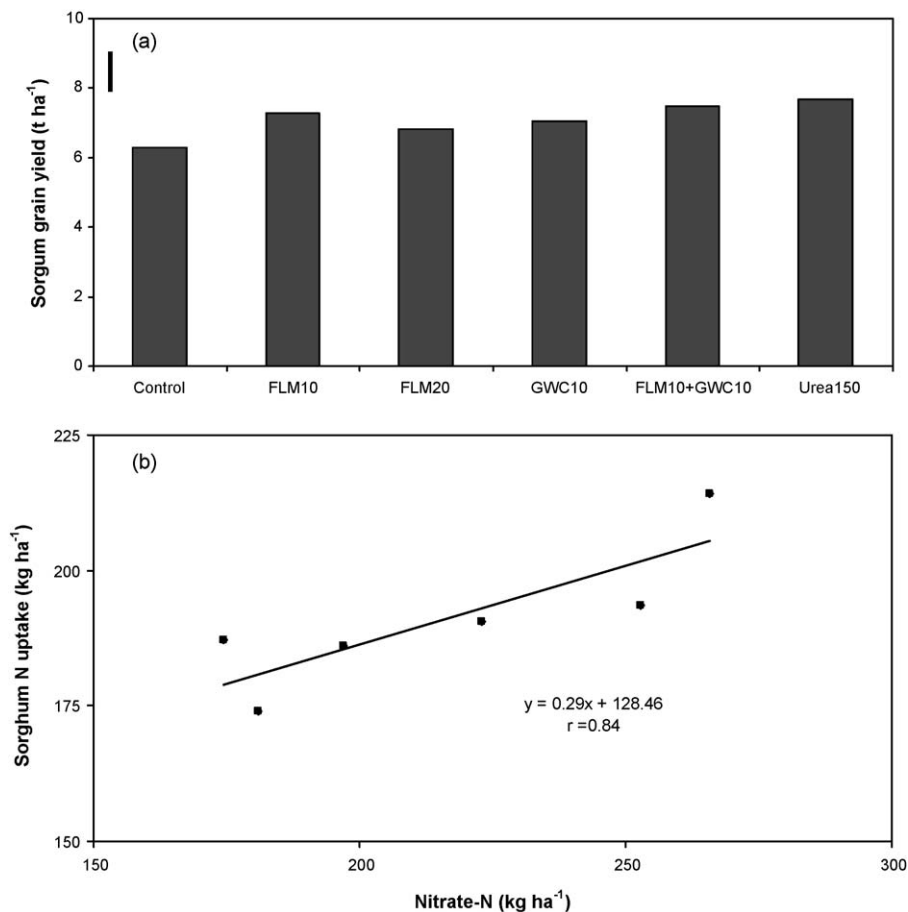


Fig. 2. (a) Effect of organic amendments and fertiliser N on sorghum grain yield and (b) relationship between NO₃-N (0–1.5 m depth) at presowing and sorghum N uptake in grain and straw. FLM10 and FLM20 are feedlot manures applied at 10 and 20 t ha⁻¹, respectively, GWC10 is green waste compost applied at 10 t ha⁻¹, and Urea150 is urea fertiliser applied at 150 kg N ha⁻¹. Vertical bar in (a) represents LSD at *P* < 0.05.

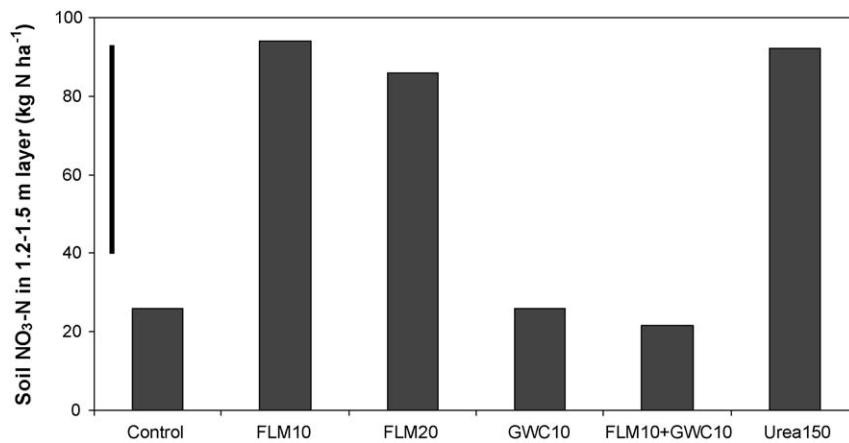


Fig. 3. Effect of organic amendments and fertiliser N application on soil NO₃-N in the 1.2–1.5 m depths at the sorghum harvest. FLM10 and FLM20 are feedlot manures applied at 10 and 20 t ha⁻¹, respectively, GWC10 is green waste compost applied at 10 t ha⁻¹, and Urea150 is urea fertiliser applied at 150 kg N ha⁻¹. Vertical bar represents LSD at $P < 0.05$.

The percent N₂O emission factor ($100 \times (\text{annual N}_2\text{O-N emission from a treatment} - \text{annual N}_2\text{O-N emission from control}) / \text{N added}$) varied from 0.71% from fertiliser N application, 0.61% from FLM applied at 10 t ha⁻¹, and only 0.22% when both FLM and GWC were applied together at 10 t ha⁻¹ each. Although total N₂O emissions were higher from FLM applied at 20 t ha⁻¹ (5.52 kg N₂O ha⁻¹) than FLM applied at 10 t ha⁻¹ (5.1 kg N₂O ha⁻¹), the emission factor was lower (0.38%) from the former. N₂O emission factor was negative when GWC was applied at 10 t ha⁻¹ (-0.74%), that is, N₂O emission rate was lower in GWC treatment than the unamended soil. The addition of GWC to FLM in equal quantities substantially reduced N₂O emissions, by 64% ($100 \times (0.61 - 0.22) / 0.61$) from this Vertisol, thus, demonstrating the effectiveness of GWC in reducing N₂O emissions from FLM applications.

3.2. Relationship of N₂O emission rates and soil properties

The rate of N₂O emission (log-transformed) from the Vertisol was significantly correlated with soil temperature ($r = 0.31$, $n = 228$, $P < 0.01$); it increased with increasing temperature, from 10 to >30 °C (Fig. 5), a typical range for the subtropical soils. For a given temperature, however, a wide range in N₂O emission rate was expected because the other biophysical factors such as NH₄-N and NO₃-N and soil moisture (WFPS) also varied during the experimental period.

The N₂O emission rate response to soil temperature, as expressed by the Q₁₀ value (that is, increase in the rate of N₂O

emission for every 10° increase in temperature; usually Q₁₀ value is close to 2 between 0 and 30 °C when the substrate is in abundance) was estimated to be about 3.5 between 15 and 25 °C, indicating a combination of biological (nitrification and denitrification) for N₂O production and physical processes (solubility, diffusion) for N₂O movement, which governed N₂O emission from this soil.

The rate of N₂O emissions (Ln, natural log-transformed) were also significantly correlated with NO₃-N concentrations (natural log-transformed) in the soil (Fig. 6a) although the correlation coefficient was rather low ($r = 0.18$, $n = 228$, $P < 0.05$), indicating that other biophysical factors, including C substrates from organic amendments, also governed the N₂O emissions from this soil. It increased with increasing amount of NO₃-N in this soil. It was observed, however, that the relationship between N₂O emission rate and NO₃-N was dominated by the urea-N and FLM10 treatments (Fig. 6b), which generally maintained ≥ 10 kg N ha⁻¹ (0–0.1 m depth) in this soil during the experimental period. The relationship between N₂O emission rate and NO₃-N was generally low in GWC and FLM10 + GWC10 treatments, which maintained low NO₃-N concentration in the soil (0–0.1 m depth). Thus, by decreasing the NO₃-N concentration, N₂O emission from this Vertisol was also reduced (Table 3).

The amount of NH₄-N (natural log-transformed) in the soil was significantly and positively correlated with the N₂O emission rate (natural log-transformed) from this soil (Fig. 7). Similar to the relationship between NO₃-N and N₂O emissions, the correlation coefficient was low ($r = 0.23$, $n = 228$, $P < 0.01$), again indicating

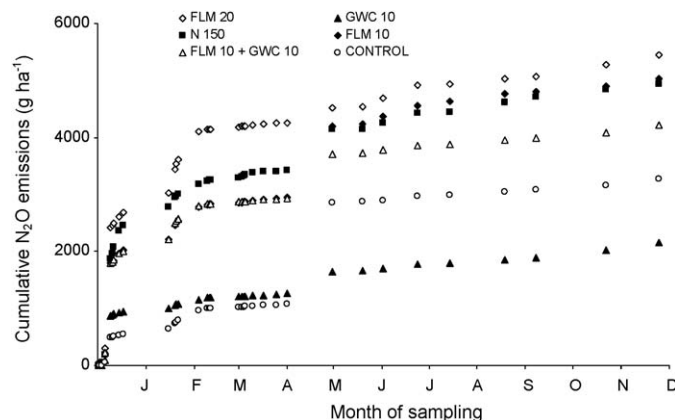


Fig. 4. Cumulative N₂O emissions from the Vertisol treated with feedlot manure (FLM), green waste compost (GWC), and fertiliser N from December 2006 to 2007.

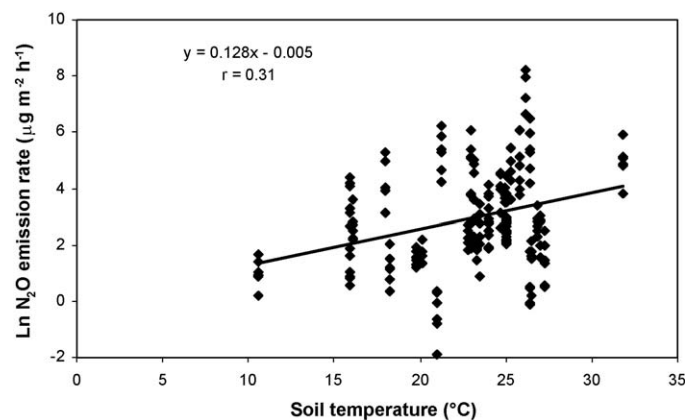


Fig. 5. Relationship between soil temperature (0.1 m depth) and N₂O emissions from the Vertisol.

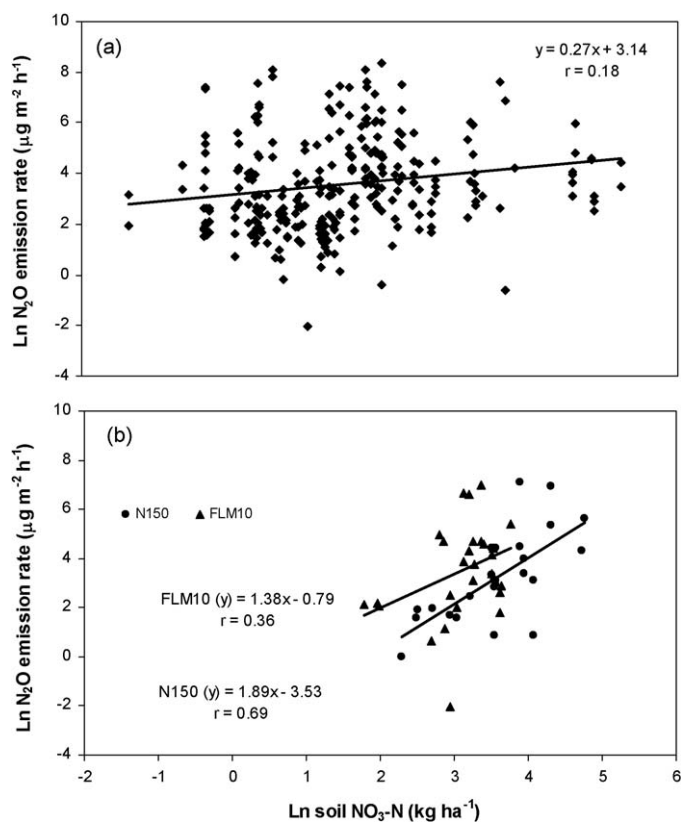


Fig. 6. Relationship between soil $\text{NO}_3\text{-N}$ (0–0.1 m depth) and N_2O emission from the Vertisol (a) all treatments, and (b) fertiliser N and FLM applied to the Vertisol.

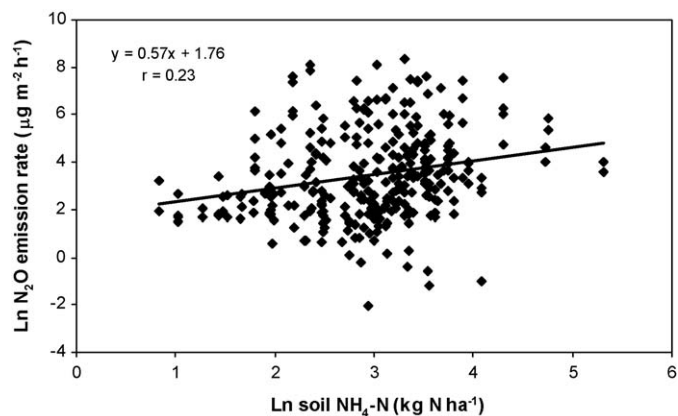


Fig. 7. Relationship between $\text{NH}_4\text{-N}$ (0–0.1 m depth) and N_2O emissions from the Vertisol.

other biophysical factors also governed the N_2O emissions from this soil. As $\text{NH}_4\text{-N}$ is the substrate for nitrifying organisms, it is likely that N_2O was also produced during the nitrification process in this soil.

Within the observed range of WFPS, 30–80%, N_2O emission rate increased with increasing WFPS (Fig. 8) although correlation coefficient was low ($r = 0.16$, $n = 228$, $P < 0.05$), again indicating a number or combination of biophysical factors governed the N_2O emissions from this soil. Also, the low correlation coefficient between water-filled pore space (WFPS) and the rate of N_2O emission from this Vertisol was possibly either due to the narrow range in WFPS measured in this study or it poorly reflected the microsite soil water–air relationships in this soil.

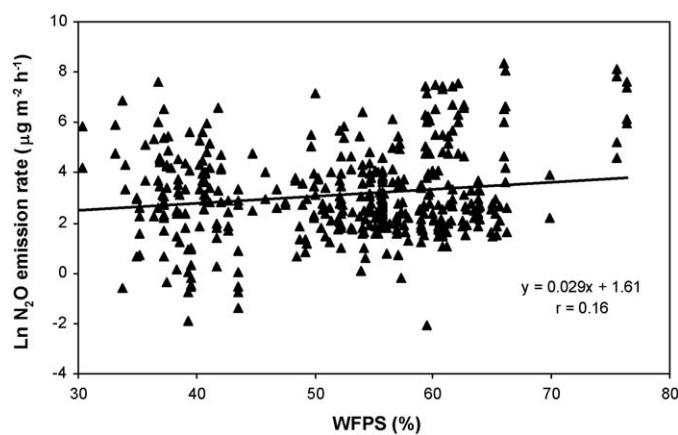


Fig. 8. Relationship between water-filled pore space (WFPS) and N_2O emissions from the Vertisol.

Table 4

Multiple regression analysis of soil biophysical factors and Ln (natural log) N_2O emission rate from Vertisol ($n = 36$).

Property	R	Significance
Temperature and Ln WFPS	0.472	0.05
Temperature and Ln $\text{NH}_4\text{-N}$	0.582	0.001
Temperature and Ln $\text{NO}_3\text{-N}$	0.569	0.01
Temperature and Ln $\text{NO}_3\text{-N}$ and Ln $\text{NH}_4\text{-N}$	0.634	0.001
Temperature and Ln $\text{NO}_3\text{-N}$ and Ln WFPS	0.657	0.001
Temperature and Ln $\text{NH}_4\text{-N}$ and Ln WFPS	0.614	0.001
Temperature and Ln $\text{NO}_3\text{-N}$ and $\text{NH}_4\text{-N}$ and Ln WFPS	0.674	0.001

We used multiple regression analysis to assess the effects of temperature, $\text{NO}_3\text{-N}$, $\text{NH}_4\text{-N}$, and WFPS on N_2O emission rate from the Vertisol. These biophysical factors accounted for only 45% ($R = 0.674$, $P < 0.001$) of the variation in N_2O emission rate from this Vertisol (Table 4). Admittedly, other biophysical factors or products of these biophysical variables must be involved in governing N_2O production and its subsequent emission from the soil.

4. Discussion

4.1. Manure N release under field conditions and sorghum grain yield

There are a number of biophysical variables that influence N mineralisation from the organic materials such as C:N ratio, soil moisture, temperature, and soil labile C and N. Addition of organic amendments of low N concentration to soil can result in net immobilisation of N as inorganic N is utilised by microbes and converted into microbial biomass.

The C:N ratio of the organic substrate is often used to predict net mineralisation or immobilisation when applied to soils (Janssen, 1996; Chadwick et al., 2000). A general rule is that a C:N ratio of higher than 15–30 indicates an initial net immobilisation of N will occur, such as from GWC while a C:N ratio less than 15 will indicate net N mineralisation such as from FLM application. Since FLM had higher N concentration and lower C:N ratio than the GWC, these results confirm the observations of many workers. For example, Chadwick et al. (2000) reported that mineralisation rate was negatively correlated with C:N ratio of the organic materials added to soil. Moreover, GWC also had higher lignin concentration and higher lignin:N ratio than FLM. High lignin concentration in organic materials slows its decomposition (Fox et al., 1990; Jalota et al., 2006). Thus, GWC with high C:N ratio favoured immobilisation of mineral N and high lignin concentration and high lignin:N

ratio slowed organic N decomposition resulting in reduced mineral N in the soil. Since, FLM and GWC differed in N mineralisation rates, a mixture of both organic amendments could be utilised for controlling or moderating nitrate (and mineral) N availability in soil.

The application of FLM at 10 and 20 t ha⁻¹ increased NO₃-N availability in soil. However, GWC treatment had lower NO₃-N concentration than FLM. Consequently, larger amounts of NO₃-N were available for leaching and gaseous N losses, for example N₂O emission from FLM than GWC. In fact, application of GWC not only reduced NO₃-N in the deep soil layer (1.2–1.5 m depths, Fig. 3), but also maintained lower NO₃-N concentration in the top soil depths (Table 2).

Sorghum grain yields were generally unaffected by organic amendments except showing a small response to the urea fertiliser N application (Fig. 2a), thus indicating an adequate supply of N to the sorghum crop.

4.2. The annual rates of N₂O emission from soil amended with FLM and GWC

The annual rates of N₂O emission from this Vertisol were 5.1 and 5.5 kg N₂O ha⁻¹ year⁻¹, from 187.5 and 375 kg manure N ha⁻¹ (10 and 20 t ha⁻¹), respectively, compared with only 1 kg and 1.6 kg N₂O ha⁻¹ year⁻¹ (non-winter 185 days) from soils amended with 170 and 339 kg ha⁻¹ of manure N in temperate region (Lessard et al., 1996). It is worth noting that the application of twice the amount of manure N did not increase N₂O emissions proportionally (Lessard et al., 1996), as we found with FLM10 and FLM20 applications in this study. At much higher manure N application rate, however, Chang et al. (1998) estimated an annual N₂O emission rate of 17 kg N₂O ha⁻¹ from 556 kg manure N ha⁻¹ application (60 t ha⁻¹), possibly due to the accumulation of NO₃-N and organic C in the soil from repeated application of feedlot manure.

In short-term controlled experiments in the laboratory, Chaves et al. (2005) found that application of GWC (low in N) with high N organic materials reduced N₂O emission from the latter by >50%. These short-term potential N₂O reduction benefits using GWC are supported by the current field study where GWC reduced annual N₂O emission rate by 33% from the control soil and by 56% from the soil amended with equal amounts of FLM and GWC compared to FLM alone. While, higher C:N ratio favours immobilisation, higher lignin concentration reduces N mineralisation (Fox et al., 1990; Jalota et al., 2006). Besides the higher C:N ratio of GWC than FLM, it also had higher lignin concentration, and therefore, the combined effect of high C:N ratio and lignin concentration and lignin:N ratio of GWC not only reduced NO₃-N in the soil but it also reduced N₂O emission rate from the GWC-amended soil as well as from the soil amended with FLM.

The N₂O emission factors for feedlot manure N applied at 187.5 kg N ha⁻¹ (10 t ha⁻¹ FLM) and fertiliser N applied at 150 kg N ha⁻¹, 0.6–0.7% were lower than the IPCC (2006) default value of 1% N for the fertiliser N. It was only 0.22% when both FLM (187.5 kg N ha⁻¹) and GWC (93.5 kg N ha⁻¹) were applied together. These N₂O emission factors from FLM were higher than the values estimated by Lessard et al. (1996) but lower than those estimated by Chang et al. (1998), who estimated the N₂O emission factors from 2 to 4% from feedlot manure applied at much higher rates than those were used in this study. Also, the emission factor was lower when twice the amount of FLM was applied (FLM10, 0.71%; FLM20, 0.38%) although annual N₂O emissions were about 25% higher in proportion to the increase in NO₃-N concentrations from FLM20 as compared to FLM10 applications. Changes in other biophysical factors governing N₂O emissions such as soil water–air relationship may have also affected N₂O production, diffusion and emission from soil to the atmosphere (Jury et al., 1982).

4.3. Relationship of N₂O emission rate with biophysical factors

As expected, N₂O emission rate was significantly correlated with soil temperature (Fig. 5). As soil temperature increases, N₂O emission rate also increases, at least up to 37 °C (Castaldi, 2000; Dalal et al., 2003). The calculated Q₁₀ value of 3.5 between 15 and 25 °C in this study indicated a combination of biological and physical processes governing N₂O emission rates from this soil. Nitrous oxide is produced from both nitrification and denitrification processes, which are strongly affected by soil temperature (Davidson, 1992; Dalal et al., 2003). Soil temperature also affects the N₂O solubility in soil solution and its diffusion through soil to its emission to the atmosphere; hence deviation from biological Q₁₀ value of 2 is expected. The estimated Q₁₀ value in this study is within the range of Q₁₀ values (between 1.6 and 12) for N₂O emission rates, estimated by Smith et al. (1998); they estimated the higher Q₁₀ values (up to 12) for a fine-textured soil. Similarly, Q₁₀ values ranging between 2 and 9 for N₂O emission rates from soil have been observed by others (Dobbie and Smith, 2001; Akiyama and Tsuruta, 2003; Zou et al., 2004; Bagherzadeh et al., 2008) for a number of ecosystems.

Since both nitrification and denitrification processes contribute to N₂O production, the significant relationships between N₂O emission rate and amount of NH₄-N in soil (a substrate for N₂O production via nitrification process) and NO₃-N in soil (a substrate for N₂O production via denitrification process) (Figs. 6 and 7) suggests the role of both of these processes in N₂O production in this soil. Nitrification process dominates under aerobic conditions, usually ≤60% WFPS, while denitrification process dominates under anaerobic conditions (>80% WFPS) (Linn and Doran, 1984). However, even under aerobic conditions, N₂O can also be produced by denitrification in anaerobic microsites within soil aggregates (Renault and Stengel, 1994). Moreover, decomposition of FLM further depletes O₂ in soil due to biological O₂ demand and thus promotes denitrification ‘hot spots’ in soil (Rochette et al., 2000; Velthof et al., 2003), especially when NO₃-N concentration in soil is also high (Fig. 6b). Thus, N₂O production from both nitrification process, especially at low WFPS, and denitrification process, especially at high WFPS and high NO₃-N concentration in soil may have occurred even though WFPS varied between 30% and 80% (Fig. 8). This confirms the findings of Akiyama et al. (2004), who found that N₂O emission rate increased with increasing moisture, up to 80% WFPS. Similarly, Rochette et al. (2008) observed increase in N₂O emission rates when both NO₃-N concentration and WFPS were high in soil.

Among biophysical factors governing N₂O emission from this Vertisol, soil temperature, NO₃-N, NH₄-N and WFPS were all significantly correlated with N₂O emission rate (Figs. 5–8) and together accounting for up to 45% variation in N₂O emission rate. Since N₂O emission from a soil in the field shows high spatial and temporal variability (Granli and Bockman, 1994; Dalal et al., 2003), and that biophysical factors may be multiplicative rather than additive in their effects on N₂O production and emission from soil (Parton et al., 2001), close relationship between biophysical factors and N₂O emission rates is rarely achieved. In general, the relationships between N₂O emission rates and the biophysical factors in the field suggests that either the relationship between N₂O emission rates and biophysical factors is complex (Chang et al., 1998) or other biophysical factors not monitored in this study such as dissolved organic C or some other factor may be important in governing N₂O emission rates from this soil amended with organic materials and fertiliser N. Even under controlled incubation experiments, a relationship between dissolved organic C and N₂O emission rates varies from a strong relationship (Bremner and Blackmer, 1981) to a poor relationship (Chaves et al., 2005). Moreover, N₂O emission from soil is not only controlled by N₂O

production rates in the soil but also its extent of dissolution in the soil solution and its rates of diffusion to the soil surface (Jury et al., 1982). Thus, N₂O emission rates from soil may rarely equate with N₂O production in soil, which is most likely governed by regulatory biophysical factors in soil.

5. Conclusion

Feedlot manure application to soil released significant amounts of N, which was either taken up by the crop and/or accumulated as NO₃-N and mineral N in the soil profile. On the other hand, application of GWC to the Vertisol at similar rates initially immobilised or retained N in the soil, resulting in low NO₃-N and mineral N in the soil profile, similar to the unamended soil. Thus, due to the differential N release characteristics of FLM and GWC, an appropriate mixture of these two amendments could be used as a management tool for moderating N release and for their efficient utilisation and environmentally safe application to soils.

We found that GWC application at 10 t ha⁻¹ reduced N₂O emissions below those from an unamended soil while annual emission rate from FLM approached that from fertiliser N application. A mixture of FLM and GWC applied at 10 t ha⁻¹ each reduced N₂O emission factor by 64% by reducing the amount of NO₃-N and mineral N in the soil. Thus, a mixture of FLM and GWC application to field could moderate N supply as well as reduce environmental pollution such as N₂O emission to the atmosphere. This will have both effective nutrient utilisations as well as reduce environmental impact resulting from NO₃-N leaching to water bodies and N₂O emission to the atmosphere.

Although GWC application initially had a small effect on soil N immobilisation, it was sufficiently large enough to moderate N release from FLM10 + GWC10 application. Chaves et al. (2005) findings under the controlled conditions in the laboratory and this field study strongly suggests the role of GWC in moderating N release from high N release organic amendments and accompanying reduction in N₂O emissions, and thus, provides a strong basis for further research in evaluating the role of GWC under different environmental conditions and soil types.

Acknowledgments

We thank Australian Centre for International Agricultural Research for funding, Incitec Pivot for the supply of urea and Starter Z Fertilisers, Aronui Feedlot for bulk feedlot manure supply, Naturfert Spreading for field trial spreading of manure, and Meat and Livestock Australia and Australian Feed Lotters Association for feedlot data. We also thank our colleagues Steven Reeves, Weijin Wang, Judy Brady, Tod Eadie and Dave Rowlings for field and laboratory assistance, and two anonymous reviewers for their constructive comments.

References

Akiyama, H., McTaggart, I.P., Ball, B.C., Scott, A., 2004. N₂O, NO, and NH₃ emissions from soil after the application of organic fertilizers, urea and water. *Water Air Soil Pollut.* 156, 113–129.

Akiyama, H., Tsuruta, H., 2003. Nitrous oxide, nitric oxide, and nitrogen dioxide fluxes from soils after manure and urea application. *J. Environ. Qual.* 32, 423–431.

ALFA, 2007. Australian Lot Feeders Association; Quarterly Reports. Rural Press, Australia. <http://www.farmonline.com.au/farmmags/alfalotfeeding/publication.aspx>.

Allen, D.E., Dalal, R.C., Rennenberg, H., Meyer, R.L., Reeves, S., Schmidt, S., 2007. Spatial and temporal variation of nitrous oxide and methane flux between subtropical mangrove sediments and the atmosphere. *Soil Biol. Biochem.* 39, 622–631.

Bagherzadeh, A., Brumme, R., Beese, F., 2008. Temperature dependence of carbon mineralization and nitrous oxide emission in a temperate forest ecosystem. *J. Forestry Res.* 19, 107–112.

Beauchamp, E.G., Reynolds, W.D., Brasche-Villeneuve, D., Kirby, K., 1986. Nitrogen mineralization kinetics with different soil pretreatments and cropping histories. *Soil Sci. Soc. Am. J.* 50, 1478–1483.

Bremner, J.M., 1965. Inorganic forms of nitrogen. In: Black, C.A. (Ed.), *Methods of Soil Analysis. Part 2*. ASA, Madison, WI, pp. 1179–1237.

Bremner, J.M., Blackmer, A.M., 1981. Terrestrial nitrification as a source of atmospheric nitrous oxide. In: Delwiche, C.C. (Ed.), *Denitrification, Nitrification and Atmospheric N₂O*. John Wiley and Sons Ltd., Chichester, UK, pp. 151–170.

Castaldi, S., 2000. Responses of nitrous oxide, dinitrogen and carbon dioxide production and oxygen consumption to temperature in forest and agricultural light-textured soils determined by model experiments. *Biol. Fert. Soils* 32, 67–72.

Castellanos, J.Z., Pratt, P.F., 1981. Mineralization of manure nitrogen—correlation with laboratory indexes. *Soil Sci. Soc. Am. J.* 45, 354–357.

Chadwick, D.R., John, F., Pain, B.F., Chambers, B.J., Williams, J., 2000. Plant uptake of nitrogen from the organic nitrogen fraction of animal manures: a laboratory experiment. *J. Agric. Sci.* 134, 159–168.

Chang, C., Cho, C.M., Janzen, H.H., 1998. Nitrous oxide emission from long-term manured soils. *Soil Sci. Soc. Am. J.* 62, 677–682.

Chaves, B., De Neve, S., Cabrera, M.D.L., Boeckx, P., Van Cleemput, O., Hofman, G., 2005. The effect of mixing organic biological waste materials and high-N crop residues on the short-time N₂O emission from horticultural soil in model experiments. *Biol. Fert. Soils* 41, 411–418.

Crooke, W.M., Simpson, W.E., 1971. Determination of ammonium in Kjeldahl digests of crops by an automated procedure. *J. Sci. Food Agric.* 22, 9–10.

Dalal, R.C., Wang, W., Robertson, G.P., Parton, W.J., 2003. Nitrous oxide emission from Australian agricultural lands and mitigation options: a review. *Aust. J. Soil Res.* 41, 165–195.

Davidson, E.A., 1992. Sources of nitric oxide and nitrous oxide following wetting of dry soil. *Soil Sci. Soc. Am. J.* 56, 95–102.

Dobbie, K.E., Smith, K.A., 2001. The effects of temperature, water-filled pore space and land use on N₂O emissions from an imperfectly drained gleysol. *Eur. J. Soil Sci.* 52, 667–673.

Eghball, B., 2000. Nitrogen mineralization from field-applied beef cattle feedlot manure or compost. *Soil Sci. Soc. Am. J.* 64, 2024–2030.

Fox, R.H., Myers, R.J.K., Vallis, I., 1990. The nitrogen mineralisation rate of legume residues in soil as influenced by their polyphenol, lignin, and nitrogen contents. *Plant Soil* 129, 251–259.

Gilmour, J.T., Cogger, C.G., Jacobs, L.W., Evanylo, G.K., Sullivan, D.M., 2003. Decomposition and plant available nitrogen in biosolids: laboratory studies, field studies, and computer simulation. *J. Environ. Qual.* 32, 1498–1507.

Granli, T., Bockman, O.C., 1994. Nitrous oxide from agriculture. *Norwegian J. Agric. Sci.* 12 (Suppl.), 7–128.

Jalota, R., Dalal, R.C., Harms, B., Page, K., Mathers, N., Wang, W., 2006. Effects of litter and fine root composition on their decomposition in a rhodic paleustalf under different land uses. *Commun. Soil Sci. Plant Anal.* 37, 1859–1875.

Janssen, B.H., 1996. Nitrogen mineralization in relation to C:N ratio and decomposability of organic materials. *Plant Soil* 181, 39–45.

Jury, W.A., Letey, J., Collins, T., 1982. Analysis of chamber methods used for measuring nitrous oxide production in the field. *Soil Sci. Soc. Am. J.* 46, 250–256.

Kliese, Y.J., Dalal, R.C., Strong, W.M., Menzies, N.W., 2005a. Piggery pond sludge as a nitrogen source for crops. 1. Mineral N supply estimated from laboratory incubations and field application of stockpiled and wet sludge. *Aust. J. Agric. Res.* 56, 245–255.

Kliese, Y.J., Dalal, R.C., Strong, W.M., Menzies, N.W., 2005b. Piggery pond sludge as a nitrogen source for crops. 2. Assay of wet and stockpiled piggery pond sludge by successive cereal crops or direct measurement of soil available N. *Aust. J. Agric. Res.* 56, 517–526.

Lessard, R., Rochette, P., Gregorich, E.G., Pattey, E., Desjardins, R.L., 1996. Nitrous oxide fluxes from manure-amended soil under maize. *J. Environ. Qual.* 25, 1371–1377.

Linn, D.M., Doran, J.W., 1984. Aerobic and anaerobic microbial populations in no-till and ploughed soils. *Soil Sci. Soc. Am. J.* 48, 794–799.

Mosier, A.R., Parton, W.J., Hutchinson, G.L., 1983. Modelling nitrous oxide evolution from cropped and native soils. *Environ. Biogeochem. Ecol. Bull.* 35, 229–241.

Parton, W.J., Mosier, A.R., Ojima, D.S., Valentine, D.W., Schimel, D.S., Weier, K., Kulmala, K.E., 2001. Generalized model for NO_x and N₂O emissions from soils. *J. Geophys. Res.* 106, 17403–17419.

Paul, J.W., Beauchamp, E.G., 1994. Short-term nitrogen dynamics in soil amended with fresh and composted cattle manures. *Can. J. Soil Sci.* 74, 147–155.

Qian, P., Schoenau, J.J., 2002. Availability of nitrogen in solid manure amendments with different C:N ratios. *Can. J. Soil Sci.* 82, 219–225.

Rayment, G.E., Higginson, F.R., 1992. *Australian Laboratory Handbook of Soil and Water Chemical Methods*. Inkata Press, Melbourne.

Renault, P., Stengel, P., 1994. Modelling oxygen diffusion in aggregated soils. 1. Anerobiosis inside the aggregates. *Soil Sci. Soc. Am. J.* 58, 1017–1023.

Rochette, P., Angers, D.A., Chantigny, M.H., Gagnon, B., Bertrand, N., 2008. N₂O fluxes in soils of contrasting textures fertilized with liquid and solid dairy cattle manures. *Can. J. Soil Sci.* 88, 175–187.

Rochette, P., van Bochove, E., Prevost, D., Angers, D.A., Cote, D., Bertrand, N., 2000. Soil carbon and nitrogen dynamics following application of pig slurry for the 19th consecutive year. II. Nitrous oxide fluxes and mineral nitrogen. *Soil Sci. Soc. Am. J.* 64, 1396–1403.

Saggar, S., Andrew, R.M., Tate, K.R., Hedley, C.B., Rodda, N.J., Townsend, J.A., 2004. Modelling nitrous oxide emissions from dairy-grazed pastures. *Nutr. Cycl. Agroecosyst.* 68, 243–255.

- Smith, K.A., Thomson, P.E., Clayton, H., McTaggart, I.P., Conen, F., 1998. Effects of temperature, water content and nitrogen fertilisation on emissions of nitrous oxide by soils. *Atmos. Environ.* 32, 3301–3309.
- Snedecor, G.W., Cochran, W.G., 1967. *Statistical Methods*, 6th Edition. Iowa State University Press, Ames.
- USDA, 1975. *Soil Taxonomy*. United States Department of Agriculture, Washington DC.
- Velthof, G.L., Kuikman, P.J., Oenema, O., 2003. Nitrous oxide emission from animal manures applied to soil under controlled conditions. *Biol. Fert. Soils* 37, 221–230.
- Weier, K.L., 1999. N₂O and CH₄ emission and CH₄ consumption in a sugarcane soil after variation in nitrogen and water application. *Soil Biol. Biochem.* 31, 1931–1941.
- Weier, K.L., Doran, J.W., Power, J.F., Walters, D.T., 1993. Denitrification and the dinitrogen/nitrous oxide ratio as affected by soil water, available carbon, and nitrate. *Soil Sci. Soc. Am. J.* 57, 66–72.
- Zou, J., Huang, Y., Zong, L., Zheng, X., Wang, Y., 2004. Carbon dioxide, methane, and nitrous oxide emissions from a rice–wheat rotation as affected by crop residue incorporation and temperature. *Adv. Atmos. Sci.* 21, 691–698.