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Modelling nitrous oxide emissions from dairy-grazed pastures

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Abstract

Soil N_2O emissions were measured during four seasons from two highly productive grass-clover dairy pastures to assess the influences of soil moisture, temperature, availability of N (NH_4^+ and NO_3^-) and soluble C on N_2O emissions, and to use the emission data to validate and refine a simulation model (DNDC). The soils at these pasture sites (Karapoti fine sandy loam, and Tokomaru silt loam) differed in texture and drainage characteristics. Emission peaks for N_2O coincided with rainfall events and high soil moisture content. Large inherent variations in N_2O fluxes were observed throughout the year in both the ungrazed (control) and grazed pastures. Fluxes averaged 4.3 and 5.0 g N_2O/ha/day for the two ungrazed sites. The N_2O fluxes from the grazed sites were much higher than for the ungrazed sites, averaging 26.4 g N_2O/ha/day for the fine sandy loam soil, and 32.0 g N_2O/ha/day for the silt loam soil. Our results showed that excretal and fertiliser-N input, and water-filled pore space (WFPS) were the variables that most strongly regulated N_2O fluxes. The DNDC model was modified to include the effects of day length on pasture growth, and of excretal-N inputs from grazing animals; the value of the WFPS threshold was also modified. The modified model 'NZ-DNDC' simulated effectively most of the WFPS and N_2O emission pulses and trends from both the ungrazed and grazed pastures. The modified model fairly reproduced the real variability in underlying processes regulating N_2O emissions and could be suitable for simulating N_2O emissions from a range of New Zealand grazed pastures. The NZ-DNDC estimates of total yearly emissions of N_2O from the grazed and ungrazed sites of both farms were within the uncertainty range of the measured emissions. The measured emissions changed with changes in soil moisture resulting from rainfall and were about 20% higher in the poorly drained silt loam soil than in the well-drained sandy loam soil. The model accounts for these climatic variations in rainfall, and was also able to pick up differences in emissions resulting from differences in soil texture.

Introduction

Atmospheric nitrous oxide (N_2O) is annually increasing at a rate of 0.2–0.3%, and this increase is thought to be due to anthropogenic emissions. Nitrous oxide is a potent greenhouse gas contributing to global warming (Bouwman 1990), and accounts for about 6% of the total anticipated Global Warming Potential (IPCC 2001). It is also involved in catalytic destruction of stratospheric ozone following photolytic oxidation to nitric oxide. The long atmospheric lifetime of N_2O (114 years) contributes to its large radiative forcing potential, which is about 296 times that of CO_2 (IPCC 2001). Soils contribute about 65% of the total N_2O produced by terrestrial ecosystems (IPCC 2001). Inventories of anthropogenic emissions of N_2O are required under the Kyoto Protocol of 1997.

Nitrous oxide contributes to about 18% (on a CO_2-equivalent basis) of New Zealand's greenhouse gas emissions. The N excreted by sheep and cattle onto

grazed pastures provides high localised concentrations of available N and C in soils, and is the main source of anthropogenic N_2O emissions from New Zealand, contributing about 85% of total N_2O emissions (Cameron et al. 2000).

Increased N_2O emissions from soils are associated with animal manure and N fertilisation, N derived from biological N_2 fixation, and enhanced N mineralisation (MacKenzie et al. 1998). The major processes responsible for N_2O emissions are nitrification and denitrification. During denitrification, N_2O is an intermediate in the dissimilatory NO_3^- and/or NO_2^- reduction to N_2 under anaerobic conditions and may, therefore, be produced and consumed by denitrifying bacteria in soil (Robertson and Tiedje 1987). Nitrous oxide production by nitrifying bacteria may either arise during NH_4^+ oxidation to NO_2^- or dissimilatory NO_2^- reduction when O_2 supply is limited. Thus the availability of N (NH_4^+ and NO_3^-) (Ball et al. 1997; Castaldi and Smith 1998), and the factors that reduce the redox potential of the soil, such as changes in soil moisture (Carran et al. 1995; MacKenzie et al. 1998; Dobbie et al. 1999; Dobbie and Smith 2001; Hedley et al. 2002), soil texture and organic C (Ineson et al. 1998; Kaiser et al. 1998), have a major affect on the production of N_2O in soils.

Under the Framework Convention for Climate Change, New Zealand currently relies on the IPCC default methodology to calculate anthropogenic emissions from agricultural soils including fertilisers, animal waste, N fixed and crop residues (New Zealand Climate Change Project 2002). The two main N inputs are excretal deposition during animal grazing and N fertiliser. Direct and indirect emissions from animal excreta (dung and urine) are estimated using N excreted by each animal type, and national animal population statistics. The emission factor used is 0.01 kg N_2O-N kg^{-1} excreted N, which is based on two New Zealand studies of representative pastures (Carran et al. 1995; Muller et al. 1995). The IPCC default methodology (IPCC 1997) is limited because of (i) uncertainty in emission factors, (ii) uncertainty in indirect emissions, (iii) limited data on the type and amount of N excreted by grazing animals, and (iv) spatial and temporal variability of N_2O emission. Therefore, this IPCC (Tier I) default methodology is only a first approximation, being too simplistic and generalised (Brown et al. 2001), and ignoring all site-specific controls. Secondly, the IPCC methodology is not sufficiently flexible to allow mitigation options to be assessed. Currently the emission rates from excretal input and from different soils in New Zealand are very uncertain. This uncertainty in New Zealand's N_2O emissions is $\pm$ 65% (Sherlock et al. 2001), which must be reduced if changes since 1990 are to be reported under the Kyoto Protocol.

Accordingly, a more robust, process-based approach is required that is internationally acceptable and quantifies N_2O emissions at the field level more accurately than the IPCC (Tier I) default methodology (IPCC 1997). Such an approach is needed to develop regional- and national-scale inventories with known levels of uncertainties. The major difficulty in quantifying annual N_2O fluxes at the field scale is the high degree of spatial and temporal variability. Few field measurements are available for commercial farms, and most are not for long enough periods to capture seasonal variations. Improved assessment of N_2O emission also requires techniques for reducing random variation within farm units, and knowledge of factors that result in systematic variation among different farms and between seasons.

In the past 20 years, considerable progress has been made on the development of accurate estimates of N_2O emissions. Most estimates were initially computed by extrapolating average fluxes from chamber-based measurements by an areal extent of that system (Davidson 1991) and, like the IPCC methodology, these estimates did not account for spatial, seasonal and interannual variability in climatic and edaphic controls on emission rates (Potter et al. 1993). Three major process-based biogechmeical models: DNDC (DeNitrification DeComposition; Li et al. 1992a, b), DAYCENT (Parton et al. 1996; Del Grosso et al. 2001) and CASA (Potter et al. 1996) are generally used to provide site-specific and regional estimate N_2O emissions from soils in recent years. These models differ in the original purposes for their development and thus have advantages and disadvantages across a potentially wide range of applications. DNDC has reasonable data requirements and has been used to produce regional estimates of N_2O production for the US (Li et al. 1996), China (Li et al. 2001), Canada (Smith et al. 2002), Germany (Butterbach-Bahl et al. 2001), and the UK (Brown et al. 2002). Application of this model directly to New Zealand soils presents a unique challenge, because the soils are distinctive and diverse within short distances, and have higher organic C contents than the world average (Saggar 2001). Furthermore, New Zealand grazed pastoral systems (grazing 24 hours a day) and climatic conditions also differ from those in

most other countries. The DNDC model had therefore, to be modified to represent New Zealand grazed pastoral systems.

Our objectives were to (i) measure N_2O emissions periodically from two dairy pastures with contrasting soils in all four seasons, (ii) assess the influence of soil moisture, temperature, availability of N (NH_4^+ and NO_3^-) and soluble C on N_2O emissions, (iii) assess the ability of DNDC to simulate N_2O emissions from pastures grazed by dairy cattle, and (iv) to make changes to the model to improve this ability.

Model description

The DNDC model of Li et al. (1992a, b) consists of four submodels: thermal-hydraulic, crop growth, decomposition, and denitrification. The model is based on the kinetic processes of N_2O production, where denitrification is activated by a rainfall event that raises soil moisture above a threshold value. Soil temperature and moisture are the key factors controlling both decomposition and denitrification. A thermal-hydraulic submodel uses basic climate data to simulate soil moisture conditions and to capture anaerobic microsite formation and sequential substrate reduction. A crop growth submodel simulates growth of various crops from sowing to harvest. Above-ground biomass is accumulated based on daily N and water uptake. Yield and N content of above- and below-ground plant components are modelled. The decomposition submodel has four soil carbon pools: litter, microbial, labile and passive. Each pool has a fixed C:N ratio and decomposition rate influenced by soil texture (clay content), and soil moisture and temperature. The decomposition submodel provides initial NO_3^- and soluble C pools for the initiation of denitrification, which is also activated by rainfall and increased water-filled pore space (WFPS). An increase in the WFPS caused by rain or irrigation events decreases soil oxygen availability. The WFPS, soluble C, soil temperature, soil pH, available N and denitrifier biomass control the rate of denitrification (Frolking et al. 1998). DNDC has been designed so that soil moisture has a large influence on nitrous oxide fluxes through its impact on the volume of soil in which denitrification occurs and the duration of denitrifying conditions. Water infiltration also causes nitrate leaching. Therefore, successful simulation of nitrous oxide emissions will depend on the successful simulation of soil moisture conditions. The original version of DNDC (DNDC 6.7) has default parameters for soil water contents at field capacity and wilting point as a function of soil texture. The model uses a multi-layered soil for simulating soil water conditions.

Materials and methods

Experimental sites

The experimental sites were part of Massey University, Turitea campus, Palmerston North (40º21' S, 175º39' E). Average rainfall (1970–2000) at these sites is about 965 mm, which is fairly evenly distributed throughout the year, with driest months being January–March. The mean annual sunshine is ~1900 h. The mean annual air temperature is 12.8 °C, and the coldest and warmest months are July (6.8 °C) and January (18.1 °C). Soil temperature and rainfall data during the study period are illustrated in Figure 1.

The two sites were an integral part of Massey University's network of commercial dairy farm systems, and represent highly productive (~16 Mg DM ha^{-1} y^{-1}) legume-based pastures on two soil types with contrasting texture and hydraulic properties (Weathered Fluvial Recent, Karapoti fine sandy loam, Dairy 1, and Argillic-fragic Perch-gley Pallic, Tokomaru silt loam, Dairy 3). These pastures were 3 km apart. Hereafter, we refer to these sites as Dairy 1 and Dairy 3.

Both the Dairy 1 and Dairy 3 received four and three split applications of urea amounting to 130 and 190 kg N ha^{-1}, respectively. The N_2-fixed by an average New Zealand dairy farm is 110 kg ha^{-1} and atmospheric deposition is only 2 kg N ha^{-1} (Ledgard et al. 1999). The farms were rotationally grazed for periods of 12 to 48 hours at a time by Friesian dairy cattle. Intervals between grazing varied from 2 to 6 weeks, depending on herbage accumulation. The pasture growth varied with season being highest in spring and lowest in winter, and herbage N contents ranging between 16.2 and 39.4 g kg^{-1} (Saggar and Hedley 2001). The numbers of cow grazing days per ha were 917 and 877 in Dairy 1 and Dairy 3, respectively. A representative area of each farm (5 × 5 m) was fenced to exclude grazing, and excretal and fertiliser inputs, and was used as a control. The control areas were mown to 20 mm height at each grazing event, with clippings removed.

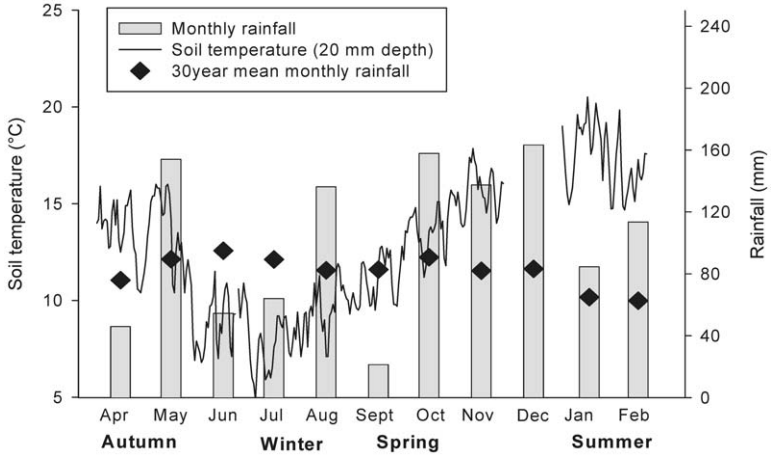

Figure 1. Distribution of daily soil temperature and monthly rainfall during the study period and 30-year mean monthly rainfall.

Nitrous oxide measurements

Nitrous oxide measurements were made periodically between April 2001 and February 2002 from the grazed and ungrazed areas at each site. To account for spatial variability, 18 replicate chambers were randomly located ca. 20 m apart on a Z-shaped transect to measure the fluxes of N_2O from the grazed area (~1 ha), and two chambers were located in the ungrazed area (~0.005 ha). To protect the chambers from damage, they were removed from the sites while stock were present.

The chambers were modified PVC 'Sewer-hatch' (Ø250 mm, 300 mm high) attached to Ø250-mm sections of PVC pipe that were 200 mm deep. The 'Sewer-hatch' rim had an internal half-turn locking system and a greased rubber O-ring, which forms a gas-tight seal. Chambers were randomly inserted 100 mm into the soil one day before each set of measurements began. Chamber heights were measured and the volume of each chamber was calculated. After sealing, three gas samples were taken at times t_0, t_{30} and t_{60} (30-min intervals) from each chamber with 60 ml polypropylene syringes fitted with 3-way stopcocks. Because of concern over the permeability of polypropylene to N_2O, field samples were analysed as soon as possible after collection, usually within 24 hours. No leaks were observed during this time. Gas samples were analysed using a Shimadzu GC-17A gas chromatograph equipped with a ^{63}Ni-electron capture detector (oven, valve and detector temperatures were operated at 65, 100 and 280 °C, respectively, using oxygen-free nitrogen) and connected to an automatic sampler capable of handling up to 120 samples. A sample of the ambient air was collected directly after closing the chambers (t_0), and was used as a reference for calculating N_2O gas fluxes. Accuracy of the gas chromatographic data at ambient concentrations was $\pm$ 1% or better. The increases in N_2O concentrations within the chamber headspace were generally linear ($R^2 > 0.90$) with time. Therefore, N_2O fluxes were calculated (Mosier and Mack 1980) using linear regression and the ideal gas law from Equation (1):

$$F = \rho \frac{V}{A} \frac{\Delta c}{\Delta t} \frac{273}{(T + 273)} \quad (1)$$

where F is the N_2O flux (mg m^{-2} h^{-1}); ρ is the density of N_2O (mg m^{-3}); V is the volume of the chamber (m^3); A is the base area of the chamber (m^2); $\Delta c/\Delta t$ is the average rate of change of concentration with time (ppmv h^{-1}); and T is the temperature (°C) in the chamber.

Soil sampling

Four soil cores were taken to 0–50 and 50–100 mm depths from outside each chamber on Day zero, and two from within each chamber on the final day of each 2–6 week measurement period between the grazing events. The cores were pooled, field-moist soils were sieved (< 4 mm) soon after collection, and subsamples were air-dried. During each measurement period at Dairy 1 six samples were also collected to 0–50 and 50–100 mm depths to determine soil water contents.

Soil analyses

The sieved (<4 mm) field-moist soils were extracted with 2M KCl for 1 h (1:10 soil:extractant ratio), and extracts analysed for mineral-N (NO_3^-–N plus NH_4^+–N) and extractable C. Ammonium–N concentrations were determined by alkaline hydrolysis and reaction with salicylate/nitroprusside and NO_3^+–N concentrations by diazotization following hydrazine reduction, and were measured by flow injection analysis on a Lachet FIA+8000 Series instrument (Quikchem Method Manual 2001). Extractable C was measured on an Elemeter High TOC II +N analyser, where the sample is combusted and CO_2 measured by infrared detectors.

Environmental variables

At the Dairy 3 site, an automated meteorological station was set up to monitor daily rainfall, wind velocity, soil temperature and soil water content (SWC). For the Dairy 1 site, data were taken from an existing automated meteorological station 2 km away. Soil temperature was measured at both sites with a thermocouple at 20 mm depth, and the data were recorded at 30-min intervals.

Soil water content (SWC)

Volumetric SWC was monitored using calibrated Delta-T theta probes (MLZx) at depths of 20, 50, 100 and 300 mm at the Dairy 3 site. At the Dairy 1 site, the gravimetric SWC was measured at 0–50 and 50–100 mm depths, only on gas sampling days. Field-moist soil samples were weighed, oven-dried (105 ºC) to constant mass, and weighed again. The final mass M_s, and the difference between the field moist and dry masses M_w were used to calculate the gravimetric SWC $= (M_w/M_s)\times 100$. The volumetric SWC was then calculated by multiplying the gravimetric SWC by the soil bulk density.

Water-filled pore space (WFPS)

Soil bulk density was determined from four undisturbed (Ø100 mm) soil core samples and particle density was assumed to be 2.65 Mg m^{-3}. Total porosity (TP) was calculated for each soil, according to Equation (2):

$$\text{Total pore space (\%)} = 100\left[1 - (\text{bulk density/particle density})\right] \quad (2)$$

Water-filled pore space (WFPS) was then calculated as the ratio of the volumetric SWC to the total pore space.

Modelling

DNDC was modified to better represent New Zealand's grazed pasture systems. Modifications were made to version 6.7 of the model, the most recent version available when the research began.

Model modifications

Pasture growth: In New Zealand, annual pasture growth rates vary with season. At these sites pasture growth was highest in spring (75–79 kg DM ha^{-1} d^{-1}) and lowest in winter (18–20 kg DM ha^{-1} d^{-1}) (Saggar and Hedley 2001). Low soil temperatures limit pasture growth during winter despite high soil moisture, whereas low soil moisture rather than temperature is the major growth-limiting factor in summer. The growth and N uptake components of the original model were developed for cropping situations, and lacked the strong seasonal pattern observed in our pastures. To account for the seasonal variations in pasture growth and N uptake rates typical of New Zealand pastures a multiplicative day-length factor was introduced. The factor was calculated as daylight hours divided by 12, so that more N was taken up on longer days and less on shorter days.

Soil infiltration and drainage: The original DNDC model simulated soil moisture by adding rainfall and immediately draining before allowing any other processes to occur. Simulations based on this assumption did not exhibit the saturated soil conditions observed at the sites during the winter campaign. By reversing the order of these two operations (drainage followed by infiltration), saturation was made possible, and simulations showed much better correspondence with measured soil moisture at both sites.

Air-soil temperature relationship: The relationship between air and soil temperatures (°C) in the original model was based on northern hemisphere conditions, and was not suitable for New Zealand conditions. Therefore, this relationship was replaced by the following simple relationship developed using New

Zealand-specific national soil and air temperature (°C) datasets (Dr Dave Scotter, personal communication):

$$T_{surface} = T_{air} + 1.3 \quad (3)$$

WFPS-denitrification threshold: Soil moisture has a large influence on N_2O fluxes in the model through its impact on the volume of soil in which denitrification occurs and the duration of denitrifying conditions. The original model uses a soil moisture threshold value of 35% WFPS to switch on/off denitrification. As the WFPS increases, diffusion of oxygen into soil aggregates will decrease, and a rapidly increasing fraction of the soil volume will become anaerobic, causing increased N_2O production by denitrification (Dobbie and Smith 2001). Our measurements showed that field capacity was a clear trigger point for the production of N_2O, so this was used in place of the original model's threshold of 35% WFPS.

N inputs from grazing animals: Nitrogen excretion values were calculated using an average animal intake of 12 kg pasture DM day^{-1} containing 2.5 to 3.0% N (see Materials and methods). The N retention in animal products was assumed to range between 10–20% of the total N intake and the remaining N excreted via faeces and urine (Haynes and Williams 1993). Therefore daily excretal N ranged between 182 and 398 g N (mean 290 g N). Dairy cows excrete 60–65% of their N as urine (Oenema et al. 1997), so a 60:40 ratio of excretal-N urine to dung was used in the model. These inputs were applied during each grazing event, based on the number of grazing cows and grazing period, using DNDC's manure-input mechanism. To facilitate this, we had to increase the number of annual-manure inputs that DNDC would accept.

The modified model, hereafter named 'NZ-DNDC', was then used to simulate N_2O emissions from our dairy cattle-grazed pastures.

Results and discussion

Climatic characteristics

Our sites have an annual average rainfall (over the last 30 years) of 965 mm, which is generally well distributed throughout the year (Figure 1). The annual rainfall during the study year (1040 mm) was slightly higher than the average rainfall received in most years. Rainfall distribution from January to April was 158 mm, lower than the average of 272 mm. A particularly mild but wet late autumn and winter period during May–July delivered 339 mm of rainfall compared with a 30-year average of 273 mm. September (22 mm rainfall) was the driest month but the rainfall during October to December was twice the 30-year average of 250 mm for this period.

The soil temperature at 20 mm depth ranged between 4.9 and 20.5 °C (see Figure 1). In autumn (April), growth and N uptake were mainly constrained by low soil moisture rather than by temperature. However, during winter (June to August), low soil temperatures and reduced daylight hours significantly limited pasture growth and N uptake despite high soil moisture levels. Pasture growth in early spring (September) was constrained by low soil moisture. However, higher than normal rainfall during the remaining spring period resulted in highest growth, which then declined in the summer season (January and February).

Seasonality of N_2O fluxes

Nitrous oxide fluxes were measured between April 2001 and February 2002, during which period environmental conditions varied considerably. The investigation period was, therefore, approximately divided into 'autumn' (April), 'winter' (June–July), 'spring' (October–November) and 'summer' (January–February) to capture the effects of all four seasons.

Large spatial and temporal variations were observed in the N_2O fluxes measured from the grazed area in both pastures (Figure 2a, b). Large fluxes were generally observed after each grazing and rainfall event, followed by a decline. The fluxes in the ungrazed (control) pastures were mostly between 0.3–25.2 g N_2O/ha/day for the fine sandy loam soil (Dairy 1), and 0.3–11.6 g N_2O/ha/day for the silt loam soil (Dairy 3); arithmetic means of all measurements of the control soils were 5.0 and 4.3 g N_2O/ha/day, respectively. These N_2O fluxes and their spatial variability at both the ungrazed sites were comparatively low, with coefficient of variation values ranging between 35–59%.

The N_2O fluxes from the grazed sites were much higher than those from the ungrazed sites (Figure 2a, b), averaging 26.4 g N_2O/ha/day for the fine sandy loam soil, and 32.0 g N_2O/ha/day for the silt loam soil. The spatial variations in N_2O fluxes observed for

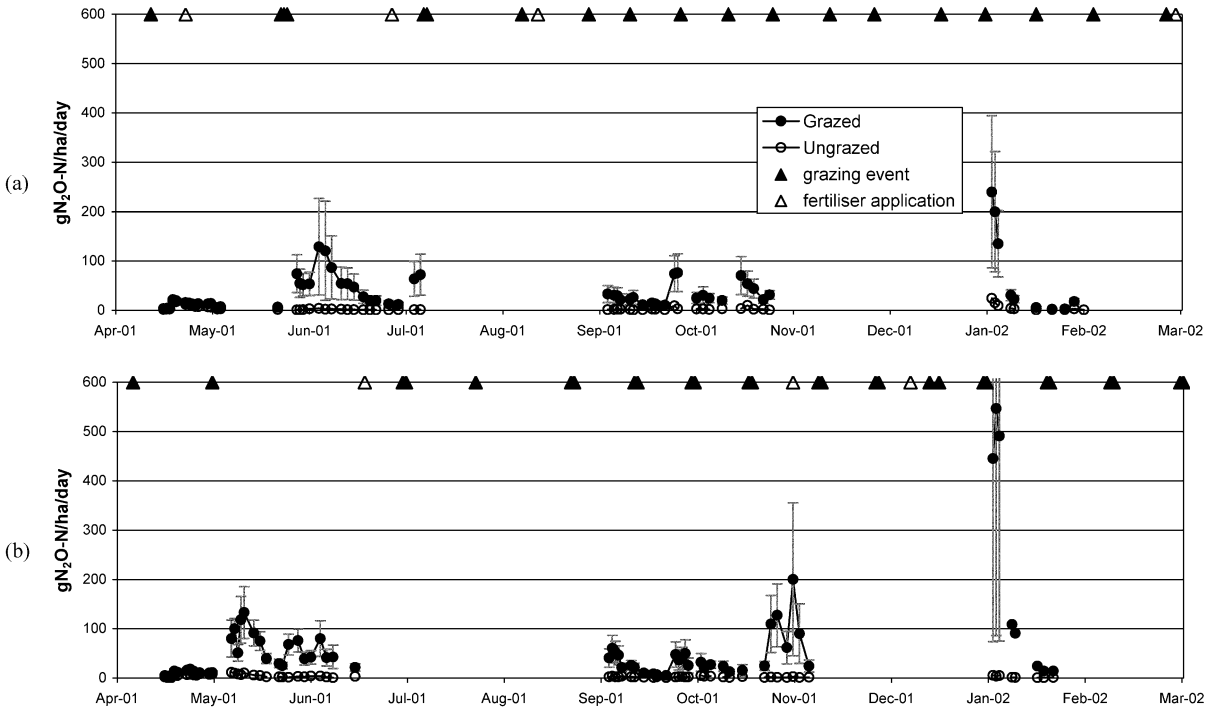

Figure 2. Measured nitrous oxide emissions in ungrazed and grazed dairy pastures; (a) Karapoti fine sandy loam soil (Dairy 1) and (b) Tokomaru silt loam soil (Dairy 3). Each value for the grazed sites represents the mean of 18 replicates with 1 standard error shown by vertical bars. Each value for the ungrazed sites is the mean of 2 replicates.

the grazed sites throughout the year were also large, with coefficient of variation values ranging between 56–262%. The fluxes were more variable during the winter and spring periods when the soils were wet than during the dry autumn period. Spatial variability in N_2O emissions is naturally large in most soils (Folorunso and Rolston 1984; Choudhary et al. 2002) because of soil heterogeneity and the episodic nature of N_2O emissions. Variability increased as a result of animal grazing and unevenly distributed excretal returns.

In both soils, the N_2O emissions for the grazed areas in winter (mean $\pm$ SE, 47.1 $\pm$ 28.7 g N_2O/ha/day for the fine sandy loam soil, and 44.0 $\pm$ 15.4 g N_2O/ha/day for the silt loam soil) were somewhat higher than those in spring (33.5 $\pm$ 16.1 and 40.4 $\pm$ 25.1 g N_2O/ha/day, respectively). The emissions were lower by about one-third in autumn (12.0 $\pm$ 1.6 and 11.2 $\pm$ 2.7 g N_2O/ha/day, respectively). Highest emissions (almost 5 to 10 times those in winter) were measured in both soils for about a week after a heavy rainfall in January, following a grazing event. These emissions dropped rapidly to the levels observed in autumn as the soils dried out. These very high summer N_2O fluxes could be attributed to a combination of high WFPS, high soil mineral-N, and very high soil temperature (~20 °C). The N_2O fluxes increase with increasing temperatures, and our temperature response was similar to that obtained by Dobbie and Smith (2001). These authors found a profound temperature effect on N_2O flux from soil cores incubated between 5 and 18 °C, with the increase over the 12–18 °C range having a Q_{10} of 8.9. The high N_2O fluxes observed during the winter could have been due to a combination of WFPS exceeding 0.60, a high soil mineral-N content and low plant uptake of N. The low rainfall in autumn resulted in a low soil water content (WFPS < 0.60); during this period N_2O emissions following the first grazing event were the lowest found over the year. Spring was characterised by significant rainfall events, rapidly fluctuating soil water content (WFPS between 0.40 and 0.95), mild temperatures, and faster plant growth that resulted in medium levels of N_2O emissions.

This seasonal pattern of N_2O fluxes is consistent with the data of Ruz-Jerez et al. (1994) and Carran et al. (1995). The lowest emissions were obtained during the periods when WFPS was < 0.60, supporting

Table 1. Mean mineral N and extractable C levels in the grazed and ungrazed pasture soils (0–10 cm depth) of Dairy 3 during different seasons. Numbers between brackets indicate the range in values for each season.

Season	NH_4-N (mg N kg^{-1} soil)		NO_3-N (mg N kg^{-1} soil)		Extractable C (mg C kg^{-1} soil)	
	Grazed	Ungrazed	Grazed	Ungrazed	Grazed	Ungrazed
Autumn	4.8 (0.0–46.7)	1.5 (0.1–3.1)	6.2 (1.0–89.9)	2.7 (1.2–5.1)	N.A.	N.A.
Winter	34.8 (0–205.6)	13.0 (2.4–30.4)	24.3 (3.4–103.4)	11.3 (8.5–14.6)	166 (109–306)	178 (152–203)
Spring	6.2 (0–300.8)	2.4 (0.8–6.7)	7.1 (0.1–115.3)	5.3 (0.8–11.4)	118 (22–429)	117 (42–243)
Summer	3.0 (1.2–11.3)	1.2	18.5 (2.3–118.2)	11.2	224 (188–253)	258

their hypothesis that nitrification and other aerobic transformations of urine-derived N contribute little directly to overall emissions, and that denitrification is the primary source of emitted N_2O in New Zealand pastures.

The N_2O emissions for the ungrazed pastures were initially 9.5 ± 2.9 g N_2O/ha/day for the fine sandy loam soil, and 6.4 ± 2.6 g N_2O/ha/day for the silt loam soil (Figure 2a, b). Emission rates subsequently declined and remained low throughout winter, spring and summer (1.7, 2.8 and 3.9 g N_2O/ha/day for the sandy loam soil, and 3.6, 2.5 and 2.0 g N_2O/ha/day for the silt loam soil, respectively). If the data from the whole investigation period are compared, N_2O emissions from the ungrazed pastures were $< 10\%$ of those from the grazed pastures. These results suggest that in grazed pastures it is the animal excreta deposited in the form of dung and urine, and applied N fertiliser that provide high concentrations of available N (Table 1), and are the principal cause of N_2O production. Generally N_2O emissions after each grazing event were linearly related to the levels of mineral N (data not reported).

Effect of soil water content (SWC)

Expressing soil water content (SWC) as WFPS normalises for differences in bulk density and particle density between soils. Our results show that, of the measured variables, WFPS most strongly influenced N_2O fluxes at both grazed sites. The respective WFPS at field capacity were estimated to be 0.66 and 0.62 for the sandy loam and silt loam soils. Generally, N_2O emissions at the grazed sites were high when the WFPS was above 'field capacity' (Figure 3a, b), indicating that formation of anaerobic sites following rainfall, a fundamental requisite for denitrification, was mainly responsible for these high N_2O fluxes. Davidson (1991) showed that nitrification was the dominant source of N_2O when WFPS was < 0.60 and

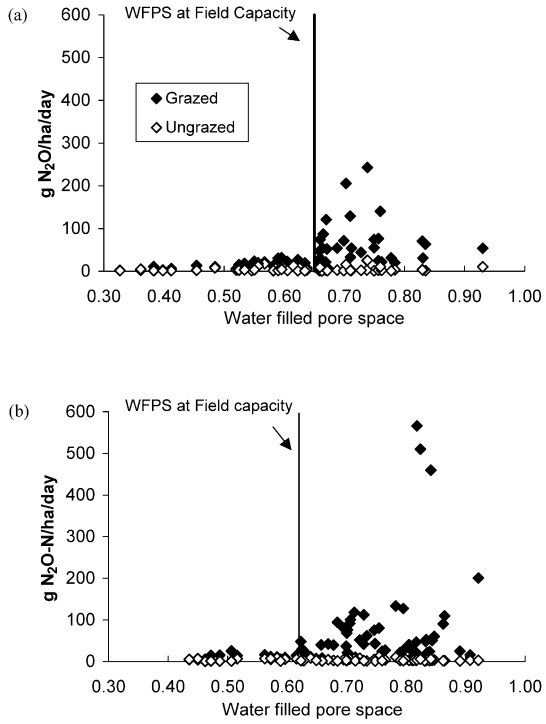

Figure 3. The relationships between nitrous oxide emissions and soil water-filled pore space (WFPS) in (a) Karapoti fine sandy loam soil (Dairy 1) and (b) Tokomaru silt loam soil (Dairy 3).

denitrification was the predominant source at WFPS > 0.60. Studies by Dobbie et al. (1999) and Hedley et al. (2002), where the N_2O emissions increased exponentially with increased SWC content above the field capacity, support these findings. Changes in WFPS above field capacity had no discernible effect on N_2O emissions at the ungrazed sites. Lack of such a relationship in these sites, where soil NO_3^- and extractable C levels were very low (Table 1), also shows N_2O emissions at the ungrazed sites are obviously limited by the lack of substrate.

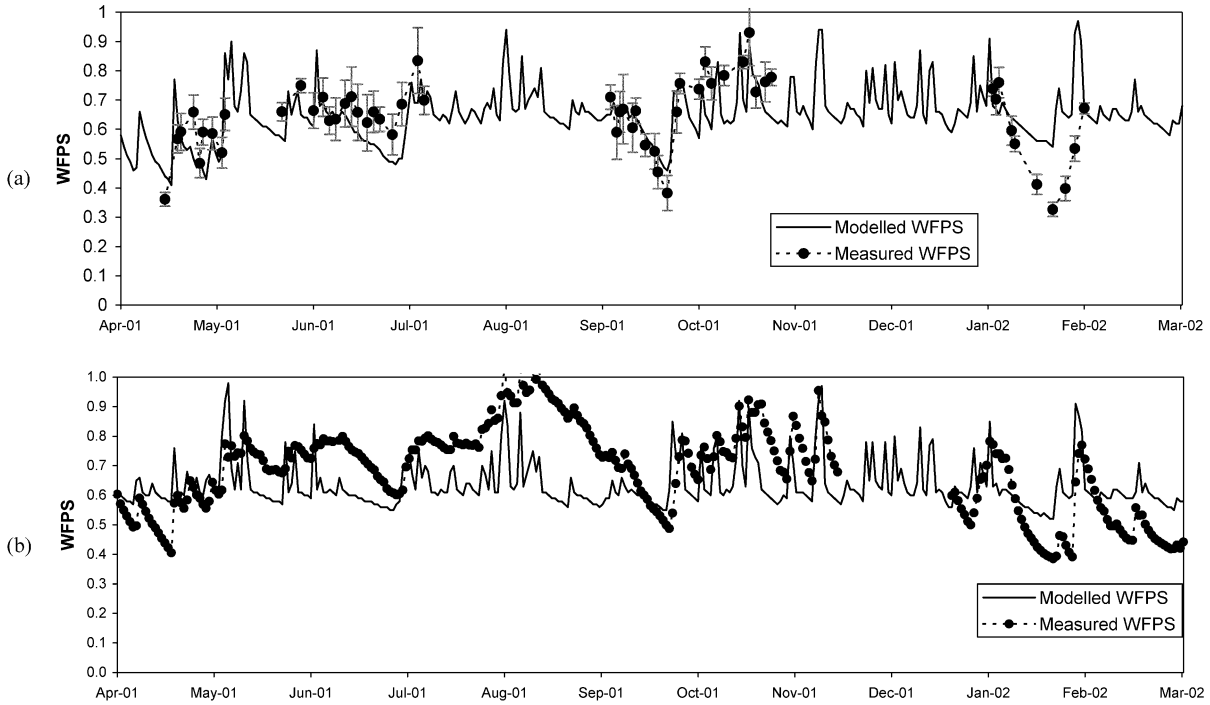

Figure 4. Measured and predicted WFPS during autumn, winter and spring seasons in the top 5 cm of (a) Karapoti fine sandy loam soil (Dairy 1) and (b) Tokomaru silt loam soil (Dairy 3).

Comparisons of model predictions with field measurements

Our modified model, NZ-DNDC, simulated changes in WFPS very well in the top 5-cm depth for the fine sandy loam soil (Figure 4a), with differences between the simulated and measured WFPS being generally less than 10%. These changes in WFPS covered all major rainfall events and subsequent water depletions. However, the agreement between simulated and measured changes in WFPS was not as good for the silt loam soil (Figure 4b); the simulations had lower WFPS at those times when the top 5 cm of soil was almost saturated with water and higher WFPS when the top soil was dry.

NZ-DNDC also generally simulated well the average daily N_2O fluxes from the control and grazed plots compared with the measured daily N_2O fluxes for the corresponding periods (Figures 5a, b and 6a, b). However, it underestimated emissions measured during winter, and slightly overestimated spring emissions. The model also underestimated the very high emissions observed at both grazed sites in summer due to high temperature and moisture after a rainfall event.

The most probable reasons for these discrepancies between measured and simulated data are: The current NZ-DNDC model does not simulate the water-saturated soil conditions observed in the Tokomaru silt loam soil. This could be because of the inability of the hydraulic model to account for the fragipan effect common in this soil (Scotter et al. 1979).

The model was originally developed for North American cropping soils and does not consider the high annual root and litter C inputs of established New Zealand grazed pastures. Therefore the current NZ-DNDC model simulated a decrease in soil C levels during the year (data not reported), whereas in New Zealand pastures soil C levels are generally at steady-state (Tate et al. 2003).

The model does not reflect the seasonal differences in pasture growth and N uptake (data not reported), including slow winter growth and a spring flush, commonly observed in New Zealand pastures (Saggar and Hedley 2001).

The model had limited success in predicting the size and timing of very high emissions observed immediately after a summer rainfall event.

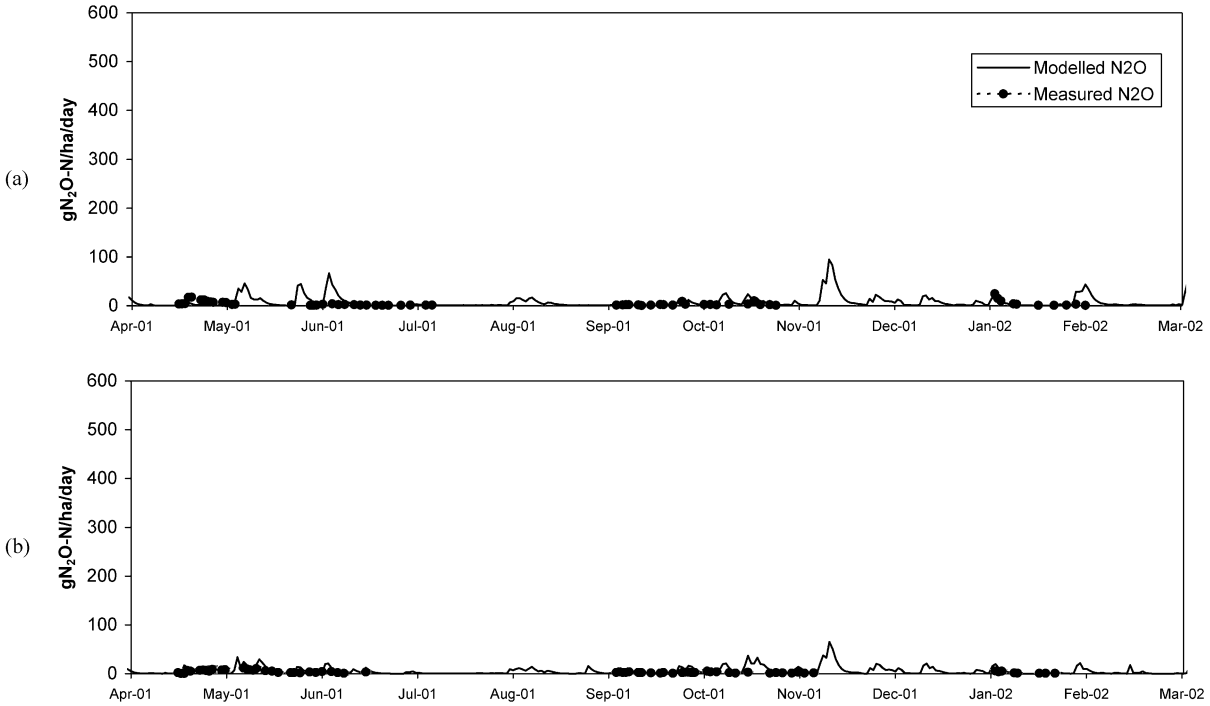

Figure 5. Measured and simulated N_2O fluxes in (a) Karapoti fine sandy loam (Dairy 1) and (b) Tokomaru silt loam (Dairy 3) ungrazed soils during autumn, winter and spring seasons.

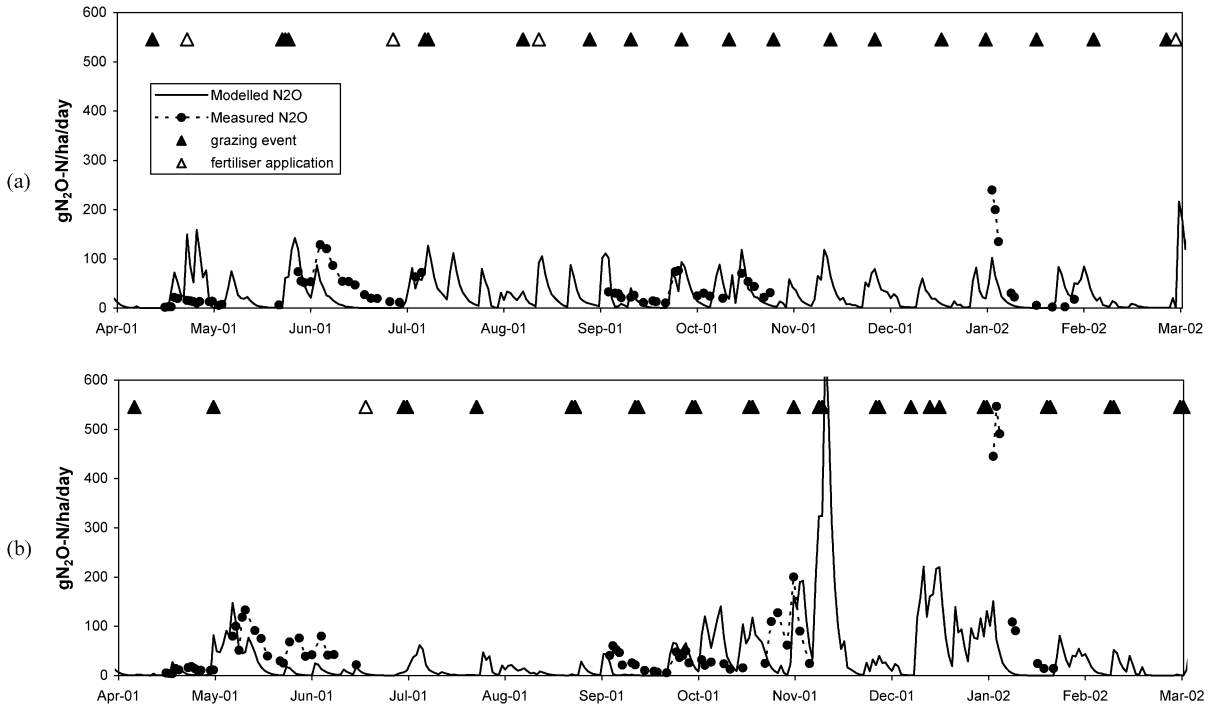

Figure 6. Measured and simulated N_2O fluxes in (a) Karapoti fine sandy loam (Dairy 1) and (b) in the Tokomaru silt loam (Dairy 3) grazed soils during autumn, winter and spring seasons.

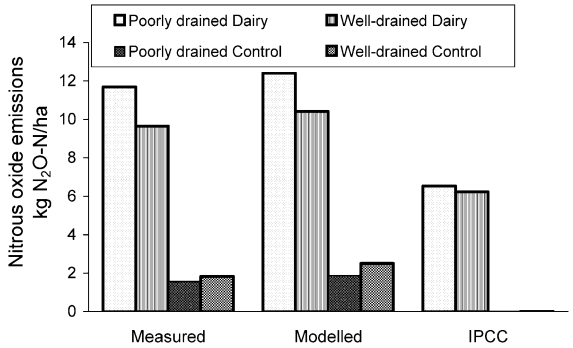

Figure 7. Annual-measured, model-predicted and IPCC-calculated nitrous oxide emissions from the two ungrazed and dairy-grazed sites.

General discussion

Overall, modified NZ-DNDC model was very well able to predict the annual measured N_2O emissions from both the grazed and ungrazed farms (Figure 7). Annual net emissions of 2.53 and 1.88 kg N_2O-N ha^{-1} (ungrazed areas), and 10.41 and 12.40 kg N_2O-N ha^{-1} (grazed areas) were predicted for the fine sandy loam and silt loam soils, respectively. The largest weekly emissions of 0.90 and 2.08 kg N_2O-N ha^{-1} were measured from the fine sandy loam and silt loam soils, respectively, in January (summer) after a heavy rainfall and under high soil temperatures (17.7–20.1 °C). Very high emissions in summer are uncommon in our grazed pasture systems and the model had limited success in predicting their size and timing; the most likely reason for the model grossly underestimating these N_2O emissions is that it under-predicted the temperature effect. Despite this inconsistency, the overall comparisons of predicted and measured annual emissions (Figure 7) indicate NZ-DNDC should be applicable to the simulation of N_2O emissions from a range of New Zealand grazed pastures. More testing is now needed on a range of different soils, and with sheep as well as other cattle-grazed pastoral systems.

Grazing by dairy cattle and fertiliser N applications markedly increased total N_2O emissions in both soils. The excretal and fertiliser N inputs during the measurement period were 396 kg N ha^{-1} in the Karapoti fine sandy loam soil and 345 kg N ha^{-1} in the Tokomaru silt loam soil. Anthropogenic emissions (grazed−ungrazed) were 1.99% of the annual excretal and fertiliser N inputs in the well-drained Karapoti fine sandy loam soil, and 2.53% in the poorly drained Tokomaru silt loam soil. In the soils of both grazed sites, N_2O emissions were mainly produced by denitrification.

The modified model (NZ-DNDC) simulated changes in WFPS very well in the top 5 cm for the fine sandy loam soil but poorly for the silt loam soil. The N_2O emissions simulated by NZ-DNDC generally show the same temporal trends as measurements from both ungrazed and grazed pastures in these commercial dairy farms. Despite the high inherent variability of N_2O emissions from soils, and increased variability in grazed pasture soils caused by uneven return of excreta from dairy cattle, the NZ-DNDC model prediction of N_2O fluxes from ungrazed and grazed pastures correspond well with the actual fluxes measured with the chambers. NZ-DNDC estimated an annual emission for both the farms within 10% of the measured values, and within the uncertainty range of the measured values. The modified model does well in estimating the magnitude of N_2O emissions, and fairly reproduces some of the variability in underlying processes regulating N_2O emissions. In contrast, anthropogenic emission estimates based on the New Zealand refined IPCC methodology (Climate Change Project 2002) were about 25 to 60% lower than for our measured values in the grazed pastures. The NZ-DNDC model was, therefore, better at predicting N_2O emissions than the refined IPCC methodology.

In order to achieve more reliable field-scale estimates of N_2O emissions further model refinement is needed to account for slow winter pasture growth and N uptake and the flush of spring pasture growth. The model, having been originally developed for cropping systems, does not reflect that soil C levels in our established grazed pastures are generally at steady-state, which has a major impact on mineralisation rates. Also, other model parameters such as K_{sat} (saturated hydraulic conductivity), *LAI* (leaf area index) and the weather-controlled evapotranspiration of pastures need to be more carefully characterised to account for the saturated soil conditions observed in the Tokomaru silt loam soil, and to improve the robustness of NZ-DNDC to predict N_2O emissions in New Zealand pastoral systems. The ultimate goal is to be able to estimate emissions accurately on a regional and national scale, based on available climatic data, soil types, and numbers and types of grazing animals and their excretal N inputs. NZ-DNDC appears to capture the key processes controlling N_2O emissions, and offers a robust platform for future achievement of this goal.

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References

Ball B.C., Horgan G.W., Clayton H. and Parker J.P. 1997. Spatial variability of nitrous oxide fluxes and controlling soil and topographic properties. J. Environ. Qual. 26: 1399–1409.

Bouwman A.F. 1990. Exchange of greenhouse gases between terrestrial ecosystems and atmosphere. In: Bouwman A.F. (ed.), Soils and Greenhouse Effect. Wiley, Chichester, UK, pp. 61–127.

Brown L., Armstrong B.S., Jarvis S.C., Syed B., Goulding K.W.T., Phillips V.R., Sneath R.W. and Pain B.F. 2001. An inventory of nitrous oxide emissions from agriculture in the UK using the IPCC methodology: emission estimates, uncertainty and sensitivity analysis. Atmospheric Environ. 35: 1439–1149.

Brown L., Syed B., Jarvis S.C., Sneath R.W., Phillips V.R., Goulding K.W.T. and Li C. 2002. Development and application of a mechanistic model to estimate emissions of nitrous oxide from UK agriculture. Atmospheric Environ. 36: 917–928.

Butterbach-Bahl K., Stange F., Papen H. and Li C. 2001. Regional inventory of nitric oxide and nitrous oxide emissions for forest soils of Southeast Germany using the biogeochemical model PnET-N-DNDC. J. Geophys. Res. 106: 34155–34165.

Cameron K.C., Kelliher F.M., Sherlock R.R., Forbes E.A., deClein C.A.M. and other members of NzOnet 2000. Nitrous oxide inventory and mitigation. A National Science Strategy Research Programme for New Zealand – A report for MAF-Policy. Centre for Soil and Environmental Quality, Lincoln University, USA, p. 146.

Carran R.A., Theobald P.W. and Evans J.P. 1995. Emission of nitrous oxide from some grazed pasture soils in New Zealand. Austr. J. Soil. Res. 33: 341–352.

Castaldi S. and Smith K.A. 1998. The effect of different N substrates on biological N_2O production from forest and agricultural light textured soils. Plant Soil 199: 229–238.

Choudhary M.A., Akramkhanov A. and Saggar S. 2002. Nitrous oxide emissions in New Zealand cropped soils: tillage effects, spatial and seasonal variability. Agric. Ecosyst. Environ. 93: 33–43.

Davidson E.A. 1991. Fluxes of nitrous oxide and nitric oxide from terrestrial ecosystems. In: Rogers J.E. and Whitman W.B. (eds), Microbial Production and Consumption of Greenhouse Gases: Methane, Nitrogen Oxides, and Halo-Methanes. Am. Soc. Microbiol., Washington, DC, USA, pp. 219–235.

Del Grosso S., Parton W., Mosier A., Hartman M.D., Brenner J., Ojima D.S. and Schimel D. 2001. Simulated interaction of carbon dynamics and nitrogen trace gas fluxes using the DAYCENT model. In: Schaffer M., Ma L. and Hansen S. (eds), Modeling Carbon and Nitrogen Dynamics for Soil Management. CRC Press, Boca Raton, FL, USA, pp. 303–332.

Dobbie K.E. and Smith K.A. 2001. The effect of temperature, water-filled pore space, and land use on N_2O emissions from an imperfectly drained gleysol. Eur. J. Soil Sci. 52: 667–673.

Dobbie K.E., McTaggart I.P. and Smith K.A. 1999. Nitrous oxide emissions from intensive agricultural systems: variations between crops and seasons, key driving variables, and mean emission factors. J. Geophys. Res. 104: 26891–26899.

Folorunso O.A. and Rolston D.E. 1984. Spatial variability of field-measured denitrification gas fluxes. Soil. Sci. Soc. Am. J. 48: 1214–1219.

Frolking S.E., Mosier A.R., Ojima D.S., Li C., Parton W.J., Potter C.S., Priesack E., Stenger R., Haberbosch C., Dörsch P., Flessa H. and Smith K.A. 1998. Comparison of N_2O emissions from soils at three temperate agricultural sites: simulations of year-round measurements by four models. Nutr. Cycling Agroecosyst. 52: 77–105.

Haynes R.J. and Williams P.H. 1993. Nutrient cycling and soil fertility in grazed pasture ecosystems. Adv. Agron. 49: 119–199.

Hedley C., Saggar S. and Dando J. 2002. Use of intact soil cores to assess the effect of soil water content on nitrous oxide emissions from a poorly drained pasture soil. In: Back to the Future, New Zealand Soil Science Society Golden Jubilee Conference, Programme and Abstracts, 25–29 November 2002. Victoria University, Wellington, New Zealand, p. 85.

IPCC (Intergovernmental Panel on Climate Change/Organization for Economic Cooperation and Development) 1997. Guidelines for National Greenhouse Gas Inventories. OECD/OCDE, Paris, France.

IPCC 2001. Technical summary. In: Houghton J.T., Ding Y, Grigg D.J., Noguer M., van der Linden P.J., Dai X., Maskell K. and Johnson C.A. (eds), Climate change 2001: the scientific basis. Contributions of Working Group I of the Intergovernmental Panel on Climete Change. Cambridge University Press, Cambridge, UK.

Ineson P., Coward P.A. and Hartwig U.A. 1998. Soil gas fluxes of N_2O, CH_4 and CO_2 beneath Lolium perenne under elevated CO_2: the Swiss free air carbon dioxide enrichment experiment. Plant Soil 198: 89–95.

Kaiser E.A., Kohrs K., Kucke M., Schnug E., Heinemeyer O. and Munch J.C. 1998. Nitrous oxide release from arable soil: importance of N-fertilization, crops and temporal variation. Soil Biol. Biochem. 30: 1553–1563.

Ledgard S.F., Penno J.W. and Sprosen M.S. 1999. Nitrogen inputs and losses from grass/clover pastures grazed by dairy cows, as affected by N fertilizer application. J. Agric. Sci. 132: 215–225.

Li C., Frolking S. and Frolking T.A. 1992a. A model of nitrous oxide evolution from soil driven by rainfall events: 1. Model structure and sensitivity. J. Geophys. Res. 97: 9759–9776.

Li C., Frolking S. and Frolking T.A. 1992b. A model of nitrous oxide evolution from soil driven by rainfall events: 2. Model applications. J. Geophys. Res. 97: 9777–9783.

Li C., Narayanan V. and Harris R. 1996. Model estimates of nitrous oxide emissions from agricultural lands in the United States. Global Biogeochem. Cycles 10: 297–306.

Li C., Zhuang Y., Cao M., Crill P., Dai Z., Frokling S., Moore B., Salas W., Song W. and Wang X. 2001. Comparing a national inventory of N_2O emissions from arable lands in China developed with a process-based agroecosystem model to the IPCC methodology. Nutr. Cycling Agroecosyst. 60: 159–175.

MacKenzie A.F., Fan M.X. and Cadrin F. 1998. Nitrous oxide emission in three years as affected by tillage, corn-soybean-alfalfa rotations, and nitrogen fertilization. J. Environ. Qual. 27: 698–703.

Mosier A.R. and Mack L. 1980. Gas chromatographic system for precise, rapid analysis of nitrous oxide. Soil Sci. Soc. Am. J. 44: 1121–1123.

Muller C., Sherlock R.R. and Williams P.H. 1995. Direct field measurements of nitrous oxide emissions from urine-affected and urine-unaffected pasture in Canterbury. In: Currie L.D. and Loganathan P. (eds), Fertiliser Requirements of Grazed Pasture and Filed Crops: Macro and Micronutrients. Occasional Report No. 8. Massey University, Plamerston North, pp. 243–247.

New Zealand Climate Change Project 2002. The National Inventory report, New Zealand Greenhouse Gas Inventory 1990–2000 including common reporting format (CRF) for 2000, April 2002. New Zealand Climate Change Project, Wellington, New Zealand (http://www.climatechange.govt.nz).

Oenema O., Velthof G.L., Yamulki S. and Jarvis S.C. 1997. Nitrous oxide emissions from grazed grassland. Soil Use Manage. 13: 288–295.

Parton W.J., Mosier A.R., Ojima D.S., Valentine D.W., Schimel D.S., Weier K. and Kulmala A. E. 1996. Generalised model for N_2 and N_2O production from nitrification and denitrification. Global Change Biol. 10: 401–412.

Potter C.S., Randerson J.T. and Field C.B. 1993. Terrestrial ecosystem production: a process model based on global satellite and surface data. Global Biogeochem. Cycles 7: 811–841.

Potter C.S., Matson P.A., Vitousek P.M. and Davidson E.A. 1996. Process modeling of controls of nitrogen trace gas emissions from soils Worldwide. J. Geophys. Res.- Atmos. 101: 1361–1377.

Quikchem Methods Manual 2001. Lachet Instruments Division. Zellweger Analytics Inc., Milwaukee, WI, USA.

Robertson G.P. and Tiedje J.M. 1987. Nitrous oxide sources in aerobic soils; nitrification, denitrification and other biological processes. Soil Biol. Biochem. 19: 187–193.

Ruz-Jerez B.E., White R.E. and Ball P.R. 1994. Long-term measurement of denitrification in three contrasting pastures grazed by sheep. Soil Biol. Biochem. 26: 29–39.

Saggar S. 2001. Soil Science at the cross roads – Norman Taylor Memorial Lecture 2001. New Zealand Soil News 49 (6): 132–139.

Saggar S. and Hedley C.B. 2001. Estimating seasonal and annual carbon inputs and root decomposition rates in a temperate pasture following field ^{14}C pulse-labelling. Plant Soil 236: 103–201.

Scotter D.R., Clothier B.E. and Corker R.B. 1979. Soil water in a fragiaqualf. Austr. J. Soil. Res. 17: 443–453.

Sherlock R.R., Johnston G., Kelliher F.M., Newsome P.F. and Walcroft A. 2001. A desktop study of regional variations in nitrous oxide emissions. A report prepared for the Ministry of Agriculture and Forestry. Lincoln University, USA, p. 53.

Smith W.N., Desjardins R.L., Grant B., Li C., Lemke R., Rochette P., Corre M.D. and Pennock D. 2002. Testing the DNDC model using N_2O emissions at two experimental sites in Canada. Can. J. Soil Sci. 82: 365–374.

Tate K.R., Scott N.A., Saggar S, Giltrap D.J., Baisden W.T., Newsome P.F., Davis M.R. and Wilde R.H. 2003. Land-use change alters New Zealand's terrestrial carbon budget: reducing uncertainties associated with estimates of soil carbon change. Tellus 55B: 364–367.