Biogeochemistry (2005) 76: 395–419
DOI 10.1007/s10533-005-6996-8

Simulation of effects of soils, climate and management on N_2O emission from grasslands

DMITRI CHATSKIKH[1,*], JØRGEN E. OLESEN[1], JØRGEN BERNTSEN[1], KRISTIINA REGINA[2] and SIRWAN YAMULKI[3]

*[1]Department of Agroecology, Danish Institute of Agricultural Sciences, Research Centre Foulum, PO Box 50, 8830 Tjele, Denmark; [2]Environmental Research, Agrifood Research Finland (MTT), FIN-31600 Jokioinen, Finland; [3]North Wyke Research Station, Institute of Grassland and Environmental Research, Okehampton Devon EX20 2SB, UK; *Author for correspondence: (e-mail: Dmitri.Chatskikh@agrsci.dk; phone: +45-89991843; fax: +45-89991619)*

Received 19 July 2004; accepted in revised form 9 May 2005

Key words: Climate, Grassland, Greenhouse gas emissions, Management, Model, N_2O, Nitrous oxide, Soil

Abstract. Nitrous oxide (N_2O) is a potent greenhouse gas with a high contribution from agricultural soils and emissions that depend on soil type, climate, crops and management practices. The N_2O emissions therefore need to be included as an integral part of environmental assessments of agricultural production systems. An algorithm for N_2O production and emission from agricultural soils was developed and included in the FASSET whole-farm model. The model simulated carbon and nitrogen (N) turnover on a daily basis. Both nitrification and denitrification was included in the model as sources for N_2O production, and the N_2O emissions depended on soil microbial and physical conditions. The model was tested on experimental data of N_2O emissions from grasslands in UK, Finland and Denmark, differing in climatic conditions, soil properties and management. The model simulated the general time course of N_2O emissions and captured the observed effects of fertiliser and manure management on emissions. Scenario analyses for grazed and cut grasslands were conducted to evaluate the effects of soil texture, climatic conditions, grassland management and N fertilisation on N_2O emissions. The soils varied from coarse sand to sandy loam and the climatic variation was taken to represent the climatic variation within Denmark. N fertiliser rates were varied from 0 to 500 kg N ha^{-1}. The simulated N_2O emissions showed a non-linear response to increasing N rates with increasing emission factors at higher N rates. The simulated emissions increased with increasing soil clay contents. N_2O emissions were slightly increased at higher temperatures, whereas increasing annual rainfall generally lead to decreasing emissions. Emissions were slightly higher from grazed grasslands compared with cut grasslands at similar rates of total N input (fertiliser and animal excreta). The results indicate higher emission factors and thus higher potentials for reducing N_2O emissions for intensively grazed grasslands on fine textured soils than for extensive cut-based grasslands on sandy soils.

Introduction

Grasslands play a major role at both national and continental scales for the total nitrous oxide (N_2O) emission into the atmosphere (Jarvis 1997). The global emission of N_2O from grasslands is estimated at about 2.5 Tg N y^{-1}, comprising 18% of total N emission (Lee et al. 1997).

N_2O is a potent greenhouse gas affecting the radiation balance of the earth (Houghton et al. 2001), and it also contributes to depletion of stratospheric ozone (Cicerone 1987). The emissions of greenhouse gases are regulated by the Kyoto Protocol under UNFCCC (United Nations Framework Convention for Climate Change). Nations under UNFCCC have to perform inventories of greenhouse gases, including N_2O, using the IPCC methodology (IPCC 1997). Based on experimental data from UK and USA, a N_2O emission factor of 12.5 g N_2O-N per kg of N input was estimated (Bouwman 1996), and this emission factor is presently recommended by IPCC. Nevertheless, nations under UNFCCC are allowed to use locally adapted emission factors, provided sufficient documentation exists (IPCC 2001).

Soil N_2O production is influenced by a range of microbiological, chemical and physical soil processes and properties (Venterea and Rolston 2002). Nitrification and denitrification are generally believed to be the most important microbiological sources for N_2O production in the soil (Maag and Vinther 1996). In addition, N_2O is produced by chemo-denitrification in neutral and acidic soils, through a number of physical and chemical reactions leading to N_2O formation largely controlled by soil pH (Bremner 1997). After a physical redistribution of N_2O between water and air phases within the soil pore space, gaseous N_2O is emitted into the atmosphere or transported as dissolved N_2O via groundwater (Yoh et al. 1997).

The emission of N_2O from grasslands is closely linked to biogeochemical and physical properties of the soils through microbiological processes (Conrad 1996) and affected by climatic conditions, type and rate of fertilisers/manure and sward management (Frolking et al. 1998; de Klein et al. 2001). The complex interactions between soil processes under different environmental conditions and crop management makes it difficult to draw firm conclusions about emission estimates from individual experiments. Modelling may therefore serve as a tool for interpreting experimental results and extrapolating to new environmental and management conditions (Smith et al. 1997).

A number of simulation studies of N_2O production and transport through the soil profile have been published (e.g. Li et al. 1992; Parton et al. 1996; Potter et al. 1997; Schmid et al. 2001). The models describe the production and transport of N_2O at hourly, daily or finer time scales, which is necessary to capture the environmental influences on N_2O production. The models also describe microbiological processes leading to N_2O emissions from the soil based on Michaelis-Menten-Monod or zero/first-order kinetics. However, there are other models in which N_2O emission is described through physical distribution, diffusion and leaching of N_2O through the soil profile (Langeveld et al. 1997).

Simulation of the N_2O emission from grasslands is difficult, not only because of the interlinked processes but also due to uncertainties in the accuracy of experimental data for N_2O emission, which hamper the development and testing of models (Ambus and Christensen 1995; Frolking et al. 1998; Parton et al. 2001). The sensitivity of the measurement technique, spatial and temporal variability of the N_2O fluxes and problems with determining the gases in

certain cases may lead to difficulties in obtaining reliable estimates of whole field N_2O emissions (van den Pol-van Dasselaar et al. 1998; Rudaz et al. 1999; Williams et al. 1999). Emissions of N_2O from soils is strongly dependent on soil and crop management, which for most livestock farms not only depends on the soil and the crop rotation, but also on the livestock and manure management, particularly for livestock systems with grazed pastures.

The FASSET whole-farm model includes all major N-flows at the farm level including a detailed dynamic simulation of the soil–plant-climate system at the field level (Olesen et al. 2002). This makes the model suitable for evaluating environmental consequences of changes in farm management (Berntsen et al. 2003). This environmental evaluation should include all relevant N losses from agriculture, including N leaching, ammonia volatilisation, N_2 and N_2O emission. The objective of the work presented here was to extend the FASSET model with an algorithm for estimating N_2O emissions from soils, and to evaluate the effects of variation in management, soils and climate for N_2O emissions from grasslands in Denmark.

Materials and methods

FASSET model

The FASSET soil–plant-atmosphere model was taken as the basis for the implementation of a denitrification and N_2O emission model. The model, which applies a daily time step, simulates N turnover and crop production as affected by daily weather and availability of water and N (Olesen et al. 2002).

The soil model has a one-dimensional vertical structure and simulates daily movement and plant availability of water and N. The soil is divided into horizontal layers, and the transport of water and N between the soil layers is calculated using the concepts of the Solute Leaching Intermediate Model, SLIM (Addiscott and Whitmore 1991). The soil organic matter (SOM) model is based on Petersen et al. (2005a, b). Briefly, an organic residue that enters the model is split into two organic pools (AOM1 and AOM2) depending on the type and quality of the residue. The AOM pools are decomposed by two microbial pools (SMB) – a slowly growing SMB1 pool and a faster growing SMB2 pool. Residues from the SMB2 pool goes to a soil microbial residue pool (SMR), while residues from the SMB1 pool goes to a SOM pool (NOM). Part of the soil organic matter is inert (IOM). Simulations of temperature, water content, nitrification, potential mineralisation, and ammonium and nitrate concentration in the soil layers were taken as input for the denitrification and N_2O emission model. Simulation of grazing and growth of grass-clover was done according to Berntsen et al. (2005).

N_2O modelling

The N_2O emission model is based on the ''Holes-In-the-Pipe'' scheme (Davidson et al. 2000), where the N intermediates from the nitrification and denitrification processes are assumed to be sources for N_2O production. Neither chemo-denitrification nor nitrifier denitrification are considered as sources for N_2O in the model.

Two steps are used to calculate the N_2O emission. Firstly, for each soil layer the N_2O production potential is determined from the simulated nitrification and denitrification by applying semi-empirical functions to estimate effects of environmental factors. Secondly, the potential N_2O emission is divided into N_2 and N_2O emission using semi-empirical relations to estimate the effects of soil physical properties and soil diffusion (layer depth) on the efficiency by which denitrification reduces N_2O to N_2. The actual N_2 and N_2O emissions are then calculated by summing the contributions from each soil layer.

N_2O production potential

Both nitrification and denitrification are affected by several environmental factors from which soil temperature is the most important factor (Bouwman 1996). The temperature response for both processes was taken from Kirschbaum (1995) and Petersen et al. (2005a):

$$F_T(T) = a_u \exp(b_u + c_u T[1 - 0.5T/d_u]) \tag{1}$$

where a_u, b_u, c_u, d_u (a_u = 7.24, b_u = −3.432, c_u = 0.168, d_u = 36.9) are parameters, and T is mean daily soil temperature (°C).

The potential N_2O production $\xi^*_{N_2O}$ (g N m^{-2} d^{-1}) is assumed to be the sum of denitrification ξ_d (g N m^{-2} d^{-1}) and a proportion of nitrification ξ_n (g N m^{-2} d^{-1}) rate:

$$\xi^*_{N_2O} = k_{np} F_{nT}(T) Q_{wfp} \xi_n + \xi_d \tag{2}$$

where k_{np} represents the proportion of N intermediates resulting in N_2O emissions from nitrification. Q_{wfp} is the soil water-filled porosity (v v^{-1}), $Q_{wfp} = \theta/P$, where θ is volumetric soil water content (v v^{-1}) and P is total soil porosity (v v^{-1}). The temperature function $F_{nT}(T)$ was parameterised using experimental data from Ingwersen et al. (1999) and the temperature function for nitrification from Li et al. (2000):

$$F_{nT}(T) = \min[1, \exp(-0.5((T - 2a_n)/a_n)^2)] \tag{3}$$

where a_n (a_n = 17.1) is a constant.

The soil nitrification is described by first-order kinetics modified by soil temperature and soil water potential:

$$\xi_n = k_n F_T(T) F_\psi(\psi) C_{NH_4} \tag{4}$$

where C_{NH_4} is the soil ammonium content (g N m^{-2}), $F_\psi(\psi)$ is the soil water function and k_n is the potential nitrification rate under optimal conditions, which is set to 0.10 d^{-1} (Hansen et al. 1990).

The soil water function $F_\psi(\psi)$ is taken from Petersen et al. (2005a):

$$F_\psi(\psi) = \begin{cases} 0.6 & \psi \geq -9.81 \cdot 10^{-5} \\ 0.6 + 0.4 \cdot \log_{10}(-\psi/9.81 \cdot 10^{-5})/1.5 & -9.81 \cdot 10^{-5} > \psi \geq -3.1 \cdot 10^{-3} \\ 1.0 & -3.1 \cdot 10^{-3} > \psi \geq -3.1 \cdot 10^{-2} \\ 1.0 - \left(\log_{10}(-\psi/9.81 \cdot 10^{-5}) - 2.5\right)/3.0 & -3.1 \cdot 10^{-2} > \psi \geq -3.1 \cdot 10^{2} \\ 0.0 & -3.1 \cdot 10^{1} > \psi \end{cases} \quad (5)$$

where ψ is soil water potential (m H_2O).

The denitrification potential ξ^*_d (g N $m^{-2}d^{-1}$) was assumed to be proportional to the SOM mineralisation rate ξ^*_m (g C $m^{-2}d^{-1}$), and to depend on soil clay content:

$$\xi^*_d = (a_d + b_d \text{ Clay})\xi^*_m \quad (6)$$

where a_d and b_d (a_d = 0.151 $_gN_gC^{-1}$, b_d = 0.015 $_gN_gC^{-1}$) are constants estimated from Drury et al. (1991), Clay is clay content (%), and the potential mineralisation rate is estimated by the FASSET SOM model using first-order kinetics (Petersen et al. 2005b).

The denitrification rate was assumed to depend on soil temperature, water-filled porosity, and nitrate concentration C_{NO_3} (mg N kg^{-1}):

$$\xi_d = \xi^*_d F_T(T) F_Q(Q_{wfp}) F_N(C_{NO_3}) \quad (7)$$

where $F_Q(Q_{wfp})$ defines the dependency of denitrification on water-filled porosity, and this function was estimated from data by Del Grosso et al. (2000):

$$F_Q(Q_{wfp}) = \max\{0, \min[1, a_Q + b_Q/(1 + \exp(-(Q_{wfp} - c_Q)/d_Q))]\} \quad (8)$$

where a_Q, b_Q, c_Q and d_Q (a_Q = 0.0116, b_Q = 1.36, c_Q = 0.815, d_Q = 0.0896) are constants. $F_Q(Q_{wfp})$ has three different fragments. When Q_{wfp} is less than ca. 0.6 the function has a value near zero, which, possibly reflects a good aeration of the soil. For Q_{wfp} above ca. 0.6 the function increases exponentially up to a maximum value of 1, which is attained when Q_{wfp} has a value of about 0.9. The nature of the function is in agreement with experimental data by Grundmann and Rolston (1987) and Johnsson et al. (1991).

$F_N(C_{NO_3})$ describes the dependency of the denitrification rate on nitrate concentration using the Langmuir's isotherm:

$$F_N(C_{NO_3}) = \max\{0, \min[1, a_N C_{NO_3}/(b_N + C_{NO_3})]\} \quad (9)$$

where a_N and b_N ($a_N = 1.17$, $b_N = 32.7$) are constants, which were estimated from experimental studies (Wijler and Delwiche 1954; Nõmmik 1956; Weier et al. 1993; Henault and Germon 2000).

N_2O emission

The actual N_2O emission ξ_{N_2O} (g N m^{-2} d^{-1}) was estimated as the N_2O production potential $\xi^*_{N_2O}$ modified by functions of environmental variables and soil depth, as the ratio of N_2 to N_2O depends on reduction potential in the soil and on the diffusion path to the soil surface (Yoh et al. 1997):

$$\xi_{N_2O} = \xi^*_{N_2O} F_{NT}(T)\big(1 - F_Q(Q_{wfp})\big) F_C(\text{Clay}) F_D(\text{Depth}) \tag{10}$$

The empirical temperature function $F_{NT}(T)$ was based on Vinther (1990):

$$F_{NT}(T) = 1/(1 + \exp(a_T + b_T T)) \tag{11}$$

where a_T and b_T ($a_T = -0.64$, $b_T = 0.08$) are constants.

The N_2O emission profile with soil depth F_D(Depth) was taken from data for a loam soil (Yasukazu et al. 2000), and adapted to soils with different clay content F_C(Clay) using the data from Letey et al. (1980):

$$F_D(\text{Depth}) = \max\{0, \min[1, \mathrm{a_D} - \mathrm{b_D}\ \text{Depth}\ - \mathrm{c_D}\ \text{Depth}^2]\} \tag{12}$$

$$F_C(\text{Clay}) = \max\{0, \min[1, \mathrm{a_C} \exp(\mathrm{b_C}\ \text{Clay}\ ) - \mathrm{c_C})]\} \tag{13}$$

where Depth is soil depth (m) and a_D, b_D and c_D ($a_D = 1.0008$, $b_D = 0.0343$, $c_D = 3.1816$), a_C, b_C and c_C ($a_C = 1.26$, $b_C = -0.0116$, $c_C = 0.249$) are constants.

The N_2 emission ξ_{N_2} (g N m^{-2} d^{-1}) was then estimated as the difference between potential and actual N_2O emissions:

$$\xi_{N_2} = \xi^*_{N_2O} - \xi_{N_2O.} \tag{14}$$

Experimental data for model evaluation

Experimental data from three North European sites (Table 1) were used for evaluating the N_2O emission model. The data included measurements of soil water and mineral N contents and N_2O emissions from the soils. Additionally, data on soil texture, initial soil C and N contents, daily meteorological data and grassland management were used in the model simulations.

Site 1. An experiment at Jægersborg in 1981 on a grass lawn (soil clay content 9%) with three treatments: control (UN) with no fertilisation, application of mineral N (MI) with 200 kg N ha^{-1} of NH_4NO_3 twice a year, and injection of cow slurry (SL) with a total 492 kg N ha^{-1} (Christensen

Table 1. Location and normal annual climatic conditions for the experimental sites used for model validation.

Site	Location	Geographical coordinates	Precipitation (mm)	Temperature (°C)
Jægersborg	East Denmark	56.50° N, 09.58° E	630	7.6
Jokioinen	South Finland	60.82° N, 23.50° E	636	5.7
Devon	South-West UK	50.70° N, 04.87° W	1141	10.3

1983b). The N_2O fluxes were determined by drawing air through chambers inserted into the soil surface and measuring N_2O content by gas chromatography (Christensen 1983a). The total N_2O emissions during May to August were calculated from hourly measurements.

Site 2. Three years of continuing N_2O measurements at Jokioinen on cut grassland on a clay (CL) and a loamy sand (LS) soil, with clay contents of 57.3 and 9.7% and SOM contents of 5.2 and 5.4%, respectively (Pihlatie et al. 2004; Syväsalo et al. 2004). Mineral N was applied at rates of 200–250 kg N ha^{-1} in two to three annual applications. Emissions of N_2O were measured 2–6 times per month from three replications during July 2000 to August 2002 with opaque chambers. Based on these values the seasonal N_2O emission budgets during the 2 years were estimated by linear interpolation.

Site 3. Measurements of N_2O emissions were performed in 2002 at 2 farms in Devon on permanent grasslands with intensive grazing under conventional (CF) and organic (OF) management (Petersen et al. 2005c). The N applications were 65 kg N ha^{-1} in animal manure and 25 kg N ha^{-1} from grazing for the OF treatment and 78 kg N ha^{-1} in mineral N fertiliser, 187 kg N ha^{-1} in animal manure and 113 kg N ha^{-1} from grazing for the CF treatment. The soil type was loam with a clay content of ca. 20%, and the C:N ratio of SOM was 9.3 and 8.3 for CF and OF, respectively. The N_2O emission was measured in six replicates approximately twice a month during February to November 2002 with the static chamber technique. Seasonal N_2O emissions were estimated by linear interpolation.

Validation procedure

The FASSET model was used to simulate N_2O emissions from the three European experimental grassland sites using the observed management (fertilisers/manure, grazing and cutting). Grazing was simulated as described below for the scenario analyses. Standard long-term daily meteorological data (minimum and maximum temperature, precipitation and global radiation) were obtained for the Danish, Finnish and UK sites. The soil hydraulic properties were predicted using pedotransfer functions based on the HYPRES database of European soils (Wösten et al. 1999). These semi-empirical func-

tions were used in the FASSET model as parameters for the Mualem–van Genuchten equation (Schaap and Leij 2000).

The model behaviour was checked by comparing observed and simulated values of crop biomass, yield, soil water and soil mineral N, which indicated no systematic deviations (data not shown). The model's ability to predict the observed N_2O emissions was estimated by regression analysis. In addition, the root mean squared error (RMSE), the coefficient of model efficiency (EF), the coefficient of model determination (CD) and the sample correlation test (*r*-test) were calculated according to Smith et al. (1997).

Scenario analysis

The response of simulated N_2 and N_2O emissions to changes in grassland management and variation in climatic and soil conditions were evaluated for a range of scenarios.

The calculations were performed for a crop rotation with 1 year of spring barley undersown with ryegrass followed by 2 years of grass. This is the typical way of establishing rotational grassland in Denmark. The emission results were taken from the second year grassland only, and the two first years of simulations were thus used for initialising soil pools. The grasslands were either grazed with heifers or treated as a cut-based system (four cuts per year) with removal of the grass. During grazing the heifers were assumed to remove all crop material above 9 cm. Of this material 20% is immediately deposited on the soil surface. This simulates several loss processes, such as spillage during grazing and treading of plants by the cattle. It was assumed that 84% of the N in intake was excreted, while the remaining 16% were used for growth. During the first year 300 kg N ha^{-1} was applied as mineral fertiliser (NH_4NO_3, 50:50), whereas in the second year the applied mineral N ranged from 0 to 500 kg N ha^{-1} in steps of 50 kg N ha^{-1}. The simulations were carried out using climate data from Research Centre Foulum (56.30° N, 9.34° E) with an average annual temperature of 7.3 °C and precipitation of 704 mm for the period 1961–1990 (Olesen et al. 2000) in combination with three different soil textures with topsoil and subsoil textures from Table 2. For estimation of the average emissions of N_2 and N_2O, 30 independent calculations were performed by starting the simulations in different years within the climatic datasets, which spanned the years 1961 to 2004.

Additional scenarios were performed for the cut-based system in which soil organic C, temperature and precipitation were varied. Soil organic C was increased by 0 to 60% relative to the original soil profile. Temperature was changed by adding a constant value to the observed daily temperature, and precipitation was changed by scaling the observed daily precipitation.

Table 2. Soil textures used for the scenario analyses.

Soil texture	Sand 20–2000 μm (%)	Silt 2–20 μm (%)	Clay <2 μm (%)	Soil organic C (%)
Topsoil (0–25 cm)				
Coarse sand	87.5	8.8	3.7	2.19
Loamy sand	66.0	26.2	7.8	2.25
Sandy loam	51.0	30.6	18.4	2.06
Subsoil (25–50 cm)				
Coarse sand	92.7	3.9	3.4	1.29
Loamy sand	68.2	24.5	7.3	1.28
Sandy loam	53.4	26.8	19.8	1.15

Result

N_2O emission model

To determine the k_{np} value, the potential nitrification was estimated from a pasture experiment at Risø (Denmark) twice in April and July 2001 by measurement of nitrification rates and N_2O emission in soil cores using ^{15}N isotope technique (Per Ambus, personal communication). Based on these experimental data, k_{np} was estimated as 0.047. In the scenario analysis the N_2O production by nitrification from loamy sand varied between 0.25 and 0.65% of the nitrification rate, which agreed with experimental values reported by Maag and Vinther (1996). They showed that despite differences between the soils, the N_2O production was generally between 0.28 and 0.48% of the nitrification rate, and Billore et al. (1996) found an interval between 0.64 and 1.00% across several grassland soils.

Experiments

The model simulations captured most of the variation in the measured N_2O emissions, which was caused by the different fertiliser and manure treatments for the Danish and UK sites (Figures 1 and 3). On a seasonal basis there was also a good correspondence between the observed and simulated values (Table 3), both of which in the Danish experiment gave the lowest emissions for the unfertilised treatment (UN), higher emissions for the mineral N treatment (MI) and the highest emissions from the slurry treatment (SL). However, the model underestimated the N_2O peaks from the SL treatment.

For the Finnish site the simulated N_2O emissions generally followed the trends of the measurements for the CL (clay soil) and LS (loamy sand soil) treatments (Figure 2 and Table 3), except for the measured spring peaks in 2001 and 2002, which were not found in the simulated data. However, the observed and simulated values for N_2O emissions broadly agreed on a seasonal basis.

Table 3. Observed and simulated seasonal N_2O emissions (kg N ha^{-1}).

Site	Experiment	Period	Observed	Simulated
Jægersborg	UN	May–Jul 1981	0.43	0.10
	UN	Jul–Aug 1981	0.24	0.12
	MI	May–Jul 1981	1.32	1.52
	MI	Jul–Aug 1981	1.06	1.01
	SL	May–Jul 1981	1.65	0.97
	SL	Jul–Aug 1981	7.70	2.48
Jokioinen	CL	Jul–Dec 2000	0.46	0.80
	CL	Jan–Dec 2001	0.96	1.15
	CL	Jan–Aug 2002	1.33	2.19
	LS	Jul–Dec 2000	0.49	0.41
	LS	Jan–Dec 2001	1.51	0.58
	LS	Jan–Aug 2002	0.82	0.56
Devon	CF	Feb–Nov 2002	5.97	9.82
	OF	Feb–Nov 2002	2.42	2.84

The measurements of N_2O emissions for the grazed grasslands in the UK site showed a large spatial variation, in particular on days with high emissions (Figure 3). There were also large variations between replicates in the observed daily emissions, which the model was not designed to simulate. The simulated seasonal N_2O emissions from the CF treatment were higher than the observed interpolated measurements.

Regression analysis showed that the simulated emissions explained 69% of the variation in observed seasonal emissions. The intercept was not significantly different from 0, and the slope was not significantly different from 1. For all experiments presented in Table 3 the RMSE is equal to 1.79 kg N ha^{-1}, the EF is small but positive, and CD is greater than 1. The *r*-test shows a positive correlation between observed and simulated values (Table 4).

Scenario analysis

The average simulated daily N_2O emissions are shown in Figure 4 for cut and grazed grasslands with application of 200 kg N ha^{-1} in mineral fertiliser (NH_4NO_3, 50:50). The grazed treatment gave higher N_2O emissions throughout the season, even before the start of the grazing period. This reflects the residual effect of the previous year with grazing. The grazing period contributed with 72 and 74% of the annually emitted N_2O for cut and grazed grassland, respectively, which accumulates to 79% for both types of grassland when this period was extend with the first N application. Peaks in the N_2O emission were simulated after each fertiliser application, and most clearly in the cut treatment. The longest average N_2O emission peak was obtained after the last fertiliser application, even though the same amount of mineral N fertiliser was applied at each application. For the grazed grassland, the late spring and

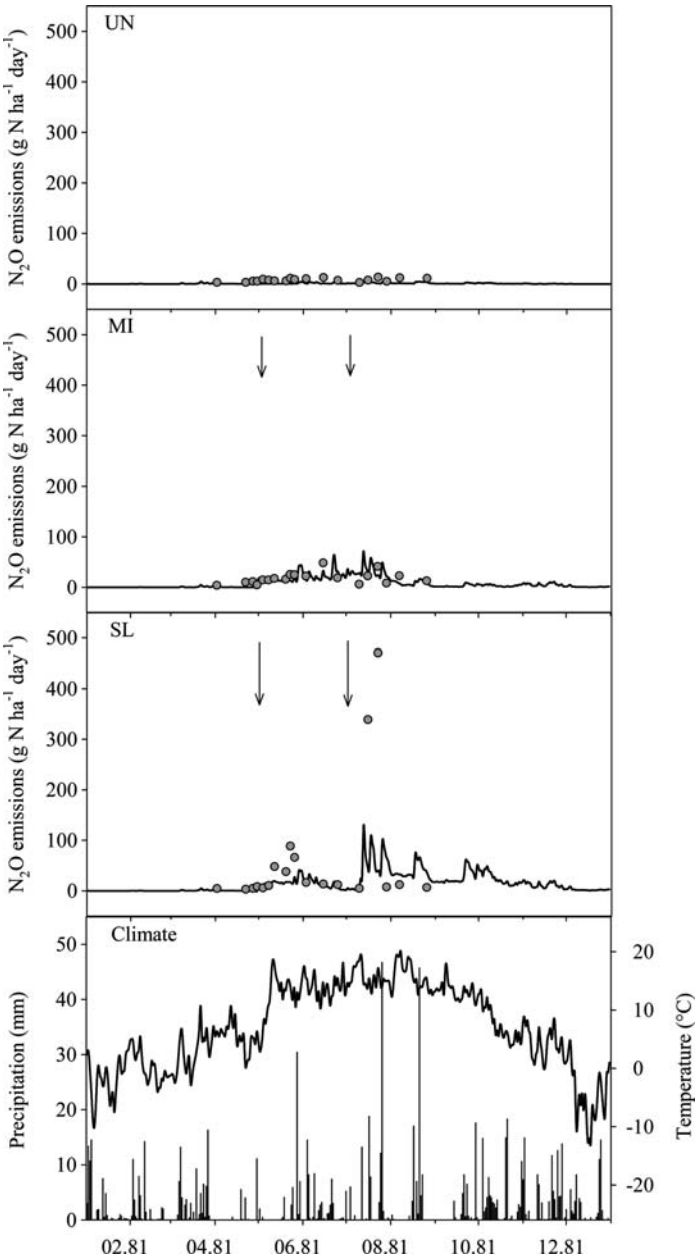

Figure 1. Simulated (lines) and measured (points) daily N_2O fluxes from January to December 1981 for a grass lawn on the Jægersborg site for unfertilised (UN), mineral fertiliser (MI) and slurry (SL) treatments. Short and long arrows indicate mineral and organic N applications, respectively. The bottom graph shows temperature (line) and precipitation (bars).

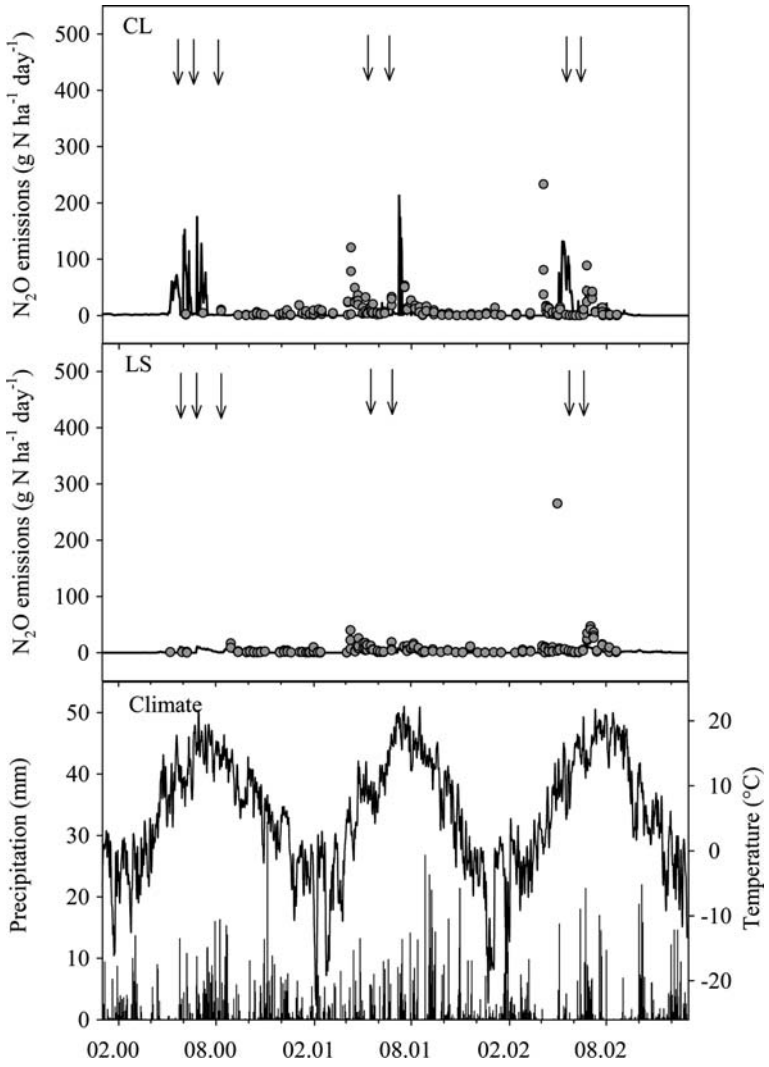

Figure 2. Simulated (lines) and measured (points) daily N_2O fluxes from January 2000 to December 2002 on the Jokioinen site for a cut grassland on clay (CL) and loamy sand (LS) soils. Arrows indicate mineral N applications. The bottom graph shows temperature (line) and precipitation (bars).

autumn N_2O emissions were considerably higher than the mid-summer emissions. Despite averaging of 30 years of simulations there remained a considerable variation in simulated N_2O emissions due to the daily variation in weather conditions.

Mean simulated N_2O emissions increased with increasing N fertilisation rates in a non-linear way such that the slope of the response curve increased at

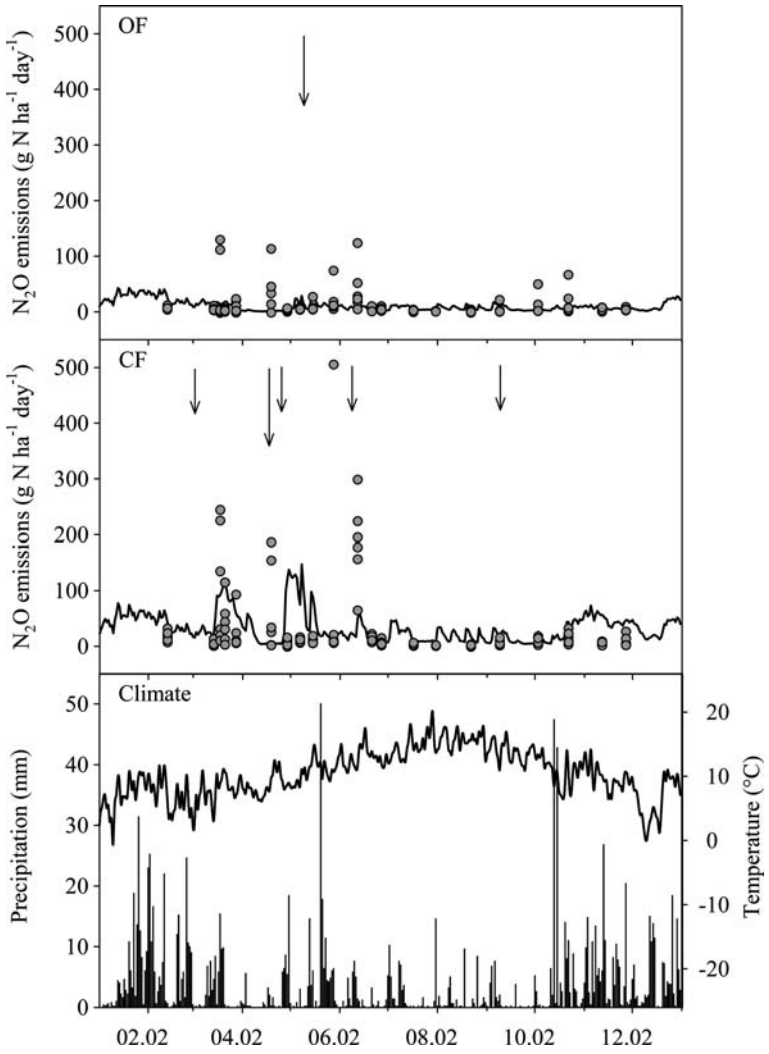

Figure 3. Simulated (lines) and measured (points) daily N_2O fluxes from January to December 2002 for grassland on conventional (CF) and organic (OF) farms in Devon. Short and long arrows indicate mineral and organic N applications, respectively. The bottom graph shows temperature (line) and precipitation (bars).

higher N rates (Figure 5). Emissions were lowest for the coarse sandy soil and highest for the sandy loam. The N_2O emissions were higher for the grazed treatment throughout the range of applied fertiliser N, but with the highest emission increase from grazing at high N fertiliser rates (Figure 5b).

The emission factors for N_2O were estimated as the slope of the N_2O emission curve at two levels of mineral fertiliser N, low (200 kg N ha^{-1} y^{-1})

Table 4. Selected statistics for comparison of observed and simulated seasonal N_2O emissions (kg N ha^{-1}).

Data source	RMSE	EF	CD	*r*-test
All experiments	1.79	0.29	1.41	0.69

RMSE: the root mean squared error; EF: the coefficient of model efficiency; CD: the coefficient of model determination; *r*-test: the sample correlation test (Smith et al. 1997).

and high (400 kg N ha^{-1} y^{-1}). Due to the non-linearity only the two simulation results surrounding these N input levels were used in the estimation. Emission factors increased with increasing N rate and soil clay content (Table 5). The emission factors were considerably higher for the grazed treatment compared with the cut treatment, when estimated from the fertiliser N input only, but the difference almost disappeared when estimated from the total N input (fertiliser plus animal excreta). The total N input was 449 ± 33 kg N ha^{-1} for the low N treatment and 683 ± 44 kg N ha^{-1} for the high N grazed treatment.

The relative importance of denitrification for N_2O emissions increased with increasing soil clay content and N input (Figure 6a, b), and the simulated denitrification constituted on average an almost constant fraction of total N input (mineral fertiliser plus animal excreta). The ratio of N_2O to total gaseous N (N_2O + N_2) emissions decreased with increasing soil clay content and increased slightly with increasing N rate (Figure 6c). Nitrification was a significant source for N_2O emissions only for the coarse sandy soil (Figure 6d), and it

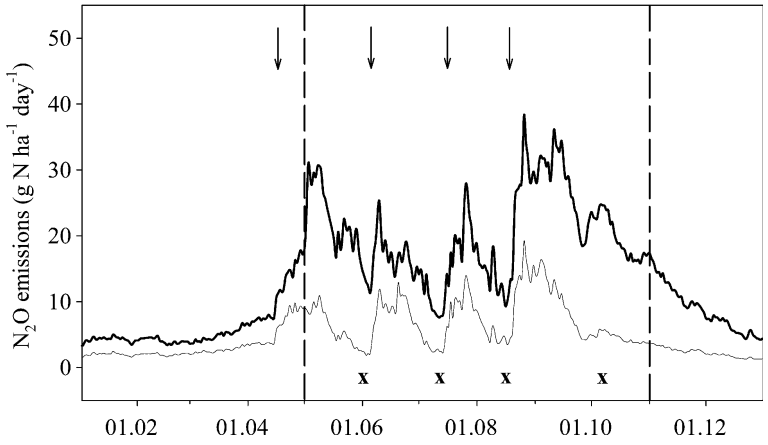

Figure 4. Simulated mean daily N_2O emissions with climate data from Foulum for a grassland using 200 kg mineral N ha^{-1}. Thin and thick solid lines indicate cut and grazed treatments, respectively. Arrows indicate dates of N application, x indicates dates of cutting, and vertical broken lines indicate the grazing period.

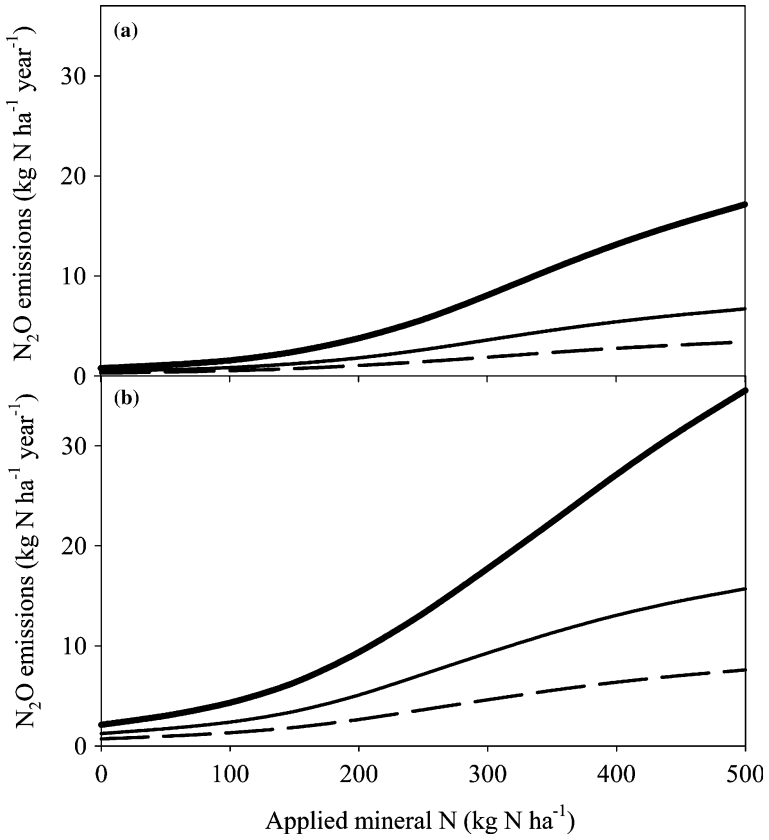

Figure 5. Simulated mean annual N_2O emissions at increasing rates of mineral N input for (a) cut and (b) grazed management. Thick solid lines indicate sandy loam, thin solid lines indicate loamy sand, and dashed lines indicate coarse sand soils.

was less important for the grazed treatment compared with the cut treatment at similar N fertiliser rates (data not shown).

The effects of changes in soil organic C content, temperature and precipitation on N_2O emissions were simulated for a cut grassland with

Table 5. Simulated mean N_2O emission factors as % of N input at low (200 kg N ha^{-1}) and high (400 kg N ha^{-1}) mineral N application rates.

Soil texture	Cut (% of applied N)		Grazed (% of applied N)		Grazed (% of total N)	
	Low	High	Low	High	Low	High
Coarse sand	0.53	0.68	1.33	1.58	0.61	0.95
Loamy sand	0.92	1.34	2.57	3.24	1.15	1.89
Sandy loam	1.91	3.24	4.73	6.73	2.18	3.96

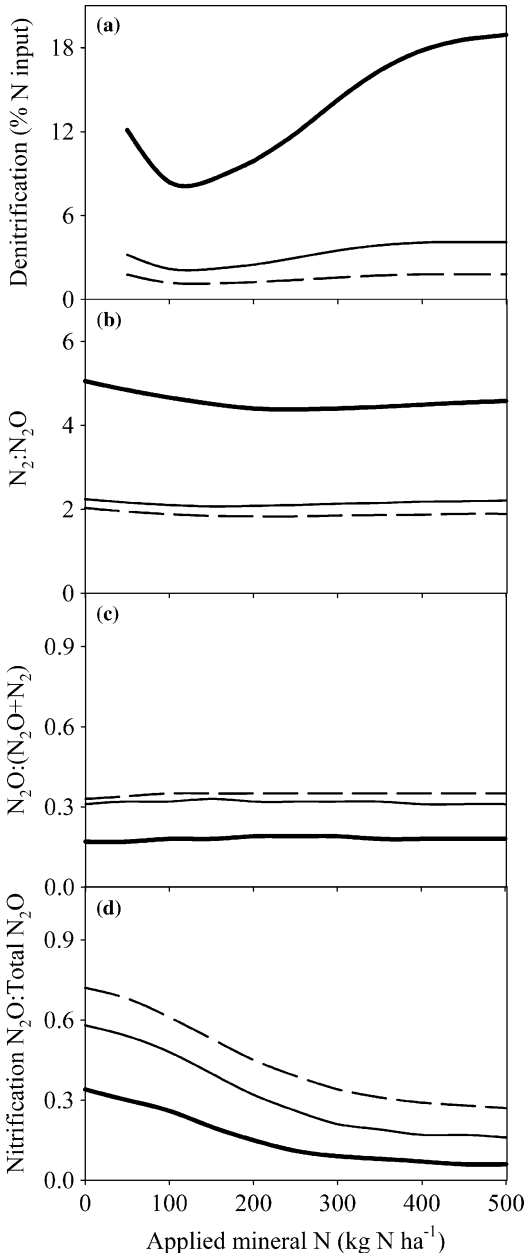

Figure 6. Simulated mean annual indicators of N_2 and N_2O emissions from cut grassland using climate data from Foulum with increasing rates of mineral N input for (a) denitrification rates, (b) N_2:N_2O and (c) N_2O:($N_2O + N_2$) ratios and (d) impact of nitrification in total N_2O emission. Thick solid lines indicate sandy loam, thin solid lines indicate loamy sand, and dashed lines indicate coarse sand soils.

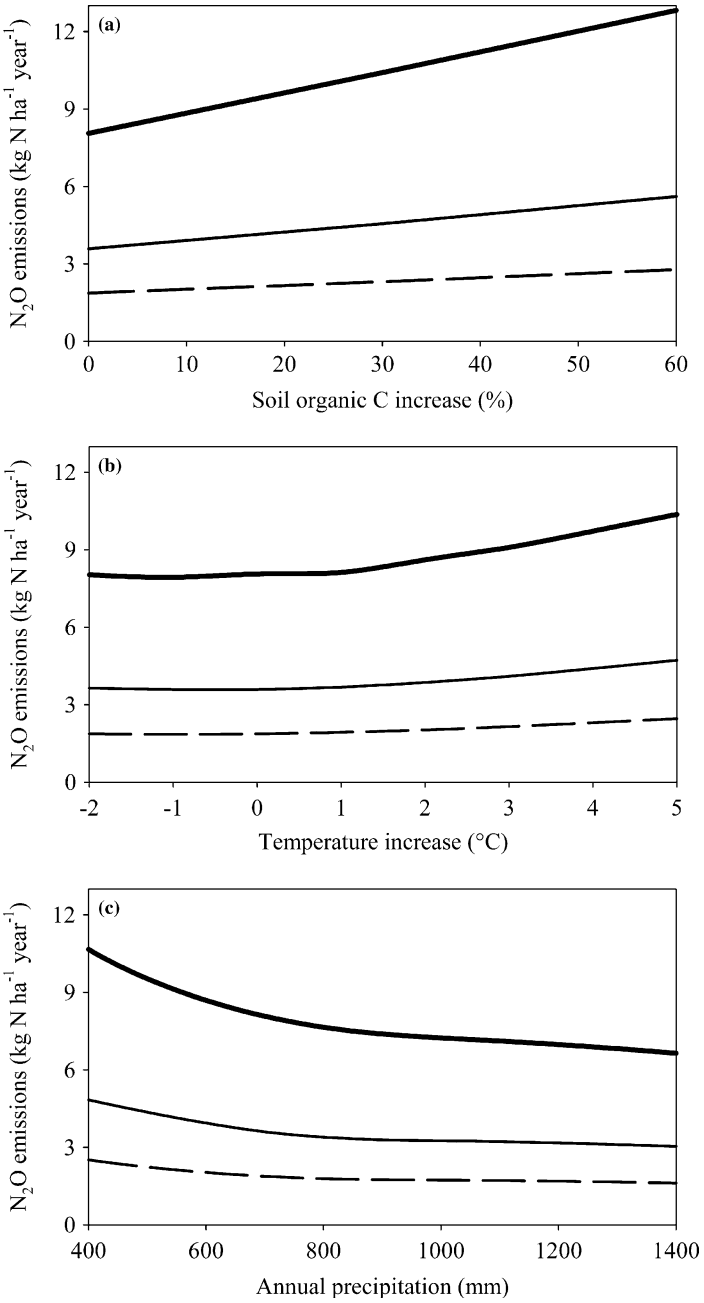

Figure 7. Simulated mean annual emissions of N_2O for cut grassland with 300 kg of mineral N ha^{-1} for variation in (a) soil organic C, (b) temperature and (c) precipitation. Thick solid lines indicate sandy loam, thin solid lines indicate loamy sand, and dashed lines indicate coarse sand soils.

300 kg N ha^{-1} in mineral fertiliser (Figure 7). Generally, the N_2O emissions were largest for the sandy loam and the response to changes in soil organic C and climate was also largest for this soil type. The N_2O emission increases linearly with soil organic C content with a mean factor of 0.82, 1.57 and 3.90 kg N_2O-N ha^{-1} % of soil organic C^{-1} for coarse sand, loamy sand and sandy loam soils, respectively (Figure 7a). The temperature response is slightly curvilinear (Figure 7b). A temperature increase of 2 °C at the mean annual temperature of 7.3 °C increased the N_2O emissions by 0.22, 0.41 and 0.93 kg N_2O-N ha^{-1} $°C^{-1}$ for coarse sand, loamy sand and sandy loam soils, respectively. There was a more complex curvilinear response of the N_2O emissions to changes in precipitation (Figure 7c). The N_2O emissions generally decreased with increasing precipitation above an annual precipitation of 400 mm. A doubling of the normal annual precipitation led to emission reductions of 0.002, 0.003 and 0.007 kg N_2O-N ha^{-1} mm^{-1} for coarse sand, loamy sand and sandy loam soils, respectively.

Discussion

The comparison between simulated and observed N_2O emissions is complicated by the large temporal and spatial variation in the measured emissions. In cases of sparse measurements this may invalidate the application of any interpolation (linear or non-linear) for estimation of seasonal fluxes. The experimental data used here from grasslands in Finland, Denmark and UK were believed to have sufficient data to allow such a linear interpolation, although there was a large spatial and temporal variation in emissions. The spatial variation was particularly large for the UK site, which in contrast to the other sites was grazed. It is possible that the heterogeneity caused by the grazing and by the urine and dung patches may have increased the spatial variation in N_2O emissions. This effect was not included in the present model study.

The FASSET model captured the increase in N_2O emissions with increasing N input in the Danish and UK experiments (Table 3). Similar increases in emissions with increasing N input have also been found in other grassland experiments (e.g. Bouwman 1996). The statistical tests also showed a good predictability of the model in general (Table 4).

The response of the model to application of mineral fertiliser (MI) versus slurry (SL) was tested for an experiment in Denmark, and the model predicted the MI treatment well, but underestimated seasonal emissions in SL experiment (Table 3). The daily emission were well simulated for the MI treatment, but underestimated for two periods for the SL treatment (Figure 1c). The July to August period for the SL treatment is based on two very high peaks. During the May to July period the simulations also miss the N_2O peaks due to slurry application. This coincides with a simulated decrease in the ratio of the mean N_2 to N_2O fluxes (N_2:N_2O) with increasing in temperature.

Rudaz et al. (1999) also found that the highest N_2:N_2O ratio occurred at low soil temperatures.

The model was unable to capture the high spring N_2O fluxes in 2001 and partly in 2002 for the Finnish site (Figure 2). Part of this can be explained by the use of a tipping-bucket model for simulating soil water transport in FASSET. This type of model will tend to overestimate water infiltration rates during periods of high rates of water input such as during thawing in spring.

The model was generally able to capture effects of nitrogen availability, soil temperature and soil water content on N_2O emissions found in many experimental studies (Castaldi and Smith 1998; Smith et al. 2003). These effects are also included in other simulation models, which have shown similar simulated seasonalities in N_2O emissions and demonstrated similar if not worse performance in simulating actual emission peaks from grasslands (e.g. Schmid et al. 2001; Xu-Ri et al. 2003; Saggar et al. 2004). These differences between simulated and measured emissions are often caused by the natural spatial variation in emissions due to heterogeneities in the spatial (horizontal and vertical) location of organic substrates and inorganic N. Much of this spatial heterogeneity is not included in the models, and probably never will be. This gives a general limitation as to how well observed emission peaks can be replicated by a simulation model. Models should therefore not only be evaluated in terms of their ability to replicate the measured time course of N_2O emissions, but also their ability to capture the influence of environmental conditions and management on emissions.

The importance of soil texture for N_2O emissions was clearly illustrated in the scenario analyses showing higher emissions with increasing soil clay content from coarse sand to sandy loam soils (Figure 5). This is in line with results from experimental data, which have shown that fine textured soils and restricted drainage favour N_2O emissions (Velthof and Oenema 1995). The model predicted a higher N_2:N_2O ratio for the fine textured soils, which is in line with experimental evidence showing higher ratios at high water-filled pore space (Weier et al. 1993). The simulated N_2O:($N_2O + N_2$) ratio were largely unaffected by N input, but was considerably lower for the fine textured soils as has also been found in experimental studies (Malone 1996). Higher N_2O:($N_2O + N_2$) ratios are characteristic of "better aerated" soils, in which N_2O can easily diffuse away, and thus is not further reduced to N_2 via denitrification (Webster and Hopkins 1996). The model also predicted that the role of nitrification for N_2O emissions is smaller for fine textured soils and that this role decreases with increasing N input, because of the increased availability of nitrate as a substrate for denitrification.

The N_2O emissions responded to both temperature and precipitation, but in different ways (Figure 7b, c). The largest non-linear responses were seen for variation in precipitation. Periods of high rainfall will increase N_2O emissions from denitrification. However, increased precipitation will also increase grass growth and possibly nitrate leaching and thus will reduce soil nitrate concentrations, which will tend to reduce N_2O emissions. At low precipitation, the

effect of increasing precipitation on denitrification probably dominated, whereas the effect of precipitation on nitrate leaching dominated at high precipitation and temperatures (Malhi et al. 1990). For a sandy loam soil Li et al. (1992) found a reduction in N_2O emissions of 0.003 kg N_2O-N ha^{-1} mm^{-1} for an increase of 20% of the mean annual precipitation and an increase in N_2O emissions of 0.31 kg N_2O-N ha^{-1} $°C^{-1}$ for an increase of 20% in mean annual temperature. Smith et al. (1998) found an increase in N_2O emissions with increasing soil temperatures of 0.97 kg N_2O-N ha^{-1} $°C^{-1}$ for temperatures in a range of 14–24 °C. These sensitivities to temperature and precipitation are similar to the sensitivities obtained with the FASSET model. The simulated response of N_2O emissions to changes in soil organic C (Figure 7a) in coarse sand and loamy sand soil textures is in line with the interval of 0.99–1.55 kg N_2O-N ha^{-1} % of soil organic C^{-1} found by Freibauer and Kaltschmitt (2003) and of 1.04 kg N_2O-N ha^{-1} % of soil organic C^{-1} found by Li et al. (1992) for loam soil. The higher simulated responses for the sandy loam reflect the importance of including clay content in the model.

The scenario analysis showed a non-linear response to fertiliser N input, in particular for the grazed treatments (Figure 5). Similar non-linearities were found at high N rates in a simulation study for a cut grass-clover pasture in Switzerland (Schmid et al. 2001). This non-linearity effect mainly occurred after many years of different N application in the Swiss study. However, in the current study the effect resulted from differences in N rates applied within the same year. The non-linear effect can most probably be ascribed to higher mineral N concentrations in the soil at high N rates, where the N uptake capacity of the plants will be exceeded. This non-linear response to increasing N fertiliser rates has also been found experimentally for grasslands in the Netherlands (Velthof and Oenema 1995; Velthof et al. 1997).

Grazing increased N_2O emissions considerably in the simulations. In the second year, this increase in the emissions from grazing occurred even before the start of grazing in spring, which could be due to higher total C and N recycling in the previous year. The higher N_2O emissions from grazing have often been attributed to the effect of dung and urine patches on N_2O production and the effect of treading and soil compaction on soil oxygen status and thus on denitrification and N_2O emissions (Velthof et al. 1997; Saggar et al. 2004). However, the effect could simply be explained by a higher total C and N input under grazing and possibly the non-linear effect of increasing N input rates on the emissions. When the N input was estimated on the basis of total N, including the amount of animal excreta, the simulated emission factors were slightly higher for the grazed versus the cut system (Table 5). The cut treatment had very sharp simulated N_2O peaks following each N application (Figure 4). However, as mentioned by Rudaz et al. (1999), it is difficult to quantify the contribution of individual effects of N input and cutting on N_2O emissions, because they occur simultaneously. The grazed treatment showed low N_2O emissions during mid-summer, probably due to dry conditions. The N_2O emissions increased at the end of season due to an increase in the importance of

denitrification and respectively sensitivity to the N_2:N_2O ratio. Rudaz et al. (1999) found higher amounts of available C at the end of growing season, which could stimulate N_2O emissions. The scenario analyses did not include the long-term effect of increased input of N in SOM for the grazed grassland, which will increase the emissions over time (Schmid et al. 2001).

The IPCC methodology for N_2O emissions inventory applies a fixed emission factor of 1.25% for N in fertiliser and manure and 2% for N excreted by grazing animals (IPCC 1997). This difference between N in fertiliser and N from grazing was smaller in the simulations with the FASSET model, when the two systems were compared at similar total N input. It may therefore be that the difference in the emission factors between the two systems is exaggerated in the IPCC methodology.

The current study along with other simulation and experimental studies have shown a non-linear response of N_2O emissions to total N input. Thus based on data in Schmid et al. (2001) increasing the application of NH_4NO_3 for cut grassland from low (200 kg N ha^{-1}) to the high (400 kg N ha^{-1}) N input level increased the N_2O emission factor for a sandy loam by 32%, where the FASSET model simulated an increase of 69% (Table 5). This effect is not included in the emission factor approach used by IPCC. The effect is that a given reduction in N input based on the IPCC methodology will yield the same reduction in the emissions irrespective of N input level, whereas in reality the highest emissions reductions will be obtained by reducing excessive N inputs. It may thus be more effective to reduce N input in high input intensive conventional farming systems than in low input organic farming systems. The study also shows the importance of including the effect of soil texture in the emission inventories. According to the current simulations, the effect of a given reduction in N input on N_2O emissions will be larger on sandy loam soil than on a coarse sandy soil. In practice this emission reduction needs to be weighed against other environmental effects of N input, such as the effect on N leaching, which will be higher on a coarse sand compared with a sandy loam soil.

Conclusions

The N_2O emission model in FASSET was capable of predicting the measured seasonal patterns and temporal variability in N_2O emissions for the most of the studied treatments from grassland experiments in Northern Europe (UK, Finland and Denmark). However, further testing against field data will be needed to improve the performance of the model for various types of N input.

The model simulations showed that within a country like Denmark, differences in soil textures are much more important for N_2O emissions than the climatic variation and in some cases even larger than variation in N input. A non-linear relation was found between N input and N_2O emissions, in particular at high N inputs and for fine textured soils. This has implications for the implementation of effective mitigation measures and for the emissions

inventory methodology, which currently relies on a constant emission factor irrespective of soil type and management.

Acknowledgements

This work was financed by the Danish Ministry of Food, Agriculture and Fisheries within the Danish Research Centre of Organic Farming. We thank Per Ambus from the Risø National Laboratory for providing additional experimental data for model calibration.

References

Addiscott T.M. and Whitmore A.P. 1991. Simulation of solute leaching in soils of different permeabilities. Soil Use Manag. 7: 94–102.

Ambus P. and Christensen S. 1995. Spatial and seasonal nitrous oxide and methane fluxes in Danish forest-, grassland-, and agroecosystems. J. Environ. Qual. 24: 993–1001.

Berntsen J., Petersen B.M., Jacobsen B.H., Olesen J.E. and Hutchings N.J. 2003. Evaluating nitrogen taxation scenarios using the dynamic whole farm simulation model FASSET. Agric. Systems 76: 817–839.

Berntsen J., Petersen B.M., Olesen J.E., Eriksen J. and Søegaard K. 2005. Simulation of residual effects and nitrate leaching after incorporation of different ley types. Eur. J. Agr. (in press).

Billore S.K., Numata M. and Minami K. 1996. Nitrous oxide emission from grassland and forest soils through nitrification. Current Sci. 70: 1010–1012.

Bouwman A.F. 1996. Direct emissions of nitrous oxide from agricultural soils. Nutr. Cycl. Agroecosyst. 46: 53–70.

Bremner J.M. 1997. Sources of nitrous oxide in soils. Nutr. Cycl. Agroecosyst. 49: 7–16.

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.

Christensen S. 1983a. Nitrous oxide emission from the soil surface: continuous measurement by gas chromatography. Soil Biol. Biochem. 15: 481–483.

Christensen S. 1983b. Nitrous oxide emission from a soil under permanent grass: seasonal and diurnal fluctuations as influenced by manuring and fertilization. Soil Biol. Biochem. 15: 531–536.

Cicerone R.J. 1987. Changes in stratospheric ozone. Science 237: 35–42.

Conrad R. 1996. Soil microorganisms as controllers of atmospheric trace gases (H_2, CH_4, OCS, N_2O and NO). Microbial. Rev. 60: 609–640.

Davidson E.A., Keller M., Erickson H.E., Verchot L.V. and Veldkamp E. 2000. Testing a conceptual model of soil emissions of nitrous and nitric oxides. BioScience 50: 667–680.

Del Grosso S.J., Parton W.J., Mosier A.R., Ojima D., Kulmala A.E. and Phongpan S. 2000. General model for N_2O and N_2 gas emissions from soils due to denitrification. Global Biogeochem. Cycles 14: 1045–1060.

Drury C.F., McKenney D.J. and Findlay W.I. 1991. Relationships between denitrification, microbial biomass and indigenous soil properties. Soil Biol. Biochem. 23: 751–755.

Freibauer A. and Kaltschmitt M. 2003. Controls and models for estimating direct nitrous oxide emissions from temperate and sub-boreal agricultural mineral soils in Europe. Biogeochemistry 63: 93–115.

Frolking S.E., Mosier A.R., Ojima D.S., Li C., Parton W.J., Potter C.S., Priesack E., Stenger R., Haberbosch C., Dorsch 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. Cycl. Agroecosyst. 52: 77–105.

Grundmann G.L. and Rolston D.E. 1987. A water function approximation to degree of anaerobiosis associated with denitrification. Soil Sci. 144: 437–441.

Hansen S., Jensen H.E., Nielsen N.E. and Svendsen H. 1990. DAISY – soil plant atmosphere system model. NPo-forskning fra Miljøstyrelsen, Nr. A10. Miljøstyrelsen, Copenhagen, Denmark.

Henault C. and Germon J.C. 2000. NEMIS, a predictive model of denitrification on the field scale. Eur. J. Soil Sci. 51: 257–270.

Houghton J.T., Ding Y., Griggs D.J., Noguer M., van der Linden P.J., Dai X., Maskell K. and Johnson C.A. 2001. Climate change 2001: The scientific basis. Contribution of working group I to the third assessment report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK.

Ingwersen J., Butterbach-Bahl K., Gasche R., Richter O. and Papen H. 1999. Barometric process separation: new method for quantifying nitrification, denitrification, and nitrous oxide sources in soils. Soil Sci. Soc. Am. J. 63: 117–128.

IPCC 1997. Greenhouse gas inventory reference manual. Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories. Volume 3. UK Meteorological Office, Bracknell, UK.

IPCC 2001. Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories. IPCC National Greenhouse Gas Inventories Programme, Technical Support Unit, Hayama, Japan.

Jarvis S.C. 1997. Emission processes and their interactions in grassland soils. In: Jarvis S.C. and Pain B.F. (eds), Gaseous Nitrogen Emissions from Grasslands. CAB International, Wallingford, UK pp. 1–17.

Johnsson H., Klemedtsson L., Nilsson A. and Svensson B.H. 1991. Simulation of field scale denitrification losses from soils under grass ley and barley. Plant Soil 138: 287–302.

Kirschbaum M.U.F. 1995. The temperature dependence of soil organic matter decomposition, and the effect of global warming on soil organic C storage. Soil Biol. Biochem. 27: 753–760.

de Klein C.A.M., Sherlock R.R., Cameron K.C. and van der Weerden T.J. 2001. Nitrous oxide emissions from agricultural soils in New Zealand – a review of current knowledge and directions for future research. J. Roy. Soc. N. Z. 31: 543–574.

Langeveld C.A., Leffelaar P.A. and Goudriaan J. 1997. Modelling nitrous oxide emissions at various scales. In: Jarvis S.C. and Pain B.F. (eds), Gaseous Nitrogen Emissions from Grasslands. CAB International, Wallingford, UK pp. 331–338.

Lee D.S., Bouwman A.F., Asman W.A.H., Dentener F.J., van der Hoek K.W. and Olivier J.G.J. 1997. Emissions of nitric oxide, nitrous oxide and ammonia from grasslands on a global scale. In: Jarvis S.C. and Pain B.F. (eds), Gaseous Nitrogen Emissions from Grasslands. CAB International, Wallingford, UK pp. 353–371.

Letey J., Jury W.A., Hadas A. and Valoras N. 1980. Gas diffusion as a factor in laboratory incubation studies on denitrification. J. Environ. Qual. 9: 223–227.

Li C., Frolking S. and Frolking T.A. 1992. 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., Aber J., Stange F., Butterbach-Bahl K. and Papen H. 2000. A process-oriented model of N_2O and NO emissions from forest soils: 1. Model development. J. Geophys. Res. 105: 4369–4384.

Maag M. and Vinther F.P. 1996. Nitrous oxide emission by nitrification and denitrification in different soil types and at different soil moisture contents and temperatures. Appl. Soil Ecol. 4: 5–14.

Malone J.P. 1996. Measuring the factors controlling the production of nitrogen and nitrous oxide in soils. PhD thesis, The Queen's University of Belfast, Ireland.

Malhi S.S., McGill W.B. and Nyborg M. 1990. Nitrate looses in soils: effect of temperature, moisture and substrate concentration. Soil Biol. Biochem. 22: 733–737.

Nõmmik H. 1956. Investigations on denitrification in soil. Acta Agric. Scand. 6: 195–228.

Olesen J.E., Askegaard M. and Rasmussen I.A. 2000. Design of an organic farming crop-rotation experiment. Acta Agric. Scand., Sect. B., Soil Plant Sci. 50: 13–21.

Olesen J.E., Petersen B.M., Berntsen J., Hansen S., Jamieson P.D. and Thomsen A.G. 2002. Comparison of methods for simulating effects of nitrogen on green area index and dry matter growth in winter wheat. Field Crops Res. 74: 131–149.

Parton W.J., Holland E.A., Del Grosso S.J., Hartman M.D., Martin R.E., Mosier A.R., Ojima D.S. and Schimel D.S. 2001. Generalized model for NO_x and N_2O emissions from soils. J. Geophys. Res. 106: 17403–17419.

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

Petersen B.M., Berntsen J., Hansen S. and Jensen L.S. 2005a. CN-SIM – a model for the turnover of soil organic matter. I. Long-term carbon and radiocarbon development. Soil Biol. Biochem. 37: 359–374.

Petersen B.M., Jensen L.S., Hansen S., Pedersen A., Henriksen T.M., Sørensen P., Trinsoutrot-Gattin I. and Berntsen J. 2005b. CN-SIM: a model for the turnover of soil organic matter. II. Short-term carbon and nitrogen development. Soil Biol. Biochem. 37: 375–393.

Petersen S.O., Regina K., Pöllinger A., Rigler E., Valli L., Yamulki S., Esala M., Fabbri C., Syväsalo E. and Vinther F.P. 2005c. Nitrous oxide emissions from organic and conventional crop rotations in five European countries. Agric. Ecosyst. Environ. (in press).

Pihlatie M., Syväsalo E., Simojoki A., Esala M. and Regina K. 2004. Contribution of nitrification and denitrification to N_2O production in peat, clay and loamy sand soils under different soil moisture conditions. Nutr. Cycl. Agroecosyst. 70: 135–141.

Potter C.S., Riley R.H. and Klooster S.A. 1997. Simulation modeling of nitrogen trace gas emissions along an age gradient of tropical forest soils. Ecol. Model. 97: 179–196.

Rudaz A.O., Wälti E., Kyburz G., Lehmann P. and Fuhrer J. 1999. Temporal variation in N_2O and N_2 fluxes from a permanent pasture in Switzerland in relation to management, soil water content and soil temperature. Agric. Ecosyst. Environ. 73: 83–91.

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

Schaap M.G. and Leij F.J. 2000. Improved prediction of unsaturated hydraulic conductivity with the Mualem–van Genuchten model. Soil Sci. Soc. Amer. J. 64: 843–851.

Schmid M., Neftel A., Riedo M. and Fuhrer J. 2001. Process-based modelling of nitrous oxide emissions from different nitrogen sources in mown grassland. Nutr. Cycl. Agroecosyst. 60: 177–187.

Smith K.A., Ball T., Conen F., Dobbie K.E., Massheder J. and Rey A. 2003. Exchange of greenhouse gases between soil and atmosphere: interactions of soil physical factors and biological processes. Eur. J. Soil Sci. 54: 779–791.

Smith K.A., Thomson P.E., Clayton H., McTaggart I.P. and Conen F. 1998. Effects of temperature, water content and nitrogen fertilization on emissions of nitrous oxide by soils. Atmos. Environ. 32: 3301–3309.

Smith P., Smith J.U., Powlson D.S., McGill W.B., Arah J.R.M., Chertov O.G., Coleman K., Franko U., Frolking S., Jenkinson D.S., Jensen L.S., Kelly R.H., Klein-Gunnewiek H., Komarov A.S., Li C., Molina J.A.E., Mueller T., Parton W.J., Thornley J.H.M. and Whitmore A.P. 1997. A comparison of the performance of nine soil organic matter models using datasets from seven long-term experiments. Geoderma 81: 153–225.

Syväsalo E., Regina K., Pihlatie M. and Esala M. 2004. Emissions of nitrous oxide from boreal agricultural clay and loamy sand soils. Nutr. Cycl. Agroecosyst. 69: 155–165.

van den Pol-van Dasselaar A., Corre W.J., Prieme A., Klemedtsson Å.K., Weslien P., Stein A., Klemedtsson L. and Oenema O. 1998. Spatial variability of methane, nitrous oxide, and carbon dioxide emissions from drained grasslands. Soi Sci. Soc. Am. J. 62: 810–817.

Velthof G.L. and Oenema O. 1995. Nitrous oxide fluxes from grassland in the Netherlands: II. Effects of soil type, nitrogen fertilizer application and grazing. Eur. J. Soil Sci. 46: 541–549.

Velthof G.L., Oenema O., Postma R. and Van Beusichem M.L. 1997. Effects of type and amount of applied nitrogen fertilizer on nitrous oxide fluxes from intensively managed grassland. Nutr. Cycl. Agroecosyst. 46: 257–267.

Venterea R.T. and Rolston D.E. 2002. Nitrogen oxide trace gas transport and transformation: 1. Evaluation of data from intact soil cores. Soil Sci. 167: 35–48.

Vinther F. 1990. Temperature and denitrification (in Danish). NPo-forskning fra Miljøstyrelsen, Nr. A3. Miljøstyrelsen, Copenhagen, Denmark.

Webster E.A. and Hopkins D.W. 1996. Contributions from different microbial process to N_2O emission from soil under different moisture regimes. Biol. Fertil. Soil. 22: 331–335.

Weier K.L., Doran J.W., Power J.F. and Walters D.T. 1993. Denitrification and the dinitrogen/nitrous oxide ratio as effected by soil water, available carbon, and nitrate. Soil Sci. Soc. Amer. J. 57: 66–72.

Wijler J. and Delwiche C.C. 1954. Investigations on the denitrification process in soil. Plant Soil 5: 155–169.

Williams D.L., Ineson P. and Coward P.A. 1999. Temporal variations in nitrous oxide fluxes from urine-affected grassland. Soil Biol. Biochem. 31: 779–788.

Wösten J.H.M., Lilly A., Nemes A. and Le Bas C. 1999. Development and use of the database of hydraulic properties of European soils. Geoderma 90: 169–185.

Xu-Ri, Wang Y., Zheng X., Ji B. and Wang M. 2003. A comparison between measured and modelled N_2O emissions from Inner Mongolian semi-arid grassland. Plant Soil 255: 513–528.

Yasukazu H., Tsuruta H. and Minami K. 2000. Effects of the depth of NO and N_2O productions in soil on their emission rates to the atmosphere: analysis by a simulation model. Nutr. Cycl. Agroecosyst. 49: 29–33.

Yoh M., Toda H., Kanda K. and Tsuruta H. 1997. Diffusion analysis of N_2O cycling in a fertilized soil. Nutr. Cycl. Agroecosyst. 49: 29–33.