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Sources of Nitrous Oxide Emissions in Intensive Grassland Managements

By

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Summary

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This paper provides a review of some recent research on nitrous oxide (N_2O) emissions conducted in intensively managed grassland-based livestock systems by The Institute of Grassland and Environmental Research (IGER). Studies have allowed improved confidence in the estimates of rates of emission from a range of sources but have also demonstrated wide-ranging temporal and spatial variability. Losses associated with the movement of NO_3^- at depth in well-drained soils, and with other un-managed components of a dairy farming landscape have been identified and quantified. The data obtained have been used to develop improved model prediction. A process-based model has been developed from the DNDC model of LI & al. 1992 which has been validated against UK information. This has produced a flexible, transparent means of predicting N_2O emissions at a range of scales ranging from field to national levels.

Introduction

Intensively managed grasslands are a dominant feature of agricultural landscapes in temperate climatic zones, and have a major role to play in many national agricultural economies. Nearly 40 % of the agricultural land in Europe is grassland which is used for livestock farming. As well as their high production potential, intensively managed grasslands systems can also act as a substantial source for many environmentally active agents: these include nitrate (NO_3^-) and phosphorous moving into waters, and ammonia (NH_3), methane and nitrous oxide (N_2O)

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emitted into the atmosphere. A key feature of much of these grasslands is the high level of N input into their management and, as a consequence, the very high rates of transfer, transformation and loss. Because of this, there is much potential for N_2O to be generated at a number of distinct phases in the grassland livestock production cycle and there is a growing awareness of the contribution that these farming systems can make to overall emissions of N_2O on a national basis.

Nitrous oxide is generated by microbial activities in soils and other environments. The International Panel on Climate Change (IPCC 1997) identifies three major categories of sources within agricultural systems for N_2O generation, namely (i) direct emissions from soils, (ii) from animal production (including emissions from housing), and (iii) emissions caused indirectly by agricultural activities. Inputs of N, of whatever origin, into agricultural management enhance the rates of microbiological processes, *i.e.* nitrification and denitrification, which generate N_2O. These processes have been described in much detail over a number of years and their overall reactions as influenced by grassland conditions are as shown in the schema below. As can be seen from these activities, grassland soils and their associated livestock either generate, or require forms of N which can be utilised from various pools by the two processes responsible for N_2O emissions.

NITRIFICATION

Supply from

mineralization of soil organic) Soil colloids

matter excreta: dung, urine,) ↑↓ $\nearrow N_2O + NO_x$ $\nearrow$

slurries, etc.,) $\rightarrow NH^+_4 \rightarrow NO^-_2 \rightarrow NO^-_3$

atmospheric deposition)

↑ ***controls*** ↑

temperature

pH

aerobicity

DENITRIFICATION

Supply from

controls

temperature

anaerobic conditions

↓ carbon (energy supply) ↓

nitrification) $\nearrow$ $\nearrow$

fertilizers) $\rightarrow NO^-_3 \rightarrow NO^-_2 \rightarrow N_2O \rightarrow$ N_2

deposition)

↓ leached

The balance of the contribution from each of the processes is not well defined and our understanding of the controls over total emissions is incomplete because of the complexity that this introduces. As can be seen by the chain of events in the above schema the processes are inter-connected with one process supplying the substrate for another. They can also take place concurrently (anaerobic sites may be in close proximity to aerobic sites in grassland soils) and can occur, not only in soil but in associated drainage waters, in stored and applied manures and off-site downstream from the farm after the transfer of NO_3^- or NH_3.

Because of all the potential sites, substrates and sources in grassland systems (see OENEMA & al. 1997), the overall emissions of N_2O per ha are generally greater than those from arable and forest soils. There have been many studies of the rates of N_2O emission, and the effects of contributing sources and their controlling factors. However, the degree of uncertainty attached to overall estimates is still large because of the high degree of spatial and temporal variability that is associated with both of the processes involved.

In the present paper, recent research conducted at IGER, North Wyke, in S.W. England, is reviewed and examined in the light of other findings and general perceptions. Opportunities to improve our understanding and to make progress in extending our knowledge base are also identified.

Sources of N_2O within a grassland context

It is usually assumed that, under mostcircumstances, the major source process for N_2O is denitrification. Much less is known about the extent of the contribution from nitrification but this is thought to be a less important source where N inputs are high. Most of the microbial activity generating N_2O occurs in the topsoil but there is also potential for this to occur at depth. This was shown in recent studies with intact soil columns (CLOUGH & al. 1999) in which the mutual presence of a suitable C supply and NO_3^- substrate stimulated denitrification activity at depths down to c. 0.8m in a well-drained soil. Within the upper horizons of grassland soils, the potential to generate N_2O is great. Their high organic matter contents are a major source of NH_4^+ through mineralization, there are substantial NH_4^+ inputs from excreta. These substrates, plus NH_4^+ and NO_3^- added in fertilizer, all contribute to a considerable opportunity for both nitrification and denitrification to occur. This being the case, background fluxes of N_2O even from un-amended grassland soils can therefore be significant. Thus YAMULKI & al. 1998 showed that over the first 100 days after application, emissions from untreated soil were only c. 33 and 39 % less than from soil where dung or urine, respectively, had been applied.

This 'background' emission is a reflection of the previous N inputs to the management system to produce situations with a low C: N ratio and therefore of the substrates (NO_3^- and NH_4^+) available for conversion. The significance of this 'background' flux has probably not been taken properly into account in previous estimates of overall contributions of agriculture to greenhouse gas emissions. Current inputs of N in excreta, as shown in Table 1, enhance the N_2O fluxes. The effects that result from the deposition of dung and urine to grassland in the SW of

England can be substantial: the excreta in this study were applied at rates which were equal to those which occur in the field.

Table 1. Emissions of N_2O from excretal patches in grassland in S W England. Values are for measurement periods (days in parentheses) after the application of excreta at rates equivalent to those occurring under grazing

Soil Type	Date of Application	Excreta	Treatment induced emission (kg N_2O-N ha^{-1})	% loss of applied N
Silty clay loam[1]	16/09	Dung	2.21	0.53
	(100)	Urine	1.15	0.85
	31/05	Dung	0.30	0.07
	(100)	Urine	0.47	0.35
	mean[3]	Dung	0.79	0.19
		Urine	0.76	0.56
Silty clay loam[2]	23/11	Dung	0.02	<0.01
	(80)	Urine	2.37	1.75
	18/04	Dung	0.68	0.14
	(95)	Urine	-0.27	-

[1] YAMULKI & al. 1998
[2] ALLEN & al. 1996
[3] Mean value for 6 application occasions through an annual grazing cycle.

Variability in responses to excretal inputs

YAMULKI & al. 1998 showed that the variation in fluxes from excreta applied at different times over a 15 month period could not be explained by the variability in environmental factors. At any one time there will be some level of competition for the major substrate (NO_3^-) from other 'removal' processes, i.e. leaching and transfer into plant or microbial biomass. These processes determine the pools of N available for denitrification, and the balance of these interactions will differ through the year. In the study of YAMULKI & al. 1998, the average annual N_2O fluxes were approximately 5x greater from urine than from dung patches. However, dung represents a significant source and on occasions on a well-drained soil, fluxes were far greater from applied dung than from urine (ALLEN & al. 1996). YAMULKI & al. 1998 calculated that the excreta deposited on grassland from grazing animals can contribute up to 22 % of the total N_2O emissions from UK grassland.

Measurements on the same poorly drained soil type as that used in the studies in Table 1, when conditions were conducive to denitrification in autumn, demonstrate the the considerable impact on fluxes of intensive management (high N input and a heavy stocking rate of sheep) (Fig. 1). The data in this figure also illustrate the large temporal variability in emissions in the two treated areas. WILLIAMS & al. 1998 also measured nitric oxide (NO) emissions and concluded, on the basis of N_2O: NO ratios, that although both denitrification and nitrification processes were contributing to N_2O production, denitrification was the dominant mechanism. Laboratory studies indicated that there was a complex denitrification–nitrification interaction with an inhibiting effect on nitrification during the first few days after deposition of urine. As the data in Fig. 1 demonstrate, there is a clear and obvious direct effect of fertilizer application on N_2O emissions as many previous studies have shown. Although specific differences which relate to fertilizer type can be demonstrated, in predictive methodologies (viz. IPCC 1997), little account is taken of these differences. OENEMA & al. 1998 recently calculated that for three dairy farming systems in the Netherlands, the average proportion of the total direct and indirect N_2O emissions which resulted from purchased fertilizer input (mean 180, range 53 –330 kg N ha^{-1}) was 22 (range 21 – 25) % of the total. The cattle slurry derived emissions, and those from grazing, accounted, on average, for only a further 2 % of the total from these farms (OENEMA & al. 1998).

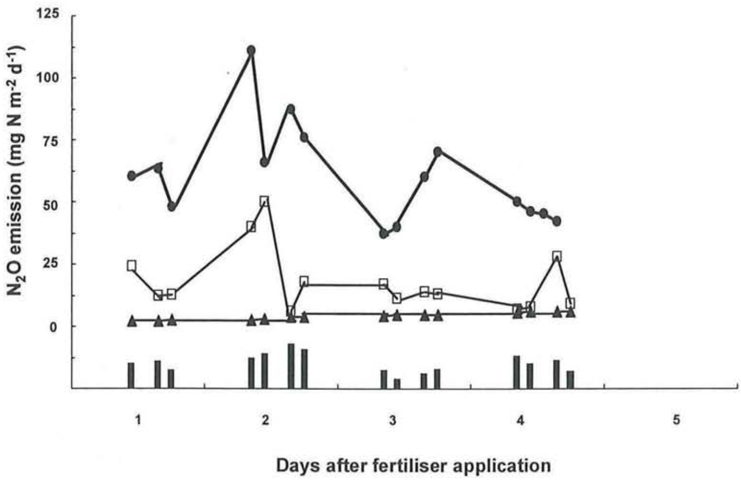

Fig. 1. Nitrous oxide emission after fertilizer application (120kg N ha^{-1}) to mown (□) or grazed (●) swards and from un-amended grassland (▲). Solid bars show SED. (from WILLIAMS & al. 1998).

Application of slurries and other manures produces conditions and substrates which encourage N_2O production. CHADWICK & al. 2000 found marked effects of manure type on fluxes (Table 2), but all produced enhanced rates of emissions. Data in Table 2 are for emissions that resulted from surface spreading; current pressures to modify manure management to reduce NH_3 emissions could be expected to have some impact through 'knock-on' effects on N_2O fluxes. However, in another study, CHADWICK & al. 1999 indicated that there was as yet no conclusive evidence on the influence of, for example, injecting slurry, and in their inventory calculation procedure they assumed no difference in fluxes between conventional broadcasting and injection techniques. CHADWICK & al. 2000 also considered emissions from other components of the livestock production system, for example from the animal houses and manure stores. Although there are only very limited data, CHADWICK & al. 1999 using the available information to produce emission factors (Efs) for their inventory, indicated that emissions from housing and manure/slurry storage comprised 13 % and 15 %, respectively, of the total from UK livestock (Fig. 2).

Table 2. N_2O emissions from slurries and manures applied to grassland on a freely drained soil in S W England. (CHADWICK & al. 2000).

	Total N application (kg N ha^{-1})	N_2O emission as the result of application (kg N_2O – N ha^{-1})	Proportion of N applied (%)
April application			
Pig slurry	97	0.43	0.97
Dairy cow slurry	125	1.21	0.44
October application			
Pig slurry	300	0.72	0.24
Dairy cow slurry	35	0.13	0.38
Farm yard manure	315	0.63	0.20

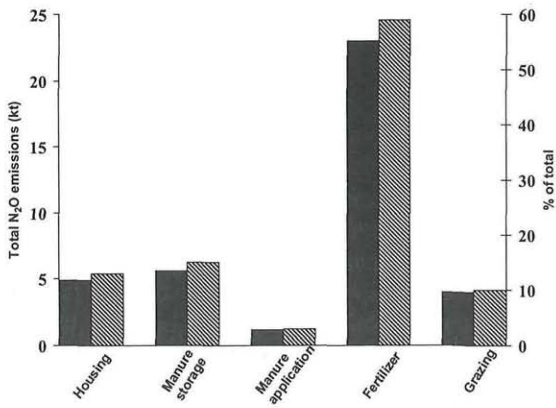

Fig. 2. UK nitrous oxide emissions from components of livestock production (from CHADWICK & al. 2000).

Temporal and spatial variability in fluxes

As already noted, there is considerable variability in the rates of emission both on a daily (Fig. 1) and a seasonal (Fig. 3) basis. Superimposed upon these temporal effects is a very strong diurnal pattern (YAMULKI unpublished)with greater rates of emissions at later times of the day. This has important implications for future development of robust EFs: most measurements have been made during only a small proportion of the day when flux rates may not be truly representative. Current, ongoing research is providing information which can be used to validate and improve existing models or to develop others which allow more confident, realistic prediction of emissions. Other effects to note in the preliminary data in Fig. 3 are (i) the low fluxes associated with the grass-clover sward, (ii) that the large peaks of fluxes usually occur after fertilizer application, and (iii) that over this period of measurement, fluxes of N_2O from the drained soil were greater than from the undrained system. The latter is contrary to expectations and may result from greater nitrification but must also involve interactions, not only between the factors directly controlling the N_2O generating process, but indirect ones (e.g. plant uptake) as well. If increased emission with drainage is a general phenomenon, this has also to be taken into account in exercises to construct inventories and budgets for N_2O.

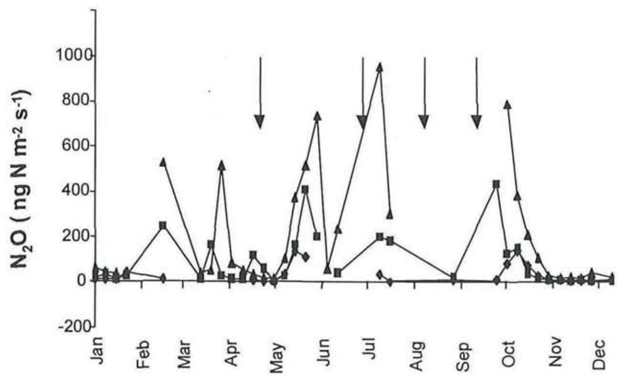

Fig. 3. Seasonal N_2O fluxes from grazed grassland: preliminary data from grass-clover (◆) or undrained (■) or drained (▲) grass (receiving c. 280 kg N ha^{-1} y^{-1}) swards. Arrows denote times of fertilizer application. (YAMULKI, unpublished information)

Spatial variability of N_2O fluxes from managed grassland is large (AMBUS & CHRISTENSEN 1994) and is controlled by differential distributions of NO^-_3, NH^+_4, water and energy (C) sources, sometimes over relatively small distances. Superimposed upon these 'background' variabilities will be the slightly larger scale effects created, for example, by the patterns of dung and urine excretion by grazing animals. Such variability makes measurement difficult and reliability/confidence in results questionable. Some recent studies have used a geostatistical approach (VELTHOF & al. 1996) to help to determine sampling strategy so that controlling factors can more precisely be determined. This study was based on similar pastures at North Wyke to those described above and tested the hypotheses that (i) the random spatial variability of N_2O fluxes was greater in grazed than in mown pastures and (ii) the pattern of N_2O fluxes was less coherent in grazed than on mown grassland. The measurements, summarised in Table 3, showed that (i) spatial variability was large over both relatively small (0.6m) and large (10 – 100m) scales; (ii) in contrast to the original hypothesis, the variability was greater on the mown than on the grazed sward (perhaps because of the high stocking rate in the latter) and (iii) there was spatial dependency in the mown but not in the grazed grassland as sugggested.

These effects have implications for the numbers of measurements required: the analysis indicated that the number of measurements required to get within 50 % of the 'true' mean was between 7 and 30, but to be within 10 % of the true value required between 375 – 1240 measurements. Clearly the spatial effects are considerable and this has presented both experimentalists and modellers with problems in being able to interpret and utilise data for predictive and up-scaling purposes.

Table 3. Summary data for N_2O flux measurements from within 18 subplots (c. 960 m^2) within main plot areas (c. 1 ha) (from VELTHOF & al. 1996).

Treatment		Mean	CVs	Maximum
Mown: day 1	range	0.4 – 6.0	73 – 159	1.3 – 20.9
	Mean	2.6	106	7.5
day 4	Range	0.4 – 2.4	77 – 273	1.5 - 14.4
	Mean	1.1	122	5.3
Grazed: day 2	Range	2.1 – 12.8	46 – 92	6.6 – 40.2
	Mean	5.1	80	15.9
day 3	Range	1.7 – 10.3	57 – 129	4.8 – 26.5
	Mean	5.5	75	15.3

Other on-farm sources of N_2O

In the current approved methodology for estimating N_2O fluxes from agricultural sources (IPCC 1997), much attention is paid to the indirect sources of this gas which arise from agricultural activities. These include volatilized NH_3 that is subsequently deposited onto land, and NO_3^- leached from managed land. Recent estimates (BROWN & al. 2001b) have shown that, for the UK, this component represented 29 % of the total agricultural sector's contribution to net emission of N_2O. The two parameters which are used to calculate this emission, i.e. the fraction of the available N in the system leached ($Frac_{Leach}$) and EF_5 (emissions from water), were the most influential of all factors considered in a sensitivity analysis and had an uncertainty of 126 %. These parameters are poorly specified and there is an urgent need to improve the emission estimate for this component. The IPCC methodology assumes (a) a constant rate of leaching (30 % of applied N) and a constant emission factor. Both of these assumptions seem to be unlikely; model predictions of leaching under grassland (NCYCLE; SCHOLEFIELD & al. 1991) suggest that the IPCC default rates for this parameter may be too high for most situations in the UK. The foundation for the emission factor is also based on few data sets. There is little doubt that denitrification can take place to at least 7m depth as NO_3^- moves from the surface soil into subsoil strata under managed grassland swards. JARVIS & HATCH 1994 demonstrated clearly that substantial potential to denitrify existed in soil and chalk substrata samples from below a range of grassland managements. This denitrification potential was equivalent to up to 200 kg N per ha to a depth of 7 m below an intensively managed sward (receiving 400 kg N ha^{-1} as fertilizer). This provides an estimate of the total potential to denitrify under 'ideal' laboratory conditions: what is not known is whether the product is N_2O or whether denitrification is complete under these conditions with N_2 as the major end product.

A recent study (CLOUGH & al. 1999) examined the fate of ^{15}N–labelled NO_3^- when this was injected into the subsoil of a freely draining soil with a source of readily available C. As the balance of ^{15}N recoveries demonstrates (Fig. 4), only small proportions of a substantial input of NO_3^- (equivalent to 120 kg N ha^{-1}) were recovered as N_2O. In total, after 38 days, 8.4 % of the initial input was detected as N_2O and only 0.4 % was emitted from the soil surface. There was, however, little doubt that denitrification had generated some very high concentrations of N_2O within the soil profile. Within one column, concentrations of c.30,000 ppm N_2O were found at a sampling point close to the injection site 13 days after NO_3^- and C were added. This clearly shows that the presence of a suitable C form together with NO_3^- in the subsoil can stimulate denitrification activity. However, the ratio N_2: (N_2 +N_2O) produced from the labelled component of the added NO_3^- decreased as the denitrification products diffused up the soil profile. Much (54 %) of the added NO_3^- was incorporated into soil organic materials.

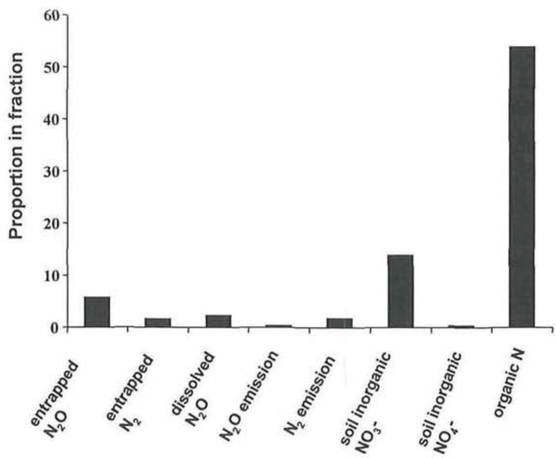

Fig. 4. Balance of ^{15}N distribution in 80 cm soil column 38 days after injection of NO_3^- and C at 70 cm depth (from CLOUGH & al. 1999).

However, although the study of CLOUGH & al. 1999 indicated that the emission of N_2O from leached NO_3^- may be limited, there may be other sources within the landscape of grassland-based livestock farming systems which may be responsible for significant fluxes of N_2O and which may not be fully taken into account. When a number of 'unmanaged' sites within a dairy farm were monitored recently (JARVIS & HEADON, unpublished), it was clear that these were responsible for significant fluxes of N_2O. Although flux rates were variable, the losses from these unmanaged areas (shown in Table 4) represented an annual loss of *c*.200 kg N_2O – N over the whole farm: this compared with an estimated (based on IPCC methodology) emission of 1116 kg N_2O – N from the managed land within the farm. More information on the extent and variability of these sources is required to allow more confident estimates of the scale of these emissions.

Table 4. Mean daily emission of N_2O–N from on-farm indirect sources (from BROWN & al. 2001a, JARVIS & HEADON, unpublished), within a dairy farming system in S W England.

Type of Site	Comments	Area on farm (ha)	kg N_2O – N d^{-1}
Gateways	Waterlogged + excreta deposition	0.24	0.40
Tracks	Excreta deposition	0.09	5.2×10^{-3}
Field feeders and water troughs	Excreta accumulation; 'poached' soil	0.15	2.4×10^{-3}
Ponds and ditches		1.52	0.03
Effluent and seepage areas	Silage and manure heaps: nutrient rich	7.1×10^{-3}	0.013
Wet areas	Nutrient transfer zones	0.13	0.02
Rivers & streams	-	1.48	4.0×10^{-3}
Unmanaged wetlands and woodlands	-	0.11	0.08
TOTAL		45.5	0.55

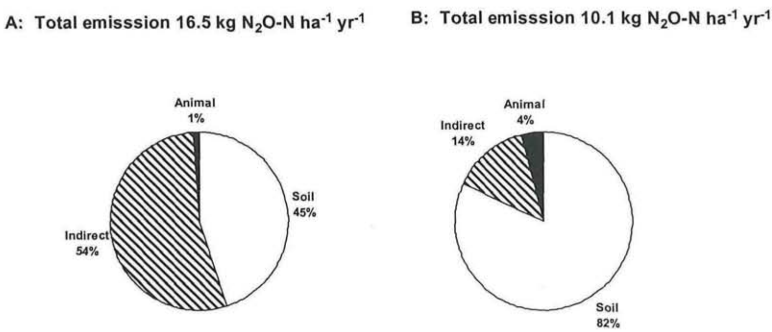

Fig. 5. Predictions of emissions of N_2O from a commercial dairy farm based on IPCC emission factors or a modified DNDC (LI & al. 1992) method (from BROWN & al. 2001a).

Modelling studies

The current policy requirements mean that there is a need to be able to predict emissions over a range of scales. BROWN & al. 20001a have recently described an attempt to estimate losses from farming systems, using an IPCC default method and one that based on a mechanistic model, the DNDC model of LI & al. 1992. The latter is a process-orientated simulation model development to assess N_2O, N_2 and CO_2 emissions from agricultural soils. We have applied both methodologies to estimate fluxes of N_2O from commercial dairy farms in S W England: an example is shown in Fig. 5. The pie chart (Fig. 5) demonstrates firstly that there is a substantial difference in the overall estimate using the two methods, and secondly the substantial contribution that the indirect component makes to the overall estimate. The DNDC methodology has also been developed to enable prediction at a national scale (BROWN & al. unpublished information). The predictions of the modified DNDC model agreed well with the 16 data sets which were used to provide validation, and produced a versatile, transparent methodology which allowed a greater degree of flexibility than the existing IPCC methodology. The method also allows scenario testing to be undertaken and has been used to develop national scale prediction for the UK (BROWN & al. unpublished).

Conclusions

This paper reviews recent IGER research into wide ranging aspects of N_2O emissions from grassland in the UK. This research is only a component of an ever-increasing literature on N_2O production, emission and their controls. The data that these studies are generating are allowing a much more confident understanding of the complex processes which are responsible for the current continuing increase in atmospheric N_2O concentrations. Clearly, the controls over N_2O production and emissions are complex and highly interactive, contributing to the extensive temporal and spatial variability in emission rates that occurs within grassland soils. However, new approaches, improved methodologies and investigations in hitherto neglected parts of the landscape are allowing a much more complete understanding of N_2O emissions. This information, coupled with improved modelling capability will allow better prediction of emissions and the development of mitigation policies and options to reduce emissions when these are required. A number of studies (OENEMA & al. 1998, JARVIS & PAIN 1994, JARVIS & al. 1996) have undertaken desk analyses of opportunities to reduce N_2O emissions from grassland based livestock systems. The new models will allow us to do this with greater confidence and accuracy over a range of scales from field, farm, regional and national points of view.

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