Soil Use and Management (1997) 13, 296–304

Emissions of N_2O and NO associated with nitrogen fertilization in intensive agriculture, and the potential for mitigation

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Abstract. Increases in the atmospheric concentrations of nitrous oxide (N_2O) contribute to global warming and to ozone depletion in the stratosphere. Nitric oxide (NO) is a cause of acid rain and tropospheric ozone. The use of N fertilizers in agriculture has direct and indirect effects on the emissions of both these gases, which are the result of microbial nitrification and denitrification in the soil, and which are controlled principally by soil water and mineral N contents, temperature and labile organic matter.

The global emission of N_2O from cultivated land is now estimated at 3.5 Tg N annually, of which 1.5 Tg has been directly attributed to synthetic N fertilizers, out of a total quantity applied in 1990 of about 77 Tg N. This amount was 150% above the 1970 figure. The total fertilizer-induced emissions of NO are somewhere in the range 0.5-5 Tg N. Mineral N fertilizers can also be indirect as well as direct sources of N_2O and NO emissions, via deposition of volatilized NH_3 on natural ecosystems and denitrification of leached nitrate in subsoils, waters and sediments.

IPCC currently assume an N_2O emission factor of $1.25 \pm 1.0\%$ of fertilizer N applied. No allowance is made for different fertilizer types, on the basis that soil management and cropping systems, and unpredictable rainfall inputs, are more important variables. However, recent results show substantial reductions in emissions from grassland by matching fertilizer type to environmental conditions, and in arable systems by using controlled release fertilizers and nitrification inhibitors. Also, better timing and placement of N, application of the minimum amount of N to achieve satisfactory yield, and optimization of soil physical conditions, particularly avoidance of excessive wetness and compaction, would be expected to reduce the average emission factor for N_2O. Some of these adjustments would also reduce NO emissions. However, increasing global fertilizer use is likely to cause an upward trend in total emissions even if these mitigating practices become widely adopted.

Keywords: Nitrous oxide, nitrogen oxides, emission, intensive agriculture, nitrogen fertilizers

INTRODUCTION

Since the start of the industrial age, significant increases have occurred in the atmospheric concentrations of several gases which are now believed to have environmental impacts at the global and/or the regional scale. Three of these gases, carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O)—the so-called greenhouse gases—contribute to global warming; N_2O also causes depletion of the stratospheric ozone layer. Nitric oxide (NO) is one of the causes of acid rain and takes part in reactions leading to the formation of ozone in the troposphere—a process which is potentially damaging to biological systems and which also adds to global warming (Prather *et al.*, 1995).

Soils, both natural and cultivated, are the major global source of N_2O, accounting for some 65% of all emissions (Prather *et al.*, 1995). Agriculture and forestry, and land-use change from natural forest to agriculture, contribute about one-third of all radiative forcing resulting from emissions of these trace gases (Cole *et al.*, 1996), including a substantial part of the N_2O and NO. This paper examines the effect of agricultural intensification over recent decades, in particular the increased use of fertilizer nitrogen, on emissions of these two gases, the likely trends in these emissions in the future, and the potential for reducing them by modifications to agronomic practices.

The emissions are the result of microbial processes in the soil: a small proportion of ammonium N is converted to N_2O and NO during nitrification, and the gas is also produced, along with N_2, during denitrification of nitrate (Granli & Bøckman, 1994). The more N that is being cycled through the system, the greater is the quantity converted into these trace gases and released to the atmosphere. This has been likened to a hole in a pipe, with an increase in flow through the pipe (the main process) resulting in an increase in the absolute quantity escaping via the the hole (the minor pathways) (Firestone & Davidson, 1989). In the conditions prevailing in wet tropical forests, there is a high rate of N turnover, and as is to be expected from the 'pipe' model, this results in a relatively high rate of N_2O emission. The IPCC has estimated the total emission of N_2O from this ecosystem at 2.2–3.7 Tg N/y, with a 'likely' value of 3 Tg N (Prather *et al.*, 1995).

The global emission of N_2O from cultivated land is now estimated to be an even larger source than the wet tropical

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forests: about 3.5 ± 1.8 Tg N_2O-N annually, of which 1.5 Tg is directly attributed to the use of synthetic N fertilizers (Prather *et al.*, 1995; Cole *et al.*, 1996). This quantity has increased as a consequence of the increased use of synthetic fertilizers in the last few decades. Fertilizers can also be indirect as well as direct sources of N_2O (and NO) emissions, via deposition of volatilized ammonia on natural ecosystems and denitrification of leached nitrate in subsoils, surface and groundwaters and sediments (Cole *et al.*, 1996). The total fertilizer-induced emissions of NO directly from agricultural land are even more uncertain than those of N_2O because of the sparse information available. The quantity of NO released appears to be 0.5–5 Tg N, which corresponds to somewhere between 5 and 50% of the total annual emissions from all soils, which are of the order of 10 Tg NO-N (Skiba *et al.*, 1997).

Global fertilizer use, and irrigation, is likely to increase significantly above current levels over the next few decades, principally in developing countries, in an effort to meet rising world food demands resulting from rising populations. This trend can be expected to result in increased emissions. There is some scope for counteracting the trend by adopting modified agronomic practices. The choice of conventional N fertilizer applied to land, the timing of N application, the crop type, soil type and tillage practice, the soil water regime, and the method of dealing with crop residues can all significantly affect N_2O emissions from soils. Furthermore, such practices as the use of controlled release fertilizers and the addition of nitrification inhibitors along with N fertilizers have been shown also to reduce fluxes of N_2O (and NO) significantly. This paper reviews the work in this field, and attempts to assess the potential for reductions in emissions by modifying current agricultural practices, within the constraint of the need to maintain, and in fact increase, world food production.

EFFECT OF FERTILIZER AMOUNT AND CHEMICAL FORM ON EMISSIONS

The effects of N fertilizer application rates, and the chemical form used, on the quantities of N_2O released from agricultural soils have been widely reviewed over the last dozen years (Galbally, 1985; Bolle *et al.*, 1986; Keller *et al.*, 1988; Eichner, 1990; Byrnes, 1990; Bouwman 1990, 1994, 1996; Granli & Bøckman, 1994; Mosier, 1994; Cole *et al.*, 1996), and the same has been done recently for NO (Skiba *et al.*, 1997). Most reported measurements cover the crop season or shorter periods (Bouwman *et al.*, 1995), particularly the few weeks after fertilizer addition when most of the N_2O is released. However, Bouwman (1994, 1996) has shown that the reported emissions from fertilized land are substantially higher when the measurements have been conducted for longer periods (Table 1), and that for the relatively small number of reported experiments of ⩾1 year's duration there was a linear relationship between fertilizer N applied and N_2O emission. The regression equation for this relationship was

$$E = 1 + 0.0125 \times F \qquad (1)$$

where E is the emission rate in kg N_2O-N/ha, and F is the fertilizer application rate in kg/ha/y. In other words, on average 1.25 kg N is emitted to the atmosphere as N_2O for every 100 kg of N fertilizer applied to agricultural land, in addition to a background flux of 1 kg, caused principally by the cycling of non-fertilizer N in cultivated land, i.e. N from crop residues, soil organic matter, atmospheric deposition and previous years' fertilization (Cole *et al.*, 1996). Emission rates vary greatly, however, because of the complex interaction of soil, crop and environmental factors; Bouwman (1994) estimated that 90% of all direct contributions of fertilizers to N_2O emissions were encompassed within the range 1.25 ± 1.0% of the applied N, which of course covers a 9-fold range, from 0.25% to 2.25%. This value is currently the basis on which OECD and IPCC are revising their recommended methodology for calculation of N_2O emissions from fertilizers, (IPCC, 1997).

Table 1. Relationship between period of measurement and estimate of N_2O emission from agricultural land (based on data in Bouwman, 1996).

Period of measurement (d)	Emission (% of fert N applied) ± std deviation	Number of studies
>30	0.6 ± 1.1	70
>100	1.1 ± 1.4	43
>200	1.6 ± 0.4	5

Recently work by one of our groups (Clayton *et al.*, 1997; Smith *et al.*, 1997a) has shown that the general trend for emissions from agricultural sites in Scotland is somewhat lower than that given by Equation 1. This may well be because the mean temperatures in Scotland at the times when N fertilizer is normally applied (and when a major part of the emissions occurs) are generally lower than those for the other sites which were the source of Bouwman's data. It is very clear that there is a large increase in emission with increasing temperature, provided that other factors are not limiting (Granli & Bøckman, 1994; Smith, 1997; Smith *et al.*, 1997b). Nonetheless, our results also show linear relationships between emission and soil mineral N content (Smith *et al.*, 1997a), which of course is heavily influenced by the amount of fertilizer applied.

This relationship clearly indicates that the more fertilizer that is applied, the greater is the likely emission. This means that the adoption of those agricultural practices which are now widely accepted as helping to minimize nitrate leaching, such as adjusting the amount of N applied to crops according to the N-supplying power of the soil and/or the amount of residual mineral N present from the previous crop, and matching the timing of application as much as possible to the crop demand, is also likely to be beneficial in reducing gaseous emissions.

One important issue is whether the chemical form of the N fertilizer has any significant impact on N_2O emissions. A review by Eichner (1990) indicated that N applied as anhydrous NH_3 had by far the highest N_2O emission coefficient (i.e. the proportion of added N evolved as N_2O) (work by Bremner *et al.*, 1981). However, more recently it has been suggested that this study may have been unrepresentative, as it involved fallow soil, so there was no competing sink for the fertilizer N, as well as leaving soyabean residues on the soil, possibly enhancing denitrification (Mosier, 1994). Mosier (1994) and Bouwman (1994) concluded that the data sets relating to N_2O emissions from agricultural soils in the literature were too limited to calculate separately the fraction of the applied fertilizer N that is emitted as N_2O for each fertilizer form. Also, in general, there has been a lack of consistency in study periods and methods, hindering further the establishment of representative flux values. In spite of this, experience in Scotland suggests that significant differences do occur, under some conditions. Clayton *et al.* (1997) have shown that in the cool wet conditions commonly occurring at the time of the first N application of the season to grass intended for conservation, in March or early April, denitrification can give rise to large emissions from nitrate-containing fertilizers, whereas emissions from urea or ammonium fertilizers are very much lower. In contrast, in warm conditions in midsummer, there are high emissions from the latter types (Fig. 1). Total emissions over a year from urea were 4 times greater than from $(NH_4)_2SO_4$, and in experiments in which urea was used in April, followed by NH_4NO_3 in June and August, significantly less N_2O was emitted over the whole season than from three successive applications of urea or NH_4NO_3 only (Table 2). The addition of slurry and NH_4NO_3 together, simulating the common practice in UK livestock farming of adding mineral N in addition to organic sources, gave significantly higher emissions than any mineral N form on its own. The differences observed between treatments are sufficiently large to provide the basis for advisory recommendations aimed at reducing emissions, by adjusting the type of fertilizer applied to intensive grassland according to prevailing soil and environmental conditions, and avoiding the practice of applying nitrate-containing fertilizers at the same time as organic manures.

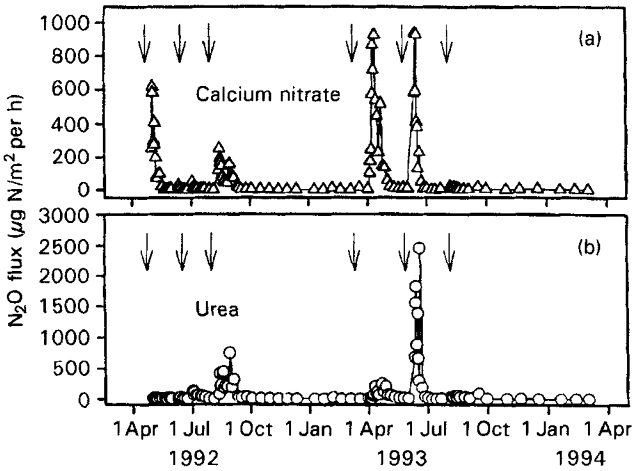

Fig. 1. Emissions of N_2O from grassland fertilized 3 times per season on dates indicated by arrows with (a) calcium nitrate and (b) urea (based on data in Clayton *et al.*, 1997).

Table 2. Cumulative N_2O emissions (over 12 months, April–March) from fertilized grassland cut for silage, after 3 applications of different types of N fertilizer (after Clayton *et al.*, 1997)

Fertilizer applications 1,2,3 (each of 120 kg N/ha)	N_2O emission over 12 months*	
	kg N_2O–N/ha	% of N applied
AN, AN, AN	4.2	1.2
U, U, U	5.2	1.4
AS, AS, AS	1.3	0.35
U, AN, AN	2.9	0.8

*from date of first fertilization. U = urea; AN = ammonium nitrate; AS = ammonium sulphate

Table 3. Emissions of N_2O and NO from an Andosol cropped with carrots in Japan, June–Sept. 1996 (H. Akiyama & H. Tsuruta, unpublished results)

Treatment	N_2O emission (mg N/m²)	NO emission (mg N/m²)	NO–N/N_2O–N
Urea + amm. sulphate	16	142	8.9
Polyolefin-coated urea ('Meister')	19	135	7.1
Urea + AM nitrification inhibitor	13	89.5	6.9

In contrast with these results for grassland, comparisons between urea and NH_4NO_3 on wheat and potato crops have shown no significant difference (K. Dobbie *et al.*, unpublished data, 1997), and it appears that alternative measures are required in arable systems, some of which are considered in the next section.

The available data on NO emissions are even more uncertain than those for N_2O. Less work has been done, and the measurements are intrinsically more demanding, requiring the use of open chambers and a chemiluminescence analyser operating in the ppb range, and air purification to remove ozone (Skiba *et al.*, 1993). Skiba *et al.* (1997) reviewed 12 studies and found that the fraction of N applied reported to be lost as NO ranged from 0.02 to 3.25% for bare soil, from 0.003 to 3.2% for grass swards, and from 0.53 to 2.5% for arable crops.

In several studies in Japan, it has been shown that the ratio of NO-N to N_2O-N was usually in the range of 3–10 (e.g. Tsuruta *et al.*, 1992), and exceeded 15 in some cases, with no obvious difference between fertilizer forms – the studies involved urea, ammonium sulphate, calcium nitrate and slow-release fertilizers. Table 3 shows results from a three month experiment on a well-aerated soil cropped with carrots, in which the overall ratio for three fertilizer forms ranged from 6.9 to 8.9.

The ratio is heavily dependent on soil water-filled pore space (WFPS), and falls rapidly as WFPS increases. In very dry conditions nitrification is inhibited, and emissions of N_2O and NO are low. At somewhat greater water contents, up to about 60% WFPS, nitrification can proceed, if the temperature is high enough, and both gases are evolved (Davidson, 1991; Granli & Bøckman, 1994). At WFPS values greater than this, and particularly above about 75%, denitrification becomes the dominant mechanism, and the rate of emission of N_2O can increase dramatically, while the ratio of NO to N_2O falls to very low levels (Skiba *et al.*, 1992). However, there is evidence of a decrease in the N_2O flux above 90% WFPS (Focht, 1978; Clayton *et al.*, 1997). In these conditions the dominant factor is likely to be the reduction in gas diffusivity in the soil, inhibiting the escape of N_2O and enhancing the probability of its reduction to N_2.

IMPACT OF FERTILIZATION AND MANAGEMENT PRACTICES

Water management

Soil water contents large enough to promote undesirable denitrification, with its associated potential for N_2O release, can come about through a variety of circumstances. Heavy rain, or too high a rate of application of irrigation water, may induce such conditions throughout the biologically active zone of the soil. On the other hand, localised spatial varia-

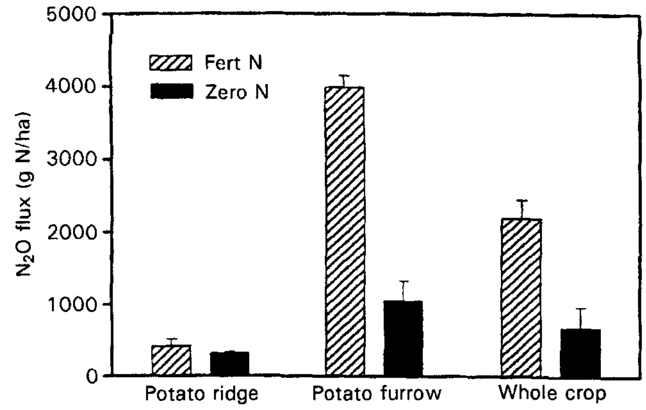

Fig. 2. Emissions of N_2O from ridges and furrows in a potato field in Scotland (after McTaggart & Smith, 1996).

tions in microrelief (e.g. ridges and furrows or raised beds), or in water infiltration rate because of compaction, can cause localised wet spots which characteristically have higher emissions. An example of this is given in Figure 2, which shows results from a recent study of a potato crop in Scotland (McTaggart & Smith, 1996), in which the N_2O emissions from the furrows were substantially greater than from the ridges. A similar result was also obtained from a study in Germany (Ruser *et al.*, 1996). The likely cause in both cases was increased soil wetness in the furrows.

In general, the available evidence on the relationship between N_2O emissions and soil wetness points to a need to apply irrigation water sufficiently slowly to prevent aeration becoming severely restricted, and to control the maximum quantity applied on any one occasion. The aim should be to ensure, where possible, that the soil water-filled pore space does not exceed about 60%. Also, tillage and trafficking should be managed so as to minimize unnecessary soil compaction or the creation of plough pans, which are likely to create excessively wet zones in the soil.

Fertilizer timing

The timing of fertilizer application can have an important role in determining the magnitude of N_2O and NO emissions from arable crops, as shown, for example, in recent work in Mexico, where wheat was grown in irrigated land under hot conditions (Ortiz-Monasterio *et al.*, 1996). The traditional practice—application of 75% of the N one month before planting and the final 25% one month after planting—was compared with a treatment in which one-third was applied at planting and the rest one month later. Emissions of both gases from the modified practice were less than half those from the traditional one (Table 4), but those of N_2O were still several times greater than from cereal crops in cool temperate conditions (McTaggart & Smith, 1996; Smith *et al.*, 1997a). The implication of the work of Ortiz-Monasterio *et al.* (1996) is that any unnecessary prolongation of the period when ammonium-based fertilizers can undergo nitrification, without any competition from plant uptake, is likely to increase emissions of both trace gases, and should be avoided wherever possible.

Table 4. Effect on emissions of timing of fertilizer application to irrigated wheat in Mexico (afer Ortiz-Monasterio *et al.*, 1996)

Amount/split of N applied (kg N/ha)	N_2O (kg N_2O–N/ha)	NO (kg NO–N/ha)	Total (kg N/ha)
Pre-planting to planting			
(187.5), 75–0–25†	2.52	5.40	7.92
(0), 0–33–66	–	–	–
Planting–1st aux. irrigation			
(0), 75–0–25	0.65	0.66	1.31
(84), 0–33–66	0.48	0.66	1.14
After 1st aux. irrigation			
(62.5), 75–0–25	0.06	0.23	0.29
(166), 0–33–66	0.97	2.05	3.02
Total			
(250), 75–0–25	3.23	6.29	9.52
(250), 0–33–66	1.45	2.71	4.16

† % of total N applied in each of three periods.

Generally, it appears that 'good agricultural practice' in the sense of promoting fertilizer use efficiency, and the adoption of practices to reduce emissions, are here as in other circumstances in harmony rather than in conflict with each other. However, there is a need for more work in tropical and subtropical conditions, to establish the size of emissions from traditional practices, and the scale of possible reductions through modified agronomic practices.

Controlled-release fertilizers

The use of controlled-release fertilizers, which are intended to supply nutrients to the soil solution and hence to the crop roots at a rate which more or less matches plant demand, has attracted considerable interest for many years, as a means of improving fertilizer use efficiency. Considerable advances have been made in the formulation of these materials, and some have entered commercial use, particularly on high-value crops where the extra cost over conventional materials is not so important.

Japan and Israel are among the countries where particular interest has been shown in these products, and in Japan studies have been carried out on the effect of their use on N_2O and NO emissions. Tsuruta *et al.* (1992) and Minami (1994) have reported work with a slow-release N fertilizer, polyolefin-coated ammonium nitrate with the trade name 'Long', which takes 100 days at an average soil temperature of 25 °C for the release of 80% of the N. Figs. 3 (a) and (b) show the very large reductions in the emissions of both gases, as a result of using 'Long' instead of ammonium sulphate as an N source. Other work with two more slow-release fertilizers, 'Meister' (polyolefin-coated urea) and CDU (a chemically synthesized product), have given conflicting results. Reductions of 23 and 38%, respectively, in NO emissions were observed when they were applied to a carrot crop, but only the 'Meister' product reduced N_2O emissions (Tsuruta, 1995). In subsequent work emissions from Meister were no different from those from unmodified urea plus ammonium sulphate (Table 3). The conclusion was drawn that the effectiveness of the polyolefin-coated products is heavily dependent on soil water content.

Recent laboratory studies by two of us (I. McTaggart and H. Tsuruta, unpublished) indicate that another controlled-release product ('Nutricote', polyolefin-coated ammonium sulphate) shows significant reductions in N_2O emissions.

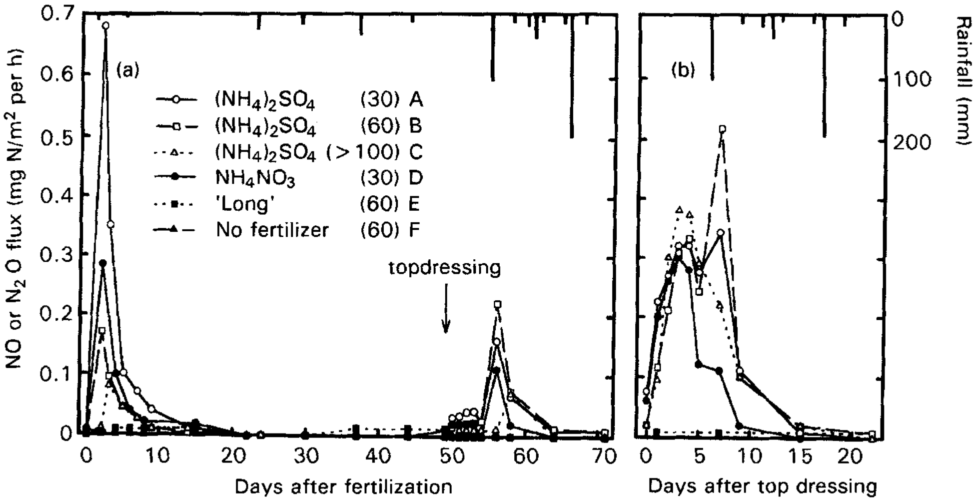

Fig. 3. Variations in (a) N_2O flux after fertilization with inorganic forms of N and with 'Long', a controlled release N fertilizer and (b) NO flux after additional top dressing. Treatments A–D: 150 kg N/ha, then topdressing of 100 kg/ha; E: 200 kg N/ha, no top dressing. Values in brackets are water table depth.

Other controlled-release fertilizers produced in several countries have as yet not been investigated, but merit similar examination in the future.

Nitrification inhibitors

Nitrification inhibitors have been studied for some years, as a means to improve fertilizer efficiency and crop yields and minimize denitrification and/or leaching losses of NO_3^- by maintaining applied fertilizer N in the soil as NH_4^+-N (Yadvinder-Singh & Beauchamp, 1988; Smith *et al.*, 1989; Bronson *et al.*, 1991). An additional consequence of keeping the N in the ammonium form is to reduce NO and N_2O emissions by nitrification, and N_2O from denitrification (Aulakh *et al.*, 1984; Bronson *et al.*, 1992).

The inhibitors most widely studied by European and N. American workers for their effects on emissions are 'N-Serve' or nitrapyrin, dicyandiamide (DCD) and encapsulated calcium carbide (ECC), which reacts with water to produce acetylene. In Japan, a range of other compounds has been explored, including AM (2-amino-4-chloro-6-methyl-pyrimidine), ST (2-sulphanilamidethiazole), DCS (N-(2,5-dichlorophenyl)-succinamic acid) and ASU (1-amidino-2-thiourea) (Asaga & Minami, 1994).

Aulakh *et al.* (1984) found that in conditions conducive to nitrification (a well-aerated clay loam soil in Saskatchewan with a moisture content of about 15–20% v/v), and in generally cool temperatures, nitrapyrin was a potent inhibitor of nitrification, and reduced N_2O emissions from urea-fertilized plots to very low levels compared with those from plots receiving urea only.

Bronson & Mosier (1991) explored the effect of ECC on N_2O and dinitrogen emissions from flooded rice, in a pot experiment. The N_2O emissions were nearly halved, from 22.8 to 12.4 μg N_2O-N per pot, and losses as N_2 nearly eliminated. Bronson *et al.* (1992) studied the effectiveness of ECC and nitrapyrin on emissions from irrigated maize, grown on a clay loam soil in semi-arid Colorado, USA, and fertilized with urea. Both the inhibitors reduced the N_2O emissions by about two-thirds in the first year of the experiment. In the second year, emissions were generally lower (attributed to the application of fewer irrigations) and were 1651, 980 and 451 g N_2O-N/ha, with urea alone, U+nitrapyrin, and U+ECC, respectively.

In studies by McTaggart *et al.* (1994, 1997) and McTaggart & Smith (1996), the emissions of N_2O from soils under grassland and arable crops were measured following application of NH_4^+, or NH_4^+-forming fertilizers with and without DCD and nitrapyrin, to determine the effectiveness of these substances as a means of reducing N_2O emissions from agricultural soils in a cool temperate climate. The measurements were made at a grassland site, on a clay loam soil, over two growing seasons. DCD was applied together with N fertilizers (ammonium sulphate, urea and ammonium nitrate) in April and August in each year. Over periods of up to 2 months following each application, emissions of N_2O were reduced by 58–78% when DCD was applied with urea, and by 41–65% when applied with ammonium sulphate. Annual emissions (April-March) were reduced by up to 58 and 56% in the two years, respectively. Applying DCD with ammonium nitrate did not reduce emissions in April, but did reduce emissions in August. Results for the second year are illustrated in Figure 4a. The inhibitor nitrapyrin was also studied, in one year only. It reduced emissions following application with urea, by 40% over the following 12 months. When inhibitor-free fertilizers were applied in June to plots that had received DCD the previous April, residual effects were still apparent: the emissions were reduced by 50–80% of the reductions observed in the period after the April fertilization.

The effectiveness of DCD as an inhibitor of N_2O emissions was also investigated in arable crops (spring and winter barley, winter wheat, spring rape and potatoes). Emissions of N_2O from arable soils varied between crops. Following spring fertilization, N_2O emissions from potato ridges were similar to those from grass, but were 1.5 times those from spring barley and >6 times the emissions from winter wheat. In treatments including DCD, emissions of N_2O from spring barley were reduced by 36% (Fig. 4b), and reduced by a similar margin in winter wheat, although due to very low overall emissions this was not significant. Emissions from potato ridges were not significantly reduced, possibly due to the surface applied DCD not immediately mixing with the injected ammonium/urea fertilizer (mineral N analy-

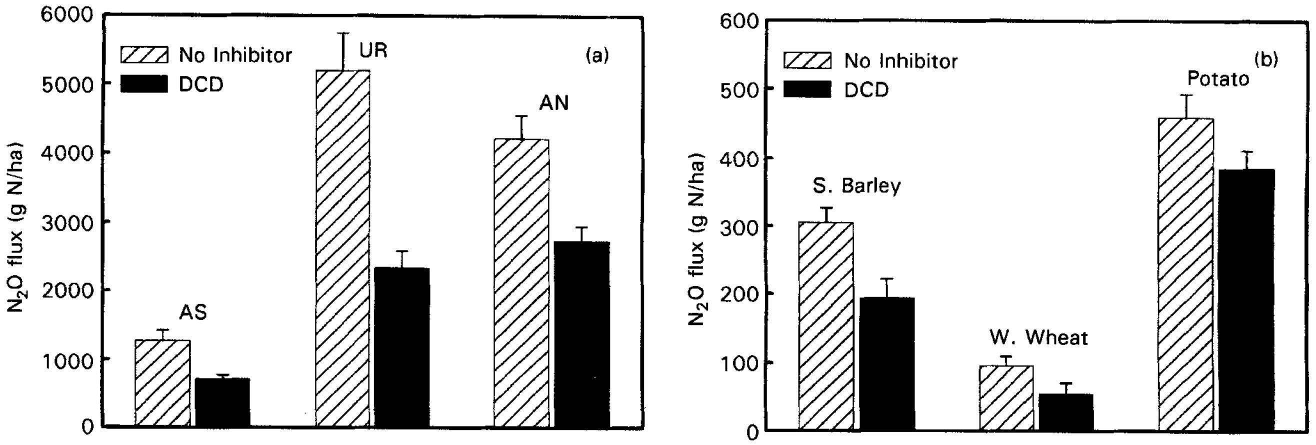

Fig. 4. (a) Total annual N_2O-N emissions from grassland fertilized with 120 kg N/ha on three occasions during the 1993 growing season, with and without the nitrification inhibitor DCD. AS: ammonium sulphate; UR: urea; AN: ammonium nitrate. (McTaggart *et al.*, 1994). (b) Total N_2O emissions following spring fertilizer applications with and without DCD to spring barley, winter wheat and potatoes. Emission periods 90, 68 and 59 days, respectively (McTaggart & Smith, 1996).

sis showed that available NH_4-N was much lower than NO_3-N, indicating rapid nitrification). In 1995 DCD reduced emissions of N_2O from winter wheat by 54%. A comparison of the effect of DCD on NO and N_2O emissions from an ammonium fertilizer by Skiba *et al.* (1993) showed that the reduction in NO flux was >90%, and more than the reduction in N_2O flux (Fig. 5).

Table 5 shows the results obtained by Minami *et al.* (1996) with DCS and ammonium sulphate applied to a carrot crop. A reduction of about a third was obtained, compared with the use of ammonium sulphate on its own. The inhibitor AM gave no reduction in a similar experiment in 1995, but brought about a 22% reduction in the following year (H. Akiyama & H. Tsuruta, unpublished data).

Delgado & Mosier (1996) compared the effectiveness of DCD in reducing emissions, when added with urea fertilizer to spring barley grown in Colorado with that of the controlled release fertilizer polyolefin-coated urea (Minami, 1994). In the initial 21 days after fertilization, DCD and POCU reduced the N_2O emissions by 82 and 71%, respectively, compared with those from urea used alone. The grain yields were 2.2, 2.5 and 2.7 t/ha for POCU, U and U +DCD, respectively, and it was concluded that although both the inhibitor and the controlled release fertilizer showed potential for use in mitigating emissions, the N release from POCU needs to be adjusted to match better the demands of the crop.

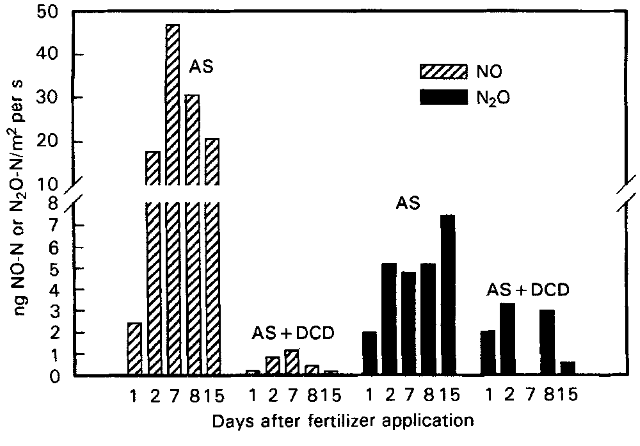

Fig. 5. Inhibitory effect of DCD on NO and N_2O emission from ammonium sulphate added to ryegrass growing on a sandy loam soil (after Skiba *et al.*, 1993).

Role of crop residues

Emissions from N-fertilized land are greater than from unfertilized areas, not only because of the direct loss of the mineral N applied, by nitrification or denitrification before any other fate is incurred, but also because of the greater quantity of crop, and associated crop residues, produced as a result of the N addition. If the residues are returned to the land, the N contained in them is mineralized and subject to the same transformations, but perhaps more importantly the decomposing organic matter has the potential to increase soil respiratory activity, and increase the chances of additional anaerobic zones developing, within which denitrification can occur (Tiedje *et al.*, 1984). In Scotland when crop residues have been ploughed into the soil in the autumn, large emissions of N_2O have been observed on some occasions. Figure 6 shows the effect of incorporating either lettuce roots, or the roots plus unharvested tops (the normal practice) into a sandy loam soil in November, when the temperature was about 10 °C. In other experiments with potato crops, nearly two-thirds of the total emissions over the season occurred in the post-harvest period, and at this time the emission, expressed as a function of the mineral N in the soil, was an order of magnitude bigger than during the growing season. This effect was attributed to the presence of labile organic matter from the crop residues, increasing soil respiration

Table 5. Amounts of N_2O emitted over 116 days from lysimeters filled with an Andosol, cropped with carrots and fertilized in the presence and absence of DCS nitrification inhibitor (after Minami *et al.*, 1996)

Treatment (kg N/ha)	N_2O–N g/ha	N_2O–N % of N applied
Control (zero N)	77	
Split N (150+50)	597	0.26
Basal N (200)	338	0.13
Basal N (200)+DCS	252	0.09

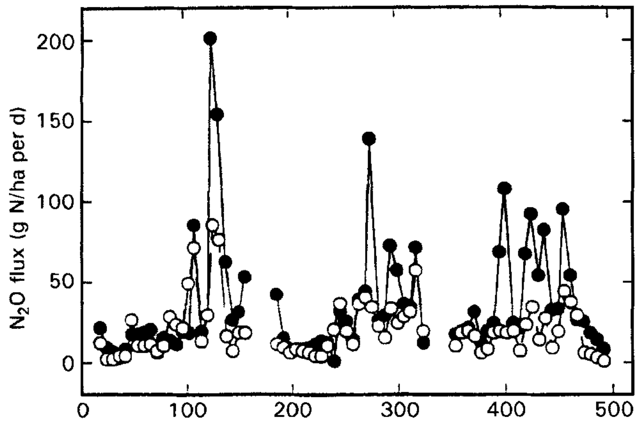

Fig. 6. N_2O emissions from a sandy soil, Fife, Scotland, Nov.–Dec. 1994, after ploughing in lettuce crop residues, and from a control plot with root residues only. ●: crop residues; O: control.

and inducing or enlarging anaerobic microsites where denitrification could take place (Smith *et al.*, 1997a).

Such releases of N_2O after harvest are a major reason why experiments of greater duration appear to produce higher total emissions from agricultural soils (Bouwman, 1996). As far as we are aware no work has been done to manipulate the management of crop residues to minimize the emissions, but in view of the magnitude and variability of the fluxes observed, there may well be considerable potential for reducing emissions by such procedures.

FUTURE TRENDS IN FERTILIZER USE AND EMISSIONS

According to Matson *et al.* (1996), the worldwide consumption of synthetic N fertilizers in 1990 was about 80 Tg, and increased to this level from only 32 Tg N in 1970; Cole *et al.* (1996) estimated a similar total of 77 Tg N (Table 6). There has been little change in total production and use during the early 1990s (O. Bøckman, pers. comm.), but there are predictions of very substantial increases over the next 30–40 years. Vitousek & Matson (1993) cited USEPA projections pointing to a doubling of N fertilizer use by 2025 in developing countries, where the increases in population will be concentrated, and where half of world use is already concentrated. If the projected extra fertilizer used over the next few decades were to be distributed over additional arable land, at broadly the same rate of application, then the most likely outcome would be a *pro rata* increase in N_2O (and NO) emissions. However, the position may be less hopeful, in that in the most populous countries the potential for additional land is limited, and a more likely scenario is the intensification of existing land use, with higher fertilizer applications, more double cropping, and more irrigation. On the basis of existing knowledge, this trend could well give rise to more than *pro rata* increases in emissions.

On the other hand, the available evidence suggests that the adoption of the best available agronomic practices that have already been worked out to optimize the agricultural efficiency of N fertilizer use and to minimize leaching of nitrate, would also have the added benefit of reducing the proportion of applied N that is emitted to the atmosphere as N_2O and NO. This would go a considerable way towards offsetting the likely increases in emissions due to greater N use. Cole *et al.* (1996) have attempted to quantify the scale of these reductions, which are summarized in Table 7. The total estimated reduction of 0.68 Tg N_2O-N/y represents about a fifth of the present total direct and indirect emissions from fertilized land. One of the more problematic areas covered in Table 7 is that of tightening N flow cycles. Cole *et al.* (1996) point out that this would reduce fertilizer N use and thus reduce the N_2O coming directly from this source. However, they also point out that part of the N mineralized from crop residues returned to the soil will be converted to N_2O, and the data shown in Figure 6 and reported by Smith *et al.* (1997a) strongly reinforce this argument. There is a need to develop residue management practices that ensure the balance is a favourable one.

Table 6. Estimated annual application to agricultural land of fertilizer and manure N (after Cole *et al.*, 1996)

	Fertilizer N	
Region	Amount consumed (Tg)	% of total
Africa	2.1	2.7
N&C America	13.1	16.9
S America	1.7	2.2
Asia	37.3	48.2
Europe	13.6	17.6
Oceania	0.9	1.2
Former SU	8.7	11.2
Total	77.4	100

Table 7. Practices likely to improve efficiency of use of synthetic fertilizer and manure N in agriculture, and expected reduction in N_2O emissions, assuming global application of the practices (after Cole *et al.*, 1996)

Agricultural practice	Estimated decrease in N_2O emissions (Tg N/y)
Match N supply with crop demand (soil/plant testing, minimum fallow periods, split applications, etc)	0.24
Tighten N flow cycles (manure reuse in plant producion, plant residue N maintained on production site)	0.14
Use advanced fertilization techniques (controlled release ferts., nitrification inhibitors, placement in soil and foliar application, match fert. type to seasonal precipitation)	0.15
Optimize tillage, irrigation and drainage	0.15
Total	0.68

So where do these conflicting trends of greater wordwide N use but improved management practices take us, in terms of likely overall emissions in the future? The conclusion must be, on the basis of present evidence, that even if all the practices listed in Table 7 were to be universally adopted, including a satisfactory solution to the residue problem, this would be insufficient to offset the increases to be expected from increased worldwide N fertilizer production and use. If, as seems possible, global warming leads to some positive feedback in terms of N_2O emission from soils (Smith, 1997), then the increases will be larger still.

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