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Nitrous oxide emissions from agricultural soils

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Abstract

This paper addresses three topics related to N_2O emissions from agricultural soils. First, an assessment of the current knowledge of N_2O emissions from agricultural soils and the role of agricultural systems in the global N_2O are discussed. Secondly, a critique on the methodology presented in the OECD/OCDE (1991) program on national inventories of N_2O is presented. Finally, technical options for controlling N_2O emissions from agricultural fields are discussed.

The amount of N_2O derived from nitrogen applied to agricultural soils from atmospheric deposition, mineral N fertilizer, animal wastes or biologically fixed N, is not accurately known. It is estimated that the world-wide N_2O emitted *directly* from agricultural fields as a result of the deposition of all the above nitrogen sources is 2–3 Tg N annually. This amounts to 20–30% of the total N_2O emitted annually from the earth's surface. An unknown, but probably significant, amount of N_2O is generated indirectly in on and off farm activities associated with food production and consumption.

Management options to limit *direct* N_2O emissions from N-fertilized soils should emphasize improving N-use efficiency. Such management options include managing irrigation frequency, timing and quantity; applying N only to meet crop demand through multiple applications during the growing season or by using controlled release fertilizers; applying sufficient N only to meet crop needs; or using nitrification inhibitors. Most of these options have not been field tested. Agricultural management practices may not appreciably affect indirect N_2O emissions.

Introduction

About 70% of the N_2O emitted from the biosphere into the atmosphere is derived from soil (Bouwman 1990; Houghton *et al.*, 1992). It seems reasonable then, to assume that human induced changes in N cycling in soil systems have influenced the increases in atmospheric N_2O during the past century and will help dictate future changes in atmospheric N_2O. What external factors perturb "normal" soil N cycling and thus increase N_2O emissions? Land use conversion has been a primary factor in the past (Houghton and Skole, 1990). Conversion of forests and grasslands to croplands accelerated C and N cycling and increased N_2O emissions from the soil. Globally, land use conversion is important now only in tropical areas. Most of the conversion of forests and grasslands in the northern hemisphere occurred 50 to 200 years ago (Hammond, 1990). Global climate change may impart changes in soil temperature and moisture which will directly influence N cycling. A direct affect, that can be quantified, is the increase in N input into soil systems. This increase in N input is derived from atmospheric deposition, which ranges from about 0.5 g N m^{-2} y^{-1} in the central U.S. to 6 g N m^{-2} y^{-1} in western Europe (Andreae and Schimel, 1989), N fertilization with mineral N sources, animal manures and biological N fixation. Nitrogen fertilizer use and biological N-fixation are projected to continue to increase during the next 100 years to meet food demands (Hammond, 1990).

To achieve the objective of determining N_2O emission budgets for various parts of the earth we must predict how much N_2O is produced from each unit of fixed N (chemically or biologically) that is added to the soil. To make this prediction we first must understand how and where N_2O is produced in the biosphere, what sinks exist for the gas, and how the gas moves from where it is produced into the atmosphere. Research during the past several decades provides an understanding of how N_2O is produced, factors that control it's production, source/sink relationships, and gas movement processes. However, even with this large amount of

knowledge, we are not yet able to reliably predict the fate of a unit of N that is applied or deposited on a specific agricultural field. Studies of emissions of N_2O from presumably "similar" agricultural systems show highly variable results in both time and space. It is the complex interactions of the physical and biological processes involved that must be understood before appropriate predictive capability can be developed.

This paper is not a comprehensive review of the N_2O literature, since several reviews have been prepared recently (those by Bouwman, 1990; Duxbury *et al.*, 1993; and Batjes and Bridges (1992) are examples). This paper addresses the issues of (I) current knowledge of N_2O emissions from agricultural soils; (II) a critique on methodology of the OECD/OCDE (1991) program on national inventories; and (III) Technical options for emission control.

Current knowledge of N_2O emissions from agricultural soils

Knowns and unknowns about N_2O flux in agricultural soils

It is surprising that during the last few years, with the renewed interest on climate change and the role of radiatively active trace gases, little new information concerning emissions of N_2O from agricultural fields has been published. Many recent review papers and inventory assessments have relied on published gas flux measurements from studies primarily conducted during the late 1970's and early 1980's. The number of flux measurements and the variety of soil conditions examined are limited. Therefore, the data from which these reviews and inventories have been drawn are also limited. Because of the lack of data, inappropriate conclusions may have been drawn. To assess the current knowledge of N_2O emissions from agricultural soils let us first briefly review some things we know and some things we don't know about this topic.

Knowns about N_2O flux in agricultural soils

As noted in the OECD/OCDE (1991) report, we know that N_2O is produced primarily from the microbial processes of nitrification and denitrification in the soil. In well aerated conditions, N_2O emissions from nitrification of ammonium based fertilizers can be substantial (Bremner and Blackmer, 1978; Duxbury and McConnaughey, 1986). Other work suggests that N_2O is a by product of nitrification (Yoshida and Alexander, 1970) and may occur by denitrification of nitrite by nitrifying organisms under oxygen stress (Poth and Focht, 1985). Recent evidence indicates that in well aerated, porous soils, little N_2O may evolve but much larger amounts of NO may be emitted during nitrification (Williams *et al.*, 1993). In wet soils where aeration is restricted, denitrification is generally the source of N_2O (Smith, 1990). Under these conditions both the rate of denitrification and the $N_2O/(N_2 + N_2O)$ ratio must be known to evaluate N_2O emissions through denitrification. According to Smith (1990), soil structure and water content which affect the balance between diffusive escape of N_2O and its further reduction to N_2 are important factors in determining the proportions of the two gases.

Research has also shown that a number of individual factors are controllers of nitrification and denitrification. Such factors include as soil water content, which regulates oxygen supply; temperature, most organisms have a temperature range over which reaction rates are optimal; nitrate or ammonium concentration, substrates may individually regulate reaction rates and in the case of denitrification regulate the NO_2/N_2 ratio; available organic carbon, denitrifiers require usable organic carbon and respiration of organic carbon may also regulate oxygen supply; and pH, is a controller of both nitrification and denitrification rates and the N_2O/N_2-ratio in denitrification.

Increases in the amount of N added to the soil generally increases N_2O emissions (Bouwman, 1990). The temporal pattern of N_2O emissions following fertilization is generally that of a large efflux of N_2O occurring for a short time (about six weeks). After this time, emission rates are reduced to fluctuate around a low base-line level independent of the amount of fertilizer applied (Mosier *et al.*, 1983). Some studies indicate that N_2O emission rates are higher for ammonium-based fertilizers than for nitrate (Eichner, 1990). For example, Bremner *et al.* (1981) found a much higher proportion of N_2O released from anhydrous ammonia than from urea or ammonium sulfate. Bouwman's (1990) review, however, suggested no particular trend in N_2O emissions related to fertilizer type. Byrnes *et al.* (1990) suggest that N_2O emissions from the nitrification of fertilizers may be more closely related to soil properties than to the N source that is supplied. Mineral N applications and organic matter amendments generally increase total denitrification and N_2O production.

As discussed in more detail by Mosier (1989), N_2O emissions from the soil in a field can vary by orders of magnitude both spatially and temporally. These heterogeneities in both space and time in measured as fluxes and in the microbial activity which produces the gases make predictions highly uncertain.

Unknowns about N_2O flux

Although the individual factors that regulate N_2O production are known, we cannot predict how these factors interact under field conditions to produce measured fluxes (OECD/OCDE, 1991). Both nitrification and denitrification and the regulators of N_2O/N_2 ratios from denitrification have their own set of optimum conditions. As a result, one process may be the primary N_2O producer in one set of field conditions, but as soil conditions change, another process may predominate. The complexity of the interactive factors important to the different processes obviously makes a simple description of N_2O production difficult (Mosier *et al.*, 1983). Complex models, such as that described by Li *et al.* (1992) may be the only way that N_2O fluxes may be predicted. Simpler, mechanistic models such as that described by Parton *et al.* (1988) may, however, play a role in simplifying estimation of N_2O emission. To accurately inventory N_2O emissions from agricultural soils we must be able to predict N_2O emissions based on N application, soil, crop, management and climatic conditions.

It is also likely that N_2O production resulting from fertilizer and increased use of biological nitrogen fixation is underestimated because the effect of a nitrogen input is usual only partially traced through the environment. Figure 1, taken from Duxbury *et al.* (1993) illustrates some of the flows of N following application of 100 kg ha^{-1} of fertilizer N to a field on a typical dairy farm in the USA. Primary and secondary flows of N are shown by dashed and solid lines, respectively. In this example, 50 of the 100 kg are removed in the harvested crop and 50 are lost by a combination of leaching (FAO, 1990), surface run-off (Bouldin *et al.*, 1984) and volatilization (Byrnes *et al.*, 1990, primarily denitrification). If N_2O comprises 10% of the volatilized N, 2 kg N_2O-N would be generated in the primary cycle. Assessments of fertilizer effects on N_2O emissions usually stop at this point even though only 20 of the 100 kg N added have been returned to the atmosphere and it can be reasonably assumed at most would be returned within 10 years.

Secondary flows, shown by the solid lines (Fig. 1), include feeding 50 kg of harvested N to animals, which generate 45 kg of manure N. The manure is returned to cropland to fertilize a second crop, however about half of this N is volatilized as NH_3 prior to or during manure application. Volatilized NH_3 is aerially dispersed and subsequently returned to and cycled through both natural ecosystems and cropland. Ammonia volatilization from agricultural systems is globally important (Isermann, 1992) but its impact on N_2O emissions has not been explicitly addressed. To provide some perspective, it should be noted that the quantities of fertilizer N used and animal manure N generated by USA agriculture are equal (Bouldin *et al.*, 1984). On a global basis, about 30 of the 80 Tg fertilizer N used each year are volatilized as NH_3.

Similarly, the amount of N_2O arising from leached nitrate, which may average 20–25% of applied N (Meisinger and Randall, 1991), is not known. Much of the nitrate may be denitrified in riparian zones or cycled through wetland or aquatic vegetation. A complete accounting of fertilizer N, biologically fixed N, and N mineralized from soil organic matter is difficult to achieve, but needed if we are to accurately assess the impact of increased use of N in agricultural systems on terrestrial N_2O emissions (Duxbury *et al.*, 1993).

Agricultural systems in the global N_2O budget

The present global average atmospheric concentration of N_2O is about 310 ppbv and it is increasing at the rate of 0.6–0.9 ppbv y^{-1} (Prinn *et al.*, 1990; Watson *et al.*, 1990). The concentration of N_2O is about 0.75 ppbv higher in the Northern Hemisphere than in the Southern Hemisphere (Prinn *et al.*, 1990), indicating greater source strength in the former. It is generally agreed that soils are the major source of N_2O but global N_2O budgeting exercises (Table 1) suggest that the strength of known sources is underestimated or that unidentified sources exist (Duxbury *et al.*, 1993; Robertson, 1993).

Analysis of the latitudinal distribution of atmospheric N_2O suggests that emissions of N_2O between 90–30N, 30N-equator, equator-30S, and 30–90S are 22–24, 32–39, 20–29 and 11–15%, respectively of the global total, and that there is a large tropical source (Prinn *et al.*, 1990). This result conflicts somewhat with the projection of Bouwman (1992) that these same latitudes contribute 32, 31, 29 and 9% respectively, to global N_2O production, reinforcing the conclusion that our knowledge of N_2O sources is incomplete.

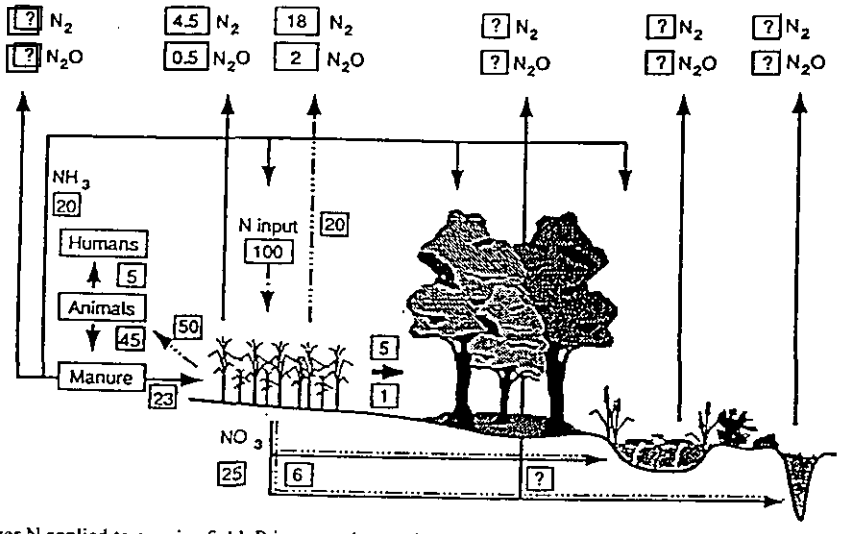

Fig. 1. Fate of fertilizer N applied to a maize field. Primary and secondary flows are shown by dashed and solid lines, respectively (Duxbury *et al.*, 1993)

Table 1. Estimated sources and sinks of nitrous oxide (Tg N per year) (Houghton *et al.*, 1992)

Sources	
Natural	
* Oceans	1.4–2.6
* Tropical Soils	
* Wet forests	2.2–3.7
* Dry savannas	0.5–2.0
* Temperate Soils	
* Forests	0.05–2.0
* Grasslands	?
Anthropogenic	
* Cultivated Soils	0.03–2.0
* Biomass Burning	0.2–1.0
* Stationary Combustion	0.1–0.3
* Mobile Sources	0.2–0.6
* Adipic Acid Production	0.4–0.6
* Nitric Acid Production	0.1–0.3
Sinks	
Removal by soils	?
Photolysis in the Stratosphere	7–13
Atmospheric Increase	3–4.5

The only known significant sink for N_2O is photolysis in the stratosphere (Watson *et al.*, 1990). Anaerobic soils have large potentials for reducing N_2O to N_2, and in fact, the major product of denitrification in soils is N_2 rather than N_2O. However, slow rates of dissolution of atmospheric N_2O and its slow transport in wet and/or flooded soils prevents this process from being a significant regulator of atmospheric N_2O (Duxbury *et al.*, 1986; Ryden, 1981).

Total global N_2O-N emissions to the atmosphere from 1978 to 1988 averaged 13.0 ± 1.5 Tg N y^{-1} according to the calculations of Prinn *et al* (1990). This compares to a range of annual estimated production of 5.2 to 16.1 Tg according to the 1992 IPCC estimate (Table 1) (Houghton, 1992). The IPCC budget seems to be incomplete as it does not include any emission value for grasslands, although the global area of this biome is almost as great as that of forest land (3.1×10^9 ha vs 4.0×10^9 ha). Using this area and data from Parton *et al.* (1988) significant emissions of N_2O roughly 1 Tg, may be ascribed to grasslands. Additionally, no contribution was included for biological N fixation in agricultural systems. By analogy to experience in the tropical environment (Matson an Vitousek, 1987), increased N cycling should lead to higher N_2O emissions in agricultural systems compared to the natural ecosystems they replaced.

A summary of the role of global agriculture in estimated total and anthropogenic emissions of greenhouse gases indicates that agriculture contributes about 70% of the anthropogenic emissions of N_2O. Recent estimates, based on the amount of N fertilizer used and published estimates of N_2O flux, ascribe an average of 1.1 kg of N_2O-N emission per 100 kg of N applied in fertilizer, about 7% of total N_2O production due to direct emission from agricultural fields each

year (CAST, 1992). Agriculture is a relatively minor contributor to anthropogenic emissions of CO_2 but it is a major contributor to anthropogenic emissions of both CH_4 and N_2O. However, emissions of CO_2 dominate anthropogenic radiative forcing of climate and this is predicted to continue into the foreseeable future (CAST, 1992).

Critique on methodology IPCC/OECD program on national inventories

The overview from Section D., "Nitrous Oxide Emissions from Fertilizer Use and Nutrient Runoff" of the OECD/OCDE (1991) report presents a concise description of the state of understanding of N_2O emissions from agricultural systems. There are a few points, however, that merit further discussion. First the statement that N_2O emissions directly from fertilizers are relatively small should be discussed. A critical look at the reviews of Eichner (1990), Bouwman (1990) and CAST (1992) indicates that a conservative estimate of direct emission of N_2O from mineral fertilizer over a full year is in the range of 1% of the N applied, currently about 1 Tg, or about 10% of current global emissions. Is this trivial? This estimate includes neither organic N fertilizer from human and farm animal excreta nor N fixed by biological N fixation. Limited data suggest that N_2O emissions from these N sources are generally greater than from mineral N application (Bouwman, 1990). Assuming that N emissions from all sources are equal, the *direct* emissions from all three N sources could total 3 Tg annually, or 20–30% of current global emissions.

A second point that needs to be addressed, but for which unfortunately there is no direct quantification, is the overall fate of N applied in agricultural systems. Generally, agricultural soils are slowly losing total N content (CAST, 1992). With this in mind it is obvious that, most of the N applied through mineral and organic N fertilization and N fixation (about 250 Tg yr^{-1} total) is returned to the biosphere within a few years. The soil retains little of the N added, crops are quickly consumed by animals and humans, and storage time of N in animals used for human consumption is short. It then follows that the N used in agriculture is returned to the atmosphere through denitrification over a period of a few years. What fraction of this N is returned as N_2O? We have no idea! Obviously only a small fraction dramatically influences the global N_2O budget.

Methodology for calculating N_2O emission from N fertilizers

A. The first methodology of OECD/OCDE (1991) is based on the amount of each type of fertilizer N consumed and an emission coefficient for the fraction of applied N that is released as N_2O-N for each fertilizer type. Emissions of N_2O-N are estimated for each fertilizer type, summed over all types (equation 1).

$$N_2O \text{ Emissions (tonnes } N_2O\text{-N)} = \sum_f (F_f \times E_f) \quad (1)$$

where F =Fertilizer Consumption (tonnes N)
E =Emission Coefficient (Tonnes N_2O-N released/tonne N applied)
f =Fertilizer type

Results are given as tonnes of N_2O emissions:

$$N_2O \text{ Emissions (tonnes } N_2O) = N_2O\text{-N Emissions} \times (\text{tonnes } N_2O\text{-N}) \times 44/28$$

A three year average of fertilizer consumption, centered on 1988, is the suggested point of reference in the OECD (1991) approach.

This methodology is based on the literature review and analysis by Eichner (1990) which reviews much of the N_2O emission research that was conducted before 1988. This review still covers most of the N_2O field flux measurements since relatively little work has been published since that time. Considering the number of agricultural systems that exist world-wide and the number of sources of N available for use, the data set available for these analyses is quite small. As a result single studies at single locations can dominate, and possibly skew the type of analysis used by Eichner (1990).

Another point is that most of the data cited by Eichner (1990) were from studies conducted only during the cropping season, or part of the cropping season, little is known about N emissions before planting in the spring and following crop harvest. Sommerfeld *et al.* (1993) found that appreciable amounts of N_2O are emitted from snow covered subalpine soils. Goodroad and Keeney (1984) measured large fluxes of N_2O during winter thaw periods.

The comments that follow should not be construed as demeaning the research cited or Eichner's (1990) analysis. In all cases the research was carefully conducted and the data representative of that found in the

Table 2. Effect of N-source on N_2O emission from three Iowa soils

Soil Texture	*N-Source*	N-Rate (kg N ha^{-1})	
		180[1]	250[2]
		N_2O-N (% of N added)	
Sandy loam	AA	2.4	4.8
	U	0.4	—
Clay loam	AA	1.1	6.0
	U	0.3	—
Loam	AA	1.2	7.8
	U	0.3	—

AA = Anhydrous ammonia, U = Urea;
[1]Breitenbeck and Bremner, 1986;
[2]Bremner *et al.*, 1981.

studies. The conclusion is that there is not enough variety in the data available to make a case for vastly differing coefficients for different fertilizers. The data in Table 2 illustrates this point. These data are from two studies conducted in Iowa on three different soils (Bremner *et al.*, 1981; Breitenbeck and Bremner, 1986). Both studies indicate high, but quite different N_2O emissions from anhydrous ammonia (AA), 1.1 to 7.8% of the N applied. Differences between soils and N application rates are large. The techniques used for the two studies were essentially the same, in application method of the AA and in flux measurements. The Bremner *et al.* (1981) study was initiated after cutting off soybean plants at the soil surface and leaving the plant residue on the soil. Did the rapidly decomposing plant residue provide an immediately available carbon source to enhance denitrification, that does not normally exist at the time of fertilizer application? No crop was growing during this study. One must question if this set of data is comparable to most systems and if it should be used in inventory calculations.

Other data indicate that overall N sources are not so important (Table 3). In 3 maize fields N_2O emissions ranged from 0.8 to 2.1% of the N applied from AA, ammonium sulfate (AS) and urea (U) fertilization. The span of variability of 0.8 to 2.1% was with U fertilization. Emissions from AA and AS fell within this range. As Byrnes *et al.* (1990) concluded from one of their studies, "N_2O emissions may be more closely related to soil properties than to the N source that is applied".

World-wide, ammonium based fertilizers are the major fertilizer N sources (FAO, 1990). Since the bulk of the data available do not indicate that ammonium-N sources dramatically influence N_2O emissions (Bouwman, 1990) it seems of little utility to worry about accounting for N source in the N_2O emission calculations, except where site specific data are available to warrant such calculations.

B. The second OECD/OCDE (1991) methodology includes the fertilizer source variable discussed in section A and also includes the crop type to which the fertilizer is applied. The approach is the same as in section A except that emissions of N_2O-N are summed over all fertilizer and crop types, instead of just over all fertilizer types.

$$N_2O\text{-N Emissions (tonnes } N_2O\text{-N)} = \sum_{fc}(F_{fc} \times E_{fc}) \quad (2)$$

where F=Fertilizer Consumption (tonnes N)
E=Emission Coefficient (tonnes N_2O-N released/tonne N applied)
f=Fertilizer Type
c=Crop Type

$$N_2O \text{ Emissions (tonnes } N_2O) = N_2O\text{-N Emissions (tonnes } N_2O\text{-N)} \times 44/28$$

Including crop type in the calculation seems reasonable since the type of crop tends to regulate soil water content, the timing of mineral N uptake, and the release of mineralizable carbon into the soil. All of these factors are regulators of N_2O-forming processes. If crop type is used what multiplication factor is appropriate? As noted in OECD/OCDE (1991) there is not enough information to calculate the necessary coefficients for each crop type.

Some of the data that are available (Table 3) is so variable that it is probably not useful for calculating emission coefficients. Timing of rainfall, frequency and intensity of irrigation, timing of N additions relative to crop development stage, and timing of N additions relative to water addition are factors that may dictate the amount of N_2O emitted. Some of these events are manageable, others are not.

Suggested N_2O emission calculation method.

In my view, the data available from which to calculate N_2O emission coefficients based on either N fertilizer

Table 3. Effect of N-Source on N_2O emissions from cereal cropped fields in Colorado, New York and England

Crop	*N-Source*	*N_2O-N Emission* (% of N applied)	*Reference*
Maize			
	AA	1.3	Mosier & Hutchinson, 1981
	AS	1.5	Mosier *et al.*, 1986
	U	1.6	Bronson *et al.*, 1992
	U	0.8	Bronson *et al.*, 1992
	U	2.1	Duxbury & McConnaughy, 1986
	CN	0.3	Duxbury & McConnaughy, 1986
Spring Barley			
	AN	0.6	Mosier *et al.*, 1982
	AS	0.4	Mosier *et al.*, 1986
	SS	1.4	Mosier *et al.*, 1982
Winter Wheat			
	AN	3.5	Burford *et al.*, 1981 direct seeded, clay soil
	AN	1.7	plowed, clay soil
	AN	0.9	direct seeded, clay loam
	AN	0.4	plowed, clay loam

AA = Anhydrous ammonia; AS = Ammonium sulfate; U = Urea;
CN = Calcium nitrate; AN = Ammonium nitrate; SS = Sewage sludge.

source or crop are not adequate to make such calculations. It is also unlikely that within the next few years sufficient studies will be conducted to make adequate coefficient calculations. Based on the reviews concerning N_2O emissions and their relationship to fertilizer applications (Bouwman, 1990; Eichner, 1990; CAST, 1992; Duxbury *et al.*, 1992; Robertson, 1992; Batjes and Bridges, 1992; and OECD/OCDE, 1991) I suggest simplifying the N_2O-fertilizer emission calculation:

$$N_2O\text{-N Emission (tonnes } N_2O\text{-N)} = \sum F \times 0.01 \quad (3)$$

where F = Fertilizer Consumption (tonnes N)

$$N_2O \text{ Emissions (tonnes } N_2O) = N_2O\text{-N Emission (tonnes } N_2O\text{-N)} \times 44/28$$

Because of the limitations of the data available and the scope of the data, a value of *1% /year of fertilizer* N evolving directly from agricultural fields does not seem unreasonable. The literature on field N_2O fluxes is adequate to provide the order of magnitude of the multiplication coefficient, that is greater than 0.001 and less than 0.1 (CAST, 1992).

There is certainly room for arguing the validity of this suggestion. For example in a flooded rice field, when fertilizer N is added immediately before flooding, little N_2O is emitted (Freney *et al.*, 1981). We do not know, however, how much N_2O evolves from the field when the water is drained for harvest or during the intercrop dry period. Some evidence indicates that appreciable N_2O is evolved from a rice field following a dry fallow period as the field becomes water saturated (Byrnes *et al.*, 1993). A simple equation relating soil mineral N content and soil% water-filled pore space to N_2O emissions integrated through the entire year may represent N_2O emissions reasonably well. There is, unfortunately, no possibility to link this to national inventory calculations.

Technical options for emission control

To increase agricultural production to meet growing demands for food required by the rapidly growing world population, N fertilizer use will necessarily increase. While N fertilization is not the only source of N_2O emitted to the atmosphere, it accounts for a

large part of the global budget. Direct and indirect emissions from N fertilization may total 2 to 3 Tg N y^{-1}. These N emissions can be in part controlled by management. Undesirable effects of fertilizer use on increased N_2O production can be mitigated by agricultural management without decreasing production; probably reducing rather than increasing costs (Mosier and Schimel, 1991)

CAST (1992) suggest a number of N management strategies:

- use soil testing to determine fertilizer N requirement; this will project and adjust for N mineralization from soil, legumes, manures, organic wastes, and any mineral N added by irrigation water or atmospheric deposition;
- dispense with the "maintenance" concept
- adjust the rate of N application to a reasonable yield goal for specific fields;
- place N fertilizers deep enough in the soil to lower the N_2O/N_2 ratio when denitrification occurs;
- time N application to when it is needed by the crop.

The rate and timing of fertilizer application should have a goal of leaving as little residual N as possible in the soil during the non-cropped periods of the ear. Additionally, agricultural systems that provide continuous plant cover should be utilized whenever feasible to minimize leaching and denitrification of nitrate associated with bare soils and to enhance nutrient recycling. In irrigated systems, better water management can be used to limit denitrification.

In irrigated systems, timing and frequency of irrigation also influence N_2O production (Rolston *et al.*, 1982). Large, less frequent irrigations result in lower N_2O production. Careful adjustments in irrigation scheduling are required, however, to minimize both N_2O emissions and nitrate leaching.

Multiple fertilizer applications or slow release fertilizer formulations will conceivably limit N_2O emissions, by controlling the nitrate supply subject to denitrification. A single application of a fertilizer formulation that provides mineral N to match crop uptake is a likely conservation mechanism. Field testing of this concept to limit N_2O production has not been conducted.

Since ammonium based fertilizers are the major fertilizer N sources world-wide (FAO, 1990), maintaining added N in the ammonium form should result in less N_2O production in fertilized soils. One mechanism of maintaining added N as ammonium is to apply a nitrification inhibitor (NI) with the fertilizer (Broadbent and Tyler, 1957; Bundy and Bremner, 1973; Braatz and Hogan, 1991). Using NI's frequently does not produce increased crop yields (Scharf and Alley, 1988) but studies suggest that NI's should decrease N_2O production from ammonium based fertilizers (Bremner *et al.*, 1981). It has been recognized for more than a decade that acetylene is one of the more effective nitrification inhibitors (Walter *et al.*, 1979; Bremner and Blackmer, 1978; Hynes and Knowles, 1978, Saharawat *et al.* (1987). However, McCarty and Bremner (1986) concluded that acetylene "has little, if any, potential practical value as a soil nitrification inhibitor", because no way existed to maintain appropriate concentrations over time in the field. They tested a number of acetylenic compounds and found that several, particularly 2-ethyenylpyridine and phenylacetylene had potential (McCarty and Bremner, 1986).

Since that time Banerjee and Mosier (1989) found that coating calcium carbide, with layers of waxes and shellac provided a slow release source of acetylene that has proven effective in limiting nitrification and increasing yield of flooded rice in India (Banerjee *et al.*, 1990) and in increasing cotton lint yield in Australia (Freney *et al.*, 1993). A number of recent field tests show that using nitrification inhibitors, such as acetylene, in conjunction with fertilizer N applications clearly decreases N_2O production (Bronson *et al.*, 1992; Keerthisinghe *et al.*, 1993) (Table 4). Decreased N_2O emissions were observed with NI's in both upland crops and flooded rice that were fertilized with urea.

Nitrification inhibitors also influence CH_4 flux in agricultural systems. Methane is generated from organic matter decomposition in flooded soils. However, atmospheric methane is oxidized in aerobic soils. The NI's nitrapyrin and acetylene both decrease methane oxidation in upland crops (Table 4). In fertilized upland systems, N_2O are relatively more important in the radiatively active trace gas balance than is the aerobic soil sink capacity so the decrease in overall "greenhouse" gas flux from upland soils is enhanced by NI's (Bronson and Mosier, 1993). Acetylene also inhibits CH_4 production in flooded rice systems (Table 4) (Bronson and Mosier, 1991; Keerthisinghe *et al.*, 1993), so it may be a possible way to increase fertilizer use efficiency while limiting CH_4 emissions from these systems.

There are a variety of management options available that may limit *direct* N_2O emissions from N-fertilized soils, but most of these options have not been field tested. Managing irrigation frequency, timing and quantity; applying N only to meet crop demand, either by multiple applications during the growing season or

Table 4. Effect of nitrification inhibitors on N_2O and CH_4 flux in temperate grasslands and cultivated, soils

	Gas Flux Rates[1]	
	N_2O gN $ha^{-1}d^{-1}$	CH_4 gC ha^{-1} d^{-1}
Native Grassland[2]	0.3	−6.3
Fertilized Grassland[2]	0.6	−4.1
Irrigated Maize[3]		
Urea	16.5	−0.6
Urea + NI[4]	4.6	−0.3
Control	1.1	−0.6
Irrigated Wheat[5]		
Urea	6.0	−0.9
Urea + NI[4]	2.5	−0.2
Control	2.0	−0.8
Dry Seeded Rice[6]		
Urea	73	3.0
Urea + NI[4]	14	4.3
Control	38	15.4
Control + NI[4]	16	5.8

[1]Mean of weekly flux measurements over 10 to 18 months, except for rice;
[2]*Mosier et al.* (1991);
[3]Bronson *et al.* (1992);
[4]Acetylene generated from encapsulated calcium carbide was the nitrification inhibitor in these studies;
[5]*Bronson and Mosier* (1993);
[6]Keerthisinghe *et al.* (1993), gas flux measurements of 37 days, between planting and permanent flooding 23 days later;

by using controlled release fertilizers; applying N only to meet crop needs and not soil maintenance; or using nitrification inhibitors to limit N_2O production. These are examples of management options which can limit *direct* N_2O emissions from N-fertilized fields while improving fertilizer use efficiency.

Agricultural management practices may not appreciably affect indirect N_2O emissions. Since we understand the magnitude or sources of indirect emissions even less than the direct emissions, it is not possible at this time to implement technical options for their control.

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