Nitrous oxide emission from Australian agricultural lands and mitigation options: a review

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Abstract

Increases in the concentrations of greenhouse gases, carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), and halocarbons in the atmosphere due to human activities are associated with global climate change. The concentration of N_2O has increased by 16% since 1750. Although the atmospheric concentration of N_2O is much smaller (314 ppb in 1998) than of CO_2 (365 ppm), its global warming potential (cumulative radiative forcing) is 296 times that of the latter in a 100-year time horizon. Currently, it contributes about 6% of the overall global warming effect but its contribution from the agricultural sector is about 16%. Of that, almost 80% of N_2O is emitted from Australian agricultural lands, originating from N fertilisers (32%), soil disturbance (38%), and animal waste (30%).

Nitrous oxide is primarily produced in soil by the activities of microorganisms during nitrification, and denitrification processes. The ratio of N_2O to N_2 production depends on oxygen supply or water-filled pore space, decomposable organic carbon, N substrate supply, temperature, and pH and salinity. N_2O production from soil is sporadic both in time and space, and therefore, it is a challenge to scale up the measurements of N_2O emission from a given location and time to regional and national levels.

Estimates of N_2O emissions from various agricultural systems vary widely. For example, in flooded rice in the Riverina Plains, N_2O emissions ranged from 0.02% to 1.4% of fertiliser N applied, whereas in irrigated sugarcane crops, 15.4% of fertiliser was lost over a 4-day period. Nitrous oxide emissions from fertilised dairy pasture soils in Victoria range from 6 to 11 kg N_2O-N/ha, whereas in arable cereal cropping, N_2O emissions range from <0.01% to 9.9% of N fertiliser applications. Nitrous oxide emissions from soil nitrite and nitrates resulting from residual fertiliser and legumes are rarely studied but probably exceed those from fertilisers, due to frequent wetting and drying cycles over a longer period and larger area. In ley cropping systems, significant N_2O losses could occur, from the accumulation of mainly nitrate-N, following mineralisation of organic N from legume-based pastures. Extensive grazed pastures and rangelands contribute annually about 0.2 kg N/ha as N_2O (93 kg/ha per year CO_2-equivalent). Tropical savannas probably contribute an order of magnitude more, including that from frequent fires. Unfertilised forestry systems may emit less but the fertilised plantations emit more N_2O than the extensive grazed pastures. However, currently there are limited data to quantify N_2O losses in systems under ley cropping, tropical savannas, and forestry in Australia. Overall, there is a need to examine the emission factors used in estimating national N_2O emissions; for example, 1.25% of fertiliser or animal-excreted N appearing as N_2O (IPCC 1996).

The primary consideration for mitigating N_2O emissions from agricultural lands is to match the supply of mineral N (from fertiliser applications, legume-fixed N, organic matter, or manures) to its spatial and temporal needs by crops/pastures/trees. Thus, when appropriate, mineral N supply should be regulated through slow-release (urease and/or nitrification inhibitors, physical coatings, or high C/N ratio materials) or split fertiliser application. Also, N use could be maximised by balancing other nutrient supplies to plants. Moreover, non-legume cover crops could be used to take up residual mineral N following N-fertilised main crops or mineral N accumulated following legume leys. For manure management, the most effective practice is the early application and immediate incorporation of manure into soil to reduce direct N_2O emissions as well as secondary emissions from deposition of ammonia volatilised from manure and urine.

Current models such as DNDC and DAYCENT can be used to simulate N_2O production from soil after parameterisation with the local data, and appropriate modification and verification against the measured N_2O emissions under different management practices.

In summary, improved estimates of N_2O emission from agricultural lands and mitigation options can be achieved by a directed national research program that is of considerable duration, covers sampling season and climate, and combines different techniques (chamber and micrometeorological) using high precision analytical instruments and simulation modelling, under a range of strategic activities in the agriculture sector.

Additional keywords: denitrification, greenhouse gases, fertilisers, pastures, cropping, carbon dioxide equivalent, nitrogen use efficiency, water-filled pore space.

Introduction

Over the 20th century, the global mean surface temperature has increased by about $0.6 \pm 0.2°C$ (IPCC 2001). This warming effect has primarily resulted from the increase in the concentrations of greenhouse gases (CO_2, CH_4, N_2O, and halocarbons) in the atmosphere due to human activities. For example, the atmospheric concentration of CO_2 has increased from 280 ppm in 1750 to 365 ppm in 1998, and N_2O has increased from 270 ppb to 314 ppb during this period (IPCC 2001). These gases absorb light in the infrared region, and therefore trap thermal radiation emitted from the earth surface. It is predicted that the global mean temperature is likely to increase by 1.4°C to 5.8°C during the next century if no action is taken to mitigate the emission rate of the greenhouse gases (IPCC 2001).

The global warming potential (GWP) of N_2O is 296 times that of CO_2 and almost 13 times that of CH_4 in a 100-year time horizon (IPCC 2001); and in spite of its lower rate of concentration rise (0.8 ppb/year from 1990 to 1999) than that of CO_2 (1.5 ppm/year), N_2O contributes approximately 6% of the warming effect caused by the increase of the various greenhouse gases.

Sources and sinks of nitrous oxide

Land and oceans are the principal natural sources of N_2O emissions, with soils contributing about 65% and oceans about 30% (IPCC 2001). N_2O is very stable in the atmosphere, with a lifetime of about 114 years (IPCC 2001). Probably the only significant process that removes N_2O is its reaction in the stratosphere with excited oxygen atoms formed by photolysis of ozone (Crutzen 1981). Microorganisms in soil can reduce N_2O into N_2 under anaerobic conditions (Ryden 1981). However, the significance of soil as a sink for N_2O remains uncertain and probably very small (Freney *et al.* 1978).

The Australian National Greenhouse Gas Inventory (NGGI) includes CO_2, CH_4, and N_2O gases. Annual emissions of N_2O exceeded 60 kt of N (94 kt of N_2O) in 1999 (Fig. 1), and nearly 80% was contributed by Australian agriculture (Australian Greenhouse Office 2001). Considering the GWP of N_2O as 296 compared with CO_2 (IPCC 2001), the CO_2-equivalent (CO_2-e) emissions from N_2O in 1999 were 28.31 Mt CO_2-e (Table 1), which was about 6% of the total CO_2-e emissions from Australia. The rate of N_2O emissions increased by 21% from 1990 to 1999; however, a much steeper increase was recorded in the last 3 years than in the 1990–96 period (Fig. 1) (Australian Greenhouse Office 2001). Thus, the rate of increase for N_2O is already more than 2.5 times that of allowable increases of 8% for all emissions under the Kyoto Protocol. This is a cause for concern and therefore requires concerted efforts in reducing N_2O emissions from Australian agriculture.

Of the total emissions of N_2O as CO_2-e of 28.31 Mt in 1999 (Table 1), almost 22.3 Mt CO_2-e was emitted from agricultural lands (16.17 Mt CO_2-e) and fire-related activities
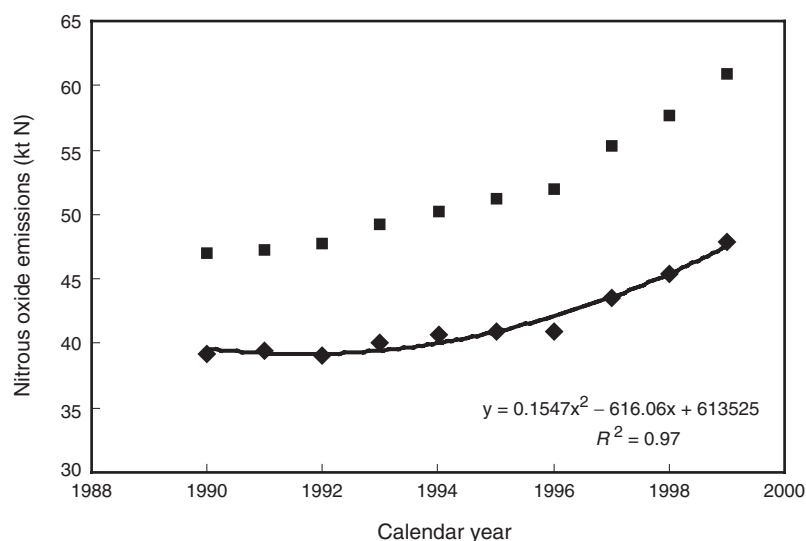
Fig. 1. Trends in N$_2$O-N emissions from Australia from 1990 to 1999: all sources (■) and agricultural lands (◆) (Australian Greenhouse Office 2001).

Table 1. Australian sources of nitrous oxide in 1999

Percentage of the total N$_2$O emissions from different Australian sectors is in parentheses

Source[A]	N$_2$O (kt N/year)	CO$_2$ equivalent (Mt/year)
Agriculture	47.92	22.27 (78.7%)
Land-use change and forestry	1.43	0.66 (2.3%)
Stationary combustion	2.09	0.97 (3.4%)
Transport and fugitive emissions from fuel	8.09	3.76 (13.4%)
Industrial plastics and solvents, etc.	1.39	0.65 (2.2%)
Total	60.92	28.31 (100%)

[A]Australian Greenhouse Office (2001).

(5.48 Mt CO$_2$-e prescribed woodland burning and 0.6 Mt CO$_2$-e from tree clearing) (Table 2) (Australian Greenhouse Office 2001). Soil and land based activities therefore account for the main sources of N$_2$O (about 80%) in Australia. However, large uncertainties exist in these estimates (Table 2). Uncertainties in estimates of direct emissions of N$_2$O from agricultural soils are caused by uncertainties related to both the emission factors and activity data, lack of coverage of measurements, spatial and seasonal aggregation, and lack of information on specific on-farm practices, as well as emissions not representative of all conditions.

The main sources of N$_2$O relating to soil and land-based activities include: N fertiliser use, soil disturbance and legume-based ley pastures, livestock excretory products (urine, faeces, and manure) (Table 3), and grassland and savanna burning (Table 2).

Nitrogen fertiliser use for agricultural production has increased sharply in recent years (Fig. 2). Nitrogen use for cereal crops increased by 314% from 1987 to 1996 and it further increased by 11% from 1996 to 2000 (Table 4). Although much smaller in amount, N fertiliser use for oilseed crops such as canola increased by 390% from 1996 to 2000. Since

Table 2.   Nitrous oxide emissions from Australian agriculture in 1999

Percentage of the total N_2O emission from different agricultural activities is in parentheses

Source[A]	N_2O (kt N/year)	CO_2 equivalent (Mt/year)	Uncertainty 95% confidence limit (%)
Agricultural soils	34.78	16.17 (72.6%)	−56 to +120
Prescribed burning of savannas	11.80	5.48 (24.6%)	−68 to +129
Crop residue burning	0.17	0.08 (0.3%)	−39 to +50
Manure management	1.17	0.54 (2.5%)	−7.6 to +22.8
Total	47.92	22.27 (100.0%)	

[A]Australian Greenhouse Office (2001).

Table 3.   Nitrous oxide emissions from agricultural soils in 1999

Percentage of the total N_2O emissions from different agricultural activities is in parentheses

Source[A]	N_2O (kt N/year)	CO_2 equivalent (Mt/year)
Fertiliser use	11.14	5.18 (32.1%)
Crop/pasture soil disturbance	13.38	6.22 (38.4%)
Animal faeces and urine	10.26	4.77 (29.5%)
Total	34.78	16.17 (100.0%)

[A]Australian Greenhouse Office (2001).

$$y = 6.6705x^2 - 26552x + 3E + 07$$
$$R^2 = 0.99$$
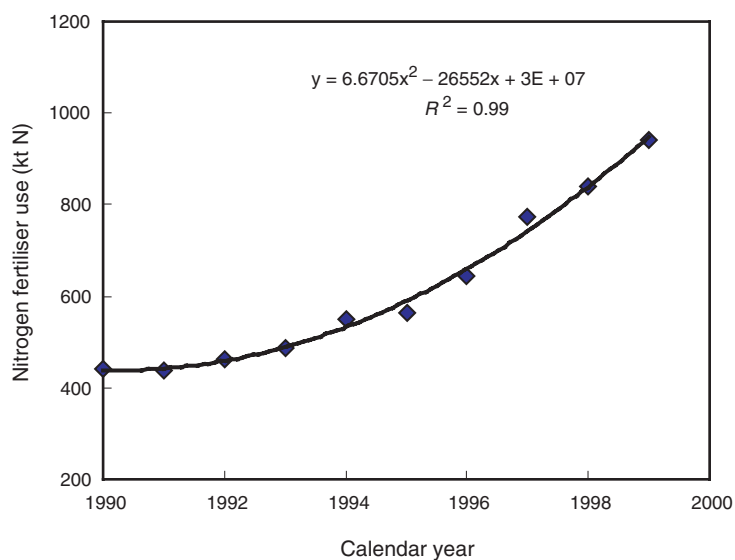
Fig. 2.   Trends in nitrogen fertiliser use in Australian agriculture (data from Chudleigh and Simpson 2001).

1996, cereal/oilseed crops accounted for more than 70% of the total N fertiliser use. For comparison, it was just over 50% of the total N fertilisers used in the agricultural sector 10 years earlier (1987). On the other hand, percentages of the N fertiliser applications to improved pasture and horticultural crops were about 12% each in 1987, but reduced to only

Table 4.   Estimates of fertiliser nitrogen use by different crops

Crop	Fertiliser N use (kt N)		
	1987[A]	1996[B]	2000[C]
Cereals	195	613	701
Sugarcane	58	100	96
Cotton	27	48	56
Oilseed/pulses	4	14	55
Horticulture	43	50	71
Pasture	44	59	76
Total	371	884	1055

[A]Bellingham (1989).
[B]Reuter (2001).
[C]Chudleigh and Simpson (2001).

6–7% each by 1996–2000. Overall, almost 90% of the increase in N$_2$O emissions from 1990 to 1999 was due to an increase in the rate of N fertiliser application (Australian Greenhouse Office 2001).

There are numerous reasons given for the increase in N fertiliser use for cereals/oilseed crops in recent years (Table 4). Some of these reasons are: (*i*) decrease in pasture area by 6 million ha in the last 10 years, hence lower legume N$_2$-fixed supply of N to the following cereal/oilseed crops; (*ii*) increased premium price paid for higher wheat grain protein content; (*iii*) introduction of disease-break crops such as canola, which itself demands higher N supply; and (*iv*) combined effect of technological changes on crop yields and N demand (Angus 2001).

Estimates of N$_2$O emissions from soil disturbance (cultivation, agricultural activities, ley/cropping) due to a decrease in ley/cropping area and shifting to continuous cropping showed a slight decline of 2.1% from 1990 to 1999. Animal faeces and urine accounted for an increase of 90% in the N$_2$O emissions from 1990 to 1999 due to increasing intensification of the livestock industries. However, there was a reduction in the field deposition of animal waste due to the reduction in sheep and cattle populations (Australian Greenhouse Office 2001).
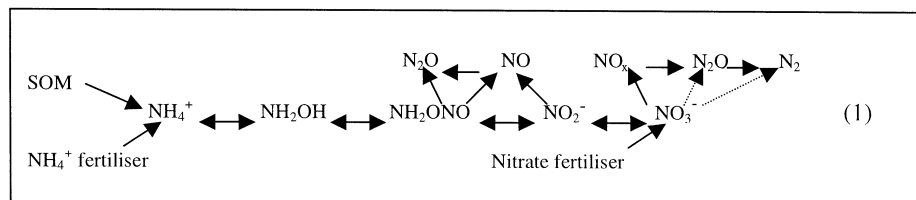
Prescribed burning of grasslands and savannas primarily in northern Australia aims at reducing the risk of uncontrolled fires during the dry winter–spring seasons in grazing lands. It is also supposed to rejuvenate pastures through recycling of P and K, which are generally low in the region's soils, although the soils get further depleted of N after every fire. The estimates of N$_2$O emissions from 1990 to 1999 showed an increase of 38% from savanna fires, although there is large uncertainty in the estimates of total area affected by fires (Australian Greenhouse Office 2001).

N$_2$O production in soil

Processes involved in N$_2$O production in soil

Nitrous oxide is produced in soil by at least 3 microbially mediated mechanisms: (*i*) nitrification, utilising nitrite as an alternative electron acceptor, thereby, reducing it to N$_2$O; (*ii*) dissimilatory nitrate reduction (denitrification); and (*iii*) assimilatory nitrate reduction. Microbial assimilatory nitrate reduction is of minor importance in soils (<6% of total nitrate reduction) because it is inhibited by very low concentrations of ammonium or soluble organic nitrogen present in soil. Dissimilatory nitrate reduction (denitrification) is

probably the main source of N_2O in soil (Tiedje 1994), although N_2O production by nitrification may sometimes be equally important (Granli and Bøckman 1994; Wang *et al.* 1997). The generalised Eqn 1 below shows the first 2 mechanisms of N_2O production in soil:

$$\text{SOM} \searrow \atop \text{NH}_4^+ \text{ fertiliser} \nearrow \quad \text{NH}_4^+ \leftrightarrow \text{NH}_2\text{OH} \leftrightarrow \text{NH}_2\text{ONO} \leftrightarrow \text{NO}_2^- \leftrightarrow \text{NO}_3^- \qquad \text{N}_2\text{O} \leftarrow \text{NO} \quad \text{NO}_x \rightarrow \text{N}_2\text{O} \rightarrow \text{N}_2 \tag{1}$$

The fourth mechanism implicated in N_2O production is the abiotic nitrite and nitrate reduction (involving organic matter and reductants such as ferrous iron), although its process mechanism is unclear and its overall contribution is uncertain (Venterea and Rolston 2000). This mechanism is favoured under acidic conditions and may occur under both aerobic and anaerobic conditions (Chalk and Smith 1983).

Factors controlling N_2O production in soil

Granli and Bøckman (1994) provided an excellent review of factors controlling N_2O production in soil. A brief description is given below.

Moisture and aeration

The rate of N_2O production from nitrification is normally low below 40% water-filled pore space (WFPS), but increases rapidly with increasing water content up to 55–65% WFPS (Fig. 3). Above 60–70% WFPS, an increase in water content hinders aeration (limits oxygen diffusion) and promotes denitrification, and release of both N_2O and N_2 occurs; the latter becomes a dominant form of gaseous N loss above 80–90% WFPS. Thus, the N_2O/N_2
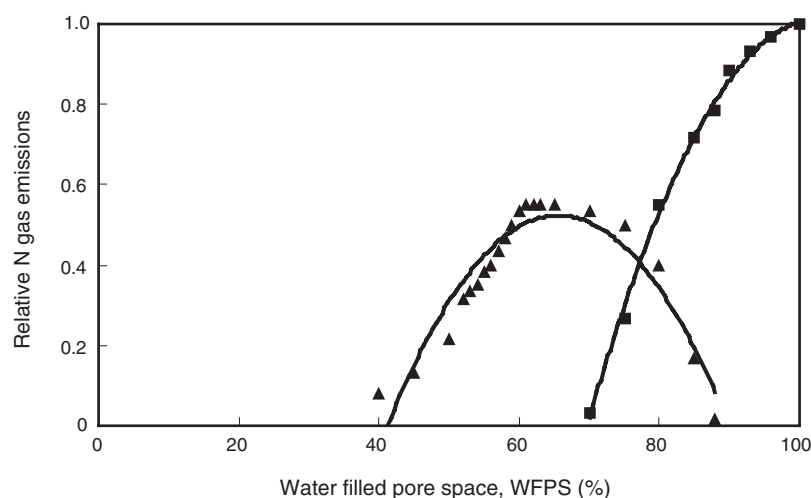
Fig. 3. A generalised relationship between water-filled pore space (WFPS) of soils and the relative fluxes of N_2O (▲) and N_2 (■) from nitrification and denitrification (see Eqn 1). (Adapted from Linn and Doran 1984 and Bouwman 1998.)

ratio decreases as the soil water content exceeds 75% WFPS (Davidson 1992; Weier *et al.* 1993*a*).

Alternate wet and dry cycles, which is the salient feature of the semi-arid Australian agricultural landscape, stimulate N mineralisation from organic matter, promote NO$_3^-$ accumulation during the dry period, and increase N$_2$O production during the wet period as long as the soil does not get waterlogged.

Temperature

Like other biological processes, nitrification and denitrification rates increase with increasing temperature within a certain range. Higher temperature favours a higher ratio of N$_2$O/NO$_3^-$ from nitrification (Goodroad and Keeney 1984). As soil temperature increases, N$_2$O emissions also increase, at least up to 37°C, but N$_2$O/N$_2$ ratio declines with increasing temperatures above 37°C (Keeney *et al.* 1979; Castaldi 2000). The principal mechanism for gaseous N production at higher temperatures (>50°C) is probably chemodenitrification of NO$_2^-$ that can be rapidly produced by thermophilic nitrate respirers (Keeney *et al.* 1979), resulting in primarily N$_2$ gas emission from soil.

Soluble and readily decomposable carbon

Organic materials such as plant litter, root exudates, manures, or native soil organic matter are the sources of carbon and energy for heterotrophic denitrifying organisms. Many studies have demonstrated that soil denitrification capacity increases with increasing organic C content, especially water-soluble C content (Burford and Bremner 1975; Drury *et al.* 1991; Iqbal 1992). Others have found that incorporation of plant materials enhances the rate of denitrification (Aulakh *et al.* 1991; Dorland and Beauchamp 1991). Although the ratio of N$_2$O/N$_2$ from denitrification decreases with increasing available C supply (Weier *et al.* 1993*a*), the total amount of N$_2$O produced from denitrification may be enhanced by the addition of organic materials.

In general, addition of degradable organic materials increases N$_2$O production in soils containing NO$_3^-$ or supplied with fertiliser NO$_3^-$ (Murakami *et al.* 1987). A high amount of N$_2$O can also be produced if materials containing degradable organic N (e.g. animal and green manures) are applied (Bremner and Blackmer 1981; Goodroad *et al.* 1984).

Soil and fertiliser nitrogen

Generally, the rate of denitrification increases with increasing NO$_3^-$ content in soil under conditions suitable for denitrification (e.g. high moisture) and when factors such as temperature and available organic C are not limiting. Under most circumstances, the presence of NO$_3^-$ inhibits the rate of N$_2$O reduction to N$_2$ (Fig. 4), which results in a higher N$_2$O/N$_2$ ratio at similar moisture and oxygen contents.

Under normal field conditions, nitrification is limited by the formation of NH$_4^+$ from mineralisation. Wang and Rees (1996) found that NH$_4^+$- or urea-N added to soils produced more N$_2$O than did NO$_3^-$ under aerobic conditions. A significant positive relationship between NH$_4^+$ content in soils and N$_2$O production was observed, confirming that nitrification is a significant source of N$_2$O under aerobic conditions.

Soil pH and salinity

Soil pH is a secondary controller of denitrification by mainly affecting the nitrification process. The optimal rate for nitrification as well as denitrification occurs at a pH range of about 7–8 (Bremner and Shaw 1958; Focht and Verstraete 1977; Haynes 1986). Although
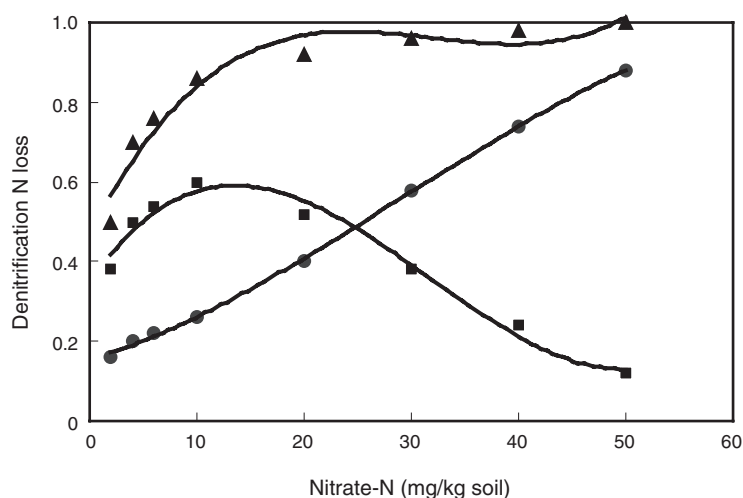
Fig. 4. A diagrammatic representation of the effect of the concentration of nitrate-N in soil on the relative N_2O (●) and N_2 (■), and total denitrification (▲) losses. (Redrawn from Mosier *et al.* 1983.)

NO_3^- reduction was detected at a pH as low as 3.5 (Focht and Verstraete 1977), denitrification declines if soil acidity shifts towards lower pH. Because the reduction of N_2O is inhibited more than the reduction of NO_3^- by acidic conditions, N_2O production is enhanced or even becomes dominant at pH <5.5–6.0 (Burford and Bremner 1975; Weier and Gilliam 1986). As the pH increases, denitrification products tend more, or completely, towards N_2 production (Focht and Verstraete 1977).

High salinity inhibits both nitrification and denitrification (Inubushi *et al.* 1999). N_2O reductase is susceptible to salt, which may result in N_2O accumulation from denitrification under saline conditions (Menyailo *et al.* 1997). Therefore, N_2O production from nitrification could be promoted by increased salt concentration (Low *et al.* 1997).

Limitation of nutrients other than nitrogen

Deficiency of other essential nutrients for plant growth limits the ability of plants to utilise ammonium and nitrate N. Hall and Matson (1999) observed that N_2O emissions from 15 kg N/ha applications were much higher from a phosphorus-limited forest soil than from soil high in plant-available phosphorus. Since many Australian soils are low in plant-available phosphorus, this has implications, especially for large areas of extensive grazing lands and rangelands, and forest soils. No Australian data exist to confirm these observations.

In summary, moisture, aeration, temperature, carbon supply, N mineralisation and nitrates, and soil pH and salinity affect the rate of N_2O emissions as well as the N_2O/N_2 ratio from agricultural soils. Although these factors interact, and the magnitude of interaction and hence N_2O emission varies both in time and space, such data from most Australian agricultural landscapes are mostly lacking or patchy.

Methods and their limitations in measuring N_2O emissions from soil

N_2O exchange between soil and the atmosphere is not difficult to measure with chamber techniques, but high spatial and temporal variability at local scales makes extrapolation to

even small landscape units very problematic. High spatial and temporal variability also hinders efforts to identify biological sources and controls of N$_2$O fluxes. The N$_2$O gas can be measured by infrared analysers (Denmead 1979) and gas chromatography using electron capture detectors, tunable diode laser, and molecular sieves (0.5 nm) (Van Cleemput *et al.* 1992; Kolb *et al.* 1995).

There are 2 main strategies for measuring N$_2$O flux: chambers placed on the soil surface for short periods, and micrometeorological approaches. Each has strengths and drawbacks.

Chamber methods

Chamber methods represent the most accessible techniques for measuring N$_2$O fluxes. Details are given by Mosier (1989), Hutchinson and Livingston (1993), Holland *et al.* (1999), and Meyer *et al.* (2001). In the simplest configuration, chambers are simply open-bottom cylinders or boxes placed on the soil surface for a period of time, usually 1–2 h, during which gases emitted from the soil accumulate within the enclosed headspace. The headspace is then sampled periodically, and the linear portion of the accumulation curve is assumed to be the gas flux rate. More sophisticated designs employ collars that are placed permanently in the soil; the flux chamber is then placed on the collar prior to sampling rather than inserted into the soil in order to avoid periodic soil disturbance, and in advanced designs, allow for automatic opening and closure (Meyer *et al.* 2001).

Although static flux chambers are inexpensive and easy to use, they generally cover only a small soil surface area and are deployed briefly because their use is labour-intensive and because they can affect the soil microclimate and thus gas flux if left in place for too long. Yet the consequences of missing an important flux event can be great, since the temporal distribution of N$_2$O fluxes tends to be log-normally skewed in most ecosystems. However, automated chambers (e.g. Brumme and Beese 1992; Wang *et al.* 1997; Ambus and Robertson 1998; Weitz *et al.* 1999; Galle *et al.* 2003) provide near-continuous measurements of gas fluxes without the labour expense of high-frequency field campaigns. They have been extremely useful for documenting short-term order-of-magnitude changes in fluxes from specific environments, and should be especially useful for parameterising quantitative models of N$_2$O flux. However, automated chambers are expensive to maintain and, perhaps more importantly, measure fluxes from relatively small areas, typically <0.25 m^2. This can be a significant limitation for whole-ecosystem estimates of N$_2$O flux because the short-range spatial variability of fluxes tends also to be high.

Because of the effort required to measure fluxes from more than a few chambers at a time, there are very few estimates of the within-site spatial variability of N$_2$O fluxes (Laville *et al.* 1999). Often temporal variability is dominated by sudden change in factors affecting N$_2$O emission events such as rainfall (Meyer *et al.* 2001). Capturing temporal and spatial variability for N$_2$O gas fluxes is thus very important to accurately quantify ecosystem-wide fluxes (Laville *et al.* 1999).

Micrometeorological methods

Researchers studying ecosystem fluxes other than N$_2$O, such as CO$_2$, NH$_3$, and CH$_4$, have turned to micrometeorology to integrate system-wide fluxes (Denmead 1983; Freney *et al.* 1983; Wofsy *et al.* 1993; Ham and Knapp 1998; Fowler 1999; Denmead *et al.* 2000). Micrometeorological approaches are based on measurements of moving air masses over ecosystems, and have the advantage over chambers of spatial and temporal integration. Fluxes are typically measured for areas as large as 1–10 km^2 to several million ha (Fowler

Table 5. A comparative estimation of N_2O emissions
(kg N/ha.year) from grazed pastures at Wagga Wagga,
NSW, using NGGI (1996) and IPCC (1996) methodology,
and micrometeorological methods (flux-gradient,
convective boundary layer, and mass balance)

Methodology	N_2O-N emission[A]
NGGI (1996)	1.0
IPCC (1996)	1.8
Micrometeorological:	
Flux-gradient	2.2 ± 1.2
Convective boundary layer	1.8 ± 1.4
Mass balance: $\varepsilon = 0.115$	6.4
Mass Balance: $\varepsilon = 0.039$[B]	2.7

[A]Calculated from Denmead *et al.* (2000).
[B]Excludes N_2O emissions following rainfall.

1999), and fluxes are measured continuously, not just while chambers are in place. Fluxes also include canopy interactions, which are important for some gases, such as CO_2 and NH_3.

Sensors mounted on towers detect the movement and gas content of air above and within the plant canopy; the rate and direction of flux can be determined from gas concentrations, gradients, and turbulence (Fowler *et al.* 1995; Lenschow 1995; Fowler 1999). Determination of CO_2 flux by eddy correlation, flux gradient, and convective and nocturnal boundary layers is now routine due to the advent of fast-reacting 3-D anemometers, continuous infrared gas absorption (IRGA) detectors, Fourier-Transform Infrared spectrometers (FTIR) (Griffith *et al.* 2002; Kelliher *et al.* 2002), or tunable diode lasers (Laville *et al.* 1999). However, these systems are not yet as sensitive as chamber-based approaches using electron capture detection systems. But for ecosystems in which annual fluxes are derived mainly from high short-interval flux events that are difficult to predict in advance, these approaches show promise for quantifying annual flux measurements for entire landscape units (Griffith *et al.* 2002).

Micrometeorological systems for N_2O have been deployed at only a handful of sites worldwide. However, where they have been deployed they have shown general agreement with short-term chamber measurements and sufficient sensitivity to detect fluxes in the systems examined (Christensen *et al.* 1996; Mosier *et al.* 1996; Laville *et al.* 1999). Very recent advances in FTIR (Griffith and Galle 2000; Griffith *et al.* 2002) suggest special promise for detecting modest fluxes of a suite of gases on a continuous basis at sites with available infrastructure (e.g. access to power and liquid nitrogen).

Denmead *et al.* (2000) measured N_2O emissions by micrometeorological methods from grazed pastures at Wagga Wagga, and found general agreement with fluxes calculated using NGGI (1996) and IPCC (1996) methodologies and micrometeorological approaches, especially the mass balance method (Table 5). These values are considerably larger than those estimated from NGGI algorithms (N_2O emission factors of 0.4% for urine and 1.25% for dung) but closer to those estimated from IPCC (1996) algorithms, which employ a N_2O emission factor of 2% for both urine and dung N (Denmead *et al.* 2000).

The major limitations of the micrometeorological methods include a high deployment cost and technical limitations associated with weather conditions. Weather limitations are related to boundary layer conditions and air turbulence and periods of rainfall. However, the potential benefits include long-term monitoring of fluxes to provide data input to model parameterisation for N_2O emission from specific landscapes, campaign measurements for

quantifying treatment or land use effects, field measurements to integrate spatial heterogeneity of N$_2$O emissions, and complementary other trace gas fluxes to assist in interpreting field measurements (Fowler 1999; Lavelle *et al.* 1999; Griffith *et al.* 2002).

For the foreseeable future it is likely that a combination approach will work best—one that uses chambers to measure low fluxes and to make comparisons among different habitats and land management strategies, and that also uses micrometeorological approaches to spatially integrate fluxes. Both automated chambers and micrometeorological approaches can provide important information about temporal fluxes, especially across environmental events that trigger high fluxes.

N$_2$O emissions from various ecosystems

N$_2$O emissions from agricultural systems

Cropping soils

Flooded rice cultivation. Rice is primarily grown in Riverina area of New South Wales and the Ord River irrigation area in Western Australia. By world standards, Australian rice yields are high (>10 t/ha), and therefore, the crop demands high rates of nitrogen fertiliser application to meet its N needs. Since the soil is usually waterlogged, N gaseous losses are expected to be mainly as N$_2$ gas (Fig. 3). Often, the limited supply of soluble and readily available C, low nitrate concentration, and competing crop demand for N limit the total gaseous N losses from flooded rice system.

For example, application of ammonium sulfate at 80 kg N/ha to flooded rice resulted in only 0.1% as N$_2$O loss (Freney *et al.* 1981). It is estimated that N$_2$O emission from the flooded rice can be as low as 0.02% from 80 kg N/ha of urea application (Mosier *et al.* 1989), provided soils are flooded for a number of days before N fertiliser is applied. This loss could be an underestimate due to the N$_2$O dissolved and retained in water and not measured (Heincke and Kaupenjohann 1999). Thus, total N$_2$O losses could be higher than measured from short-period observations. Also, when soil experiences wetting–drying cycles such as in dry-seeded rice, a common practice in the Riverina region, it could be a source of N$_2$O if mineral N (ammonium and/or nitrate) is also present in soil. For example, over an 18-day period after flooding, 0.8–1.4% of nitrate-nitrogen present initially in soil was lost as N$_2$O (Denmead *et al.* 1979*a*).

Secondary emissions of N$_2$O could occur from the deposition of NH$_3$ volatilised from flooded rice fields after the application of ammoniacal fertilisers, since ammonia has a short lifetime in the atmosphere. In flooded rice, NH$_3$ volatilisation losses can account for 20% to >80% of the total fertiliser N losses (Mosier *et al.* 1989).

Sugarcane cropping. Main areas of sugarcane cultivation are coastal Queensland and northern New South Wales, and the Ord River irrigation area in Western Australia, thus encompassing tropical to subtropical environments.

About 10% of N fertiliser is used for sugarcane production (Table 4). The rates of fertiliser application vary from 100 kg N/ha to 300 kg N/ha. It is estimated that sugarcane soils emit more than 10 kt of N$_2$O-N per year (Weier 1998). Improved crop management practices such as sugarcane trash retention and no-till have improved the productivity of sugarcane soils. However, it may not have reduced the total N$_2$O emissions from the sugarcane lands. For example, nitrate applied to a sugarcane soil at the rate of 160 kg N/ha lost 0.13% and 15.4% of N as N$_2$O from the cultivated and no-till soil, respectively, over a 4-day period after fertiliser application and irrigation. Although N$_2$ is the major product of denitrification loss of fertiliser N from fine-textured

soils, N_2O becomes a major gaseous N loss when soil nitrate concentrations are high (Weier *et al.* 1996; Fig. 4).

Split application of N fertiliser may reduce the rapid N_2O losses. Split application of urea and ammonium sulfate to sugarcane soil at 80% WFPS and 100% WFPS results in lower N_2O emissions initially compared with full application of urea, although total N_2O emissions over a given season are not significantly different (Weier 1999).

Application of sugarcane trash (10 t/ha) to soil fertilised with KNO_3 or urea at the rate of 160 kg N/ha, and followed by 50 mm of irrigation, increased N_2O emissions, and CO_2 respiration, especially from KNO_3. The mean values were 13.3 kg N_2O-N/ha.day for the trash-retained soil compared with 11.3 kg N_2O-N/ha.day without trash retention (Weier 2000).

Nitrous oxide emissions from N added to sugarcane soils of different texture differed. For example, more gaseous N was lost from a fine-textured soil than from a coarse-textured soil; 3.2–19.6% of the applied N was lost from a fine-textured soil, compared with <1% from a coarse-textured soil, with 45–78% appearing as N_2O gas. The N loss was closely related to the soil nitrate concentration and the soil water content (Weier *et al.* 1996).

Measured N_2O emissions from the sugarcane soils in northern NSW and Queensland are among the highest from agricultural soils. This may be due to 3 reasons: high fertiliser N rates, high soil moisture regimes, and high available C source, especially from green trash retention. Comparable N_2O emission rates have been measured in intensively managed (high fertiliser N rates and irrigated) dairy pastures in Victoria (see below; Eckard *et al.* 2001) and in vegetable-cropped fields in Britain (Ryden and Lund 1980).

Cotton cultivation. Although cotton is grown in drier areas than sugarcane, large areas of cotton crop are flood-irrigated, thus increasing the frequency of anaerobic conditions. Nitrogen fertiliser applications are also higher in irrigated cotton than in rain-fed cotton, often in excess of 100 kg N/ha (Rochester and Constable 2000). Also, cotton is grown in warmer conditions than sugarcane. It is therefore expected that N_2O emissions will be substantial from irrigated cotton areas (Granli and Bøckman 1994), although no detailed studies are available for Australia.

Grain cropping. Cereal and oilseed crops are grown on about 22 million ha, and another 23 million ha carries improved pastures, which are brought under cultivation every 3–4 years. In south-eastern and Western Australia, improved pastures with a significant legume component historically provided a large part of the N supply to cereals, whereas in northern New South Wales and Queensland, soil N was the primary source of crop N. Reduced area under improved pastures and depleting N supply from Australian soils have resulted in increasing use of N fertilisers for cereal/oilseed cropping, since grain legumes only occupy 2 million ha compared with a 6 million ha reduction in improved pastures (Angus 2001). The combined N_2O emissions from cropping and ley farming soils were in the order of 24.5 kt N/year (11.9 Mt CO_2-e) in 1999, and are probably increasing every year. Available N resulting from N fertiliser application or from the mineralisation of soil N and legume N essentially behaves similarly, provided soil water and aeration, temperature, soluble and readily available C, pH, salinity, and soil texture are also similar.

Loss of N as N_2O from fertiliser as well as from soil mineral N varies considerably. Application of urea to 10 soils resulted in N loss as N_2O ranging from none to 9.9% of the fertiliser application. Nitrous oxide losses were significantly correlated with nitrite concentration, soil pH, and organic carbon content (Magalhaes and Chalk 1987; Magalhaes

et al. 1987). About 5.8% N as N$_2$O was emitted from urea applied at 600 mg N/kg soil to an acid loamy sand (Magalhaes and Chalk 1987).

Provided nitrates are present in soil, crop growth generally increases N$_2$O emission by increasing the oxygen demand due to the presence of decomposable organic matter and root exudates. Restricted pore space in many cropping soils further retards oxygen diffusion as the water content in the soil increases, resulting in substantial N$_2$O emissions (Burford and Stefanson 1973; Stefanson 1973). Once the soil is waterlogged, N$_2$ is the main source of denitrification losses. Also, as nitrate concentration in soil is depleted due to the crop uptake, N$_2$O emissions also decrease.

Crop residue retention (stubble retention) and minimum till are increasingly practiced in the Australian cereal belt. This practice changes the soil nitrate levels, soluble and readily available organic matter, soil water content and aeration, and, frequently, soil surface temperature. These changes in soil management have the potential to make the N$_2$O emissions significantly different from those of conventional cultivation (Linn and Doran 1984; Robertson *et al.* 2000). For example, addition of wheat straw at the rate of 10.5 t/ha to a Vertisol doubled the rate of denitrification loss of applied urea. The N$_2$O emissions were about 0.5% in Vertisol and 3.0% in Alfisol of applied N when straw was added, and remained higher than from the conventional till treatment, although total denitrification rates were similar in both soils (Avalakki *et al.* 1995). In Western Australia, texture-contrast soils (Chromosols) experience temporary waterlogging in winter and have the potential to denitrify and emit N$_2$O (Bronson and Fillery 1998).

Soil temperature effects on N$_2$O emissions are noticed when seasonal effects on N$_2$O emissions from cropping soils are studied. Decreasing temperature from 30°C to 5°C slowed the rate of N$_2$O emission from a Vertisol and an Alfisol; but the N$_2$O/N$_2$ ratio increased (Avalakki *et al.* 1995). However, in subtropical environments, the temperature effects appear to be less marked. For example, almost 75% of N$_2$O emissions occurred in equal amounts in spring and summer under black gram (*Vigna mungo*) in subtropical Queensland. About 88% of the variation in N$_2$O emissions is found to be associated with monthly rainfall and nitrate concentration (Weier *et al.* 1991). The microbial population of N$_2$O producers appears to be controlled by the soil water content under a black gram (*Vigna mungo*) crop (Weier *et al.*1993*b*).

Emissions of N$_2$O from the nitrification and the denitrification processes may occur at different times. When 120 kg N/ha as urea was applied in irrigation water to sunflower growing on a calcareous Vertisol, about 2 kg N/ha was emitted as N$_2$O during the first 23 days after fertiliser application—about half in the first 11 days and the other half after the second irrigation, when most of the fertiliser was present as nitrate. Thus, an almost equal amount of N as N$_2$O was lost from ammonium initially and then from nitrate after the second irrigation (Freney *et al.* 1985).

Ley cropping. Pastures in rotation with crops have been successfully used for soil fertility restoration and animal grazing in temperate and Mediterranean areas of southern and south-western Australia. They are also increasingly being used in north-eastern Australia as the native soil fertility declines under continuous cereal cropping (Weston *et al.* 2000). Legume-based ley pastures have made significant contribution to soil fertility and the crops following these pastures gain N benefits from the mineralisation of legume N after termination of the pasture phase.

Under the pasture phase, N$_2$O emissions from annual Wimmera ryegrass (*Lolium perenne*) and subterranean clover (*Trifolium subterraneum*) pastures were estimated to vary from 0.35 ± 0.03 kg N/ha.year (Galbally *et al.* 1994; Wang *et al.* 1997) to 2.2 ± 1.2 kg

Table 6.   Estimates of mineral N (mostly nitrate-N) produced during 3–8 months following the termination of the pasture phase and N$_2$O emission due to denitrification

Previous pasture	Location	Soil type	Soil depth (m)	Mineral N (kg /ha)	Calculated emission[A] N$_2$O (kg N/ha)	CO$_2$-e (kg/ha)
Legume + grass[B]	Warra, Qld	Vertisol	0–1.2	147	0.7	340
Lucerne[B]	Warra, Qld	Vertisol	0–1.2	123	0.5	230
Medic[B]	Warra, Qld	Vertisol	0–1.2	96	0.3	135
Pasture[C]	Junee, NSW	Kandosols	0–1.5	148	0.7	340
Pasture[D]	SA	Chromosols	0–0.6	79	0.4	185
Subterranean clover based[E]	Moora, WA	Deep sand	0–1.4	106	0.5	250

[A]Per cent denitrification measured for winter-grown legume, 12%; perennial legume, 16%; and legume + grass pasture, 20% (Pu *et al.* 1999); and 2.5% of the total denitrification loss is assumed to be N$_2$O emission (Avalakki *et al.* 1995).
[B]Hossain *et al.* (1996).
[C]Angus *et al.* (1998).
[D]Xu *et al.* (1996).
[E]Anderson *et al.* (1998).

N/ha.year from a lucerne–ryegrass pasture near Wagga Wagga, NSW (Denmead *et al.* 2000); the latter values were apparently affected by recent grazing. Meyer *et al.* (2001) and Griffith *et al.* (2002) reported that N$_2$O emissions were <1 kg N/ha.year from lucerne pastures in temperate Australia. Since most non-dairy pastures in Australia are not N fertilised, the N$_2$O emission estimates are within the range of those from unfertilised soils (1–2 kg N/ha.year) in the Northern Hemisphere (Bouwman 1994). However, information is lacking on legume-based pastures in subtropical and tropical Australia, with summer-dominant rainfall.

The termination of a ley pasture containing legumes for the cropping phase mineralises a substantial amount of organic N into mineral N (Table 6), and also produces soluble organic C from litter and root decomposition. In Table 6, a conservative estimate of N$_2$O emission from denitrification following the termination of the pasture phase indicates losses about similar to or higher than those during the pasture phase (Galbally *et al.* 1994; Wang *et al.* 1997; Meyer *et al.* 2001; Griffith *et al.* 2002). Waterlogging of these soils during and after organic N mineralisation leads to a very high potential for denitrification (12–89%), including N$_2$O emission, especially in summer-dominant rainfall regions (Pu *et al.* 1999).

No account was taken of the N$_2$O emission during nitrification in Table 6. Wang *et al.* (1997) estimated that 55–73% of the total soil N$_2$O emission occurs during aerobic nitrification. Thus N$_2$O emissions during nitrification of residue N from legume-based pastures will be twice the estimated N$_2$O emission, i.e. from 0.6 kg N/ha to 1.4 kg N/ha, after the termination of the pasture phase.

Increasing use of lucerne is being made for N$_2$ fixation and for lowering the watertable in cropping regions, especially where salinity may become a problem. Lucerne is very effective in fixing substantial amounts of atmospheric N$_2$, often exceeding 100 kg N/ha.year (Peoples and Baldock 2001; Table 6). However, after termination of the lucerne phase, nitrate builds up rather rapidly, especially in subtropical environments (Hossain *et al.* 1996). When soil experiences wet and dry cycles during the summer months, nitrate may be lost due to denitrification, including N$_2$O emission. A somewhat similar situation

may occur in southern and Western Australia in late summer and the early autumn period. It has implications for the termination time for a legume phase, especially in subtropical and tropical regions, and for management of legume residues in other environments. Unfortunately, data are lacking to confirm these statements.

Similarly, the extent of N$_2$O emissions during and following a pulse crop in the Australian cereal belt is unknown.

Horticultural soils

Nitrogen application rates on horticultural soils are generally high, often exceeding 200 kg N/ha.year for banana plantations (Reuter 2001). Although the denitrification rates may be low since waterlogging is avoided, the N$_2$O emissions are likely to be high due to high fertiliser rates. Also, many horticultural soils, especially in eastern Australia, are acidic. The N$_2$O emissions measured elsewhere suggest that N$_2$O may be the dominant denitrification product from acidic soils (Ryden and Lund 1980). On banana plantations, loss rates for N$_2$O are estimated to range from 1.3% to 2.9% of applied N (Bouwman 1998). In vegetable crops, N$_2$O emissions ranged from <2% to >10% of fertiliser N applied (Ryden and Lund 1980); thus, they are much higher than from the grain cropping systems.

N$_2$O emissions from grazing systems

Intensive grazing (dairying)

Historically, most of the N supply for dairy pastures was derived from pasture legumes. With increasing competition in the dairy industry, N fertiliser is being increasingly used to enhance productivity. Since optimum production is achieved at 450–600 kg N/ha.year (Eckard 1998), pasture legumes, which fix about 20–200 kg N/ha.year (Peoples and Baldock 2001), do not supply enough N for optimum production; hence N fertiliser application is needed.

Over 60% of farms in south-eastern Australia now apply up to 200 kg N/ha.year (Eckard 1998), and in New South Wales and Queensland about 100–150 kg N/ha.year is applied to subtropical pastures and up to 300 kg N/ha.year to temperate pastures (Cowan *et al.* 1995). Besides, another 30–50 kg N/ha.year is supplied as feed supplements.

Nitrous oxide emissions from dairy pasture soils in Victoria vary from 6 to 11 kg N/ha from 0 and 200 kg N/ha applications, respectively (Eckard *et al.* 2001). At 2 intensively grazed pastures in New Zealand, N$_2$O emission losses varied from 0.5 kg N/ha.year on a low fertility site to 3 kg N/ha.year on a high fertility site. These losses were associated with soil water content, nitrate and ammonium concentration, and temperature. It is suggested that grazing animals have a significant effect on emission through hoof damage on wet soils (Carran *et al.* 1995), especially in a warmer environment. Peak losses of N may occur in early spring and autumn in the Mediterranean environment, and lower N losses in summer due to low soil water contents. However, in subtropical and tropical environments, large N losses are also likely to occur in summer.

Higher N losses are expected from flood-irrigated pastures but no Australian data are available on N$_2$O emissions over an extended period. Prasertsak *et al.* (2001) estimated that 40% of applied urea at the rate of 115 kg N/ha was lost from an ungrazed dairy pasture in tropical Queensland; about half was lost as ammonia volatilisation and the other half presumed to be denitrified, although the proportion of N$_2$O emissions is not known. Addition of as little as 5 mm of water to a grass sward increased N$_2$O emission rates markedly. Nitrous oxide was produced near the soil surface, and was accompanied by a net

increase in soil nitrate (Denmead *et al.* 1979*b*). With the soil moist to less than field capacity (~60% WFPS), average daily emissions ranged from 6 to 25 g N/ha.day (Denmead *et al.* 1979*b*). Thus, the population of N_2O producers appears to be controlled by the soil water content (Weier *et al.*1993*a*). In the USA, almost 37% of N (about 240 kg N/ha) was lost from the irrigated pastures (Barton *et al.* 1999). Studies on N_2O emissions from irrigated pastures are required to estimate N_2O losses, both in the temperate and subtropical regions.

Another source of N_2O emissions from the dairy industry is the application of dairy waste to pastures, including manure and effluents. Both components are high in soluble C and hence have a potential to produce N_2O when applied to soil, although overall impact on total N_2O emissions is small (Australian Greenhouse Office 2001).

In summary, N_2O emission rates from arable cropped soils for sugarcane, cotton, grain production, and leys and intensively managed pastures vary from <1 kg N_2O-N/ha.year from extensive pastures and leys to >10 kg N_2O-N/ha.year from intensively managed dairy pastures and sugarcane crops. Factors causing large differences in N_2O emissions from these agricultural systems, e.g. for high emissions, include high N fertiliser application rates, wetter moisture regimes, and large plant residue additions. There may be minor effects of soil pH, especially soil acidity for sugarcane soils.

Extensive grazing (improved pastures v. rangelands)

Large areas of rangelands are used for extensive grazing, although in many areas exotic pastures have been introduced since tree clearing to enhance pasture productivity. The N_2O emissions are likely to be low in most rangelands; the values reported range from 0 to 2 kg N/ha.year, with a median value of 0.2 kg N/ha.year (Galbally *et al.* 1992; Denmead *et al.* 2000). Although the rates of N_2O emissions from rangeland soils may be low, considering the extent of the total rangeland area (>400 million ha) it is important to measure N_2O emissions in the field over an extended period, since small amounts of rainfall may trigger significant nitrification activity and hence N_2O production following a prolonged dry period. For tropical savannas the rates could be much higher, but data are lacking.

Introduction of exotic species may result in increased pasture productivity and thus increased soluble C even in deeper layers. For example, in a green panic (*Panicum maximum*) pasture, significant amounts of N_2O emissions were measured at 50–55 cm depth in the field (Weier *et al.* 1991, 1993*b*).

Soil temperatures in rangelands vary considerably not only between seasons but also diurnally. The N_2O emissions show a diurnal cycle, varying with the temperature of the surface soil. Peak N_2O emission rates occur in the afternoon and are minimum near sunrise. Seasonal variation shows the largest emissions of N_2O in spring in temperate Australia (Denmead *et al.* 1979*a*). Almost 75% of N_2O emissions occur in spring (55%) and summer (20%). These are associated with CO_2 concentration due to microbial respiration (Weier *et al.* 1991), and hence intense oxygen demand, resulting in partial anaerobiosis.

The N_2O emissions from the urine patches could be significant in both improved pastures and rangelands (Galbally *et al.* 1994; Wang *et al.* 1997; Bronson *et al.* 1999). Usually, animal faeces are voided in large areas, thus avoiding anaerobic N mineralisation from animal waste. The IPCC (1996) default value for 1.25% of total N voided may be different for these systems and needs to be evaluated again after strategic field measurements.

In summary, N_2O emissions from extensively managed pastures and rangelands are <1 kg N_2O-N/ha.year, with an average value of about 0.2 kg N_2O-N/ha.year. However, data

are lacking from tropical savannas, where fire is frequently used as a pasture management tool. In such a system, where N is mineralised rapidly after fire, N$_2$O emission rates could be higher, especially if fire is followed by moderate rainfall events.

N$_2$O emissions from forestry systems

Nitrous oxide emissions from natural forests in temperate region are <0.2 kg N/ha.year (Khalil *et al.* 1990), although N$_2$O emissions following fire or prescribed burning of a eucalyptus forest are enhanced (Meyer *et al.* 1997). For tropical forests, however, the median rate of N$_2$O emission is reported to be 1.7 kg N/ha.year (Bouwman 1998).

Annual denitrification rates for plantation forest soils vary, depending on stand age, site preparation, fertilisation, and wastewater application. Increased denitrification rates after clear felling and N fertilisation are likely to be higher than from the natural forest but probably lower than from rangelands. The rates of denitrification may range from 0.3 kg N/ha.year to 13 kg N/ha.year under N fertilisation (Barton *et al.* 1999). However, exact measurements of N$_2$O emissions or N$_2$O/N$_2$ ratio are rare under Australian forests and plantation soils.

N$_2$O emissions from other systems

Termite mounds emit significant amounts of N$_2$O during spring and summer (Khalil *et al.* 1990; French *et al.* 1997). However, the exact contribution of termite mounds to N$_2$O emissions is not known since it is calculated from the estimates of CH$_4$ production as a ratio of CH$_4$/N$_2$O of 20:1 on a mass basis (Galbally *et al.* 1992).

In summary, N$_2$O emissions from agricultural lands in Australia vary widely, from 0.2 kg N$_2$O-N/ha.year from rangelands and eucalyptus forests to 10–15 kg N$_2$O-N/ha.year from irrigated pastures and sugarcane lands. This variability in N$_2$O emissions is primarily attributed to N fertiliser inputs and soil water regime, which favour higher emissions from highly fertilised sugarcane lands and irrigated pastures. The increasing trend in N fertiliser use for grain cropping is also likely to lead to significant N$_2$O emissions on a national scale since these lands occupy an area 1–2 orders of magnitude greater than sugarcane and irrigated pastures.

Mitigating N$_2$O emissions in various ecosystems

Mitigating N$_2$O emissions from agricultural systems

Australian broadacre agriculture, which produces wool, sheepmeat, beef, cereals, and other broadacre crops, also produces significant emissions of greenhouse gases (Phipps and Hall 1994). The marginal cost of reducing greenhouse gases from Australian agriculture by 20% is estimated to be \$20/t of CO$_2$ equivalent, or \$4/t for N$_2$O mitigation alone (Phipps and Hall 1994). Increasing nitrogen use efficiency should reduce gaseous losses of N, including that of nitrous oxide.

Flooded rice cropping

In waterlogged or flooded soils, where aeration is restricted, less N$_2$O is emitted into the atmosphere because more N$_2$O is converted into N$_2$ through the denitrification process. Since mineral N content governs the N supply both to crops and the denitrifiers, increasing the efficiency of mineral N use to crops should result in a lower amount of mineral N available for denitrification. Addition of nitrapyrin to soil after anhydrous ammonia application of 60–80 kg N/ha significantly reduced the loss of N$_2$O only in a calcareous

soil, which accumulated nitrite in the fertiliser band (Magalhaes *et al.* 1984). Wax-coated calcium carbide significantly reduced the rate of N_2O emission, whereas nitrapyrin was much less effective in a flooded rice system (Keerthisinghe *et al.* 1993). Denitrification and N_2O losses of urea from a flooded rice system were further reduced when urea was deeply placed, compared with surface-broadcast application (Keerthisinghe *et al.* 1993).

The strategies that increase the efficiency of N fertiliser use will reduce N_2O emissions (Aulakh *et al.* 1992; Mosier *et al.* 1996). These strategies include: form (reduce anhydrous ammonia use), rate, and method of application; matching N supply with demand; supplying fertiliser in the irrigation water; applying fertiliser to the plant rather than to the soil; and the use of slow-release, and urease and nitrification inhibitors (Freney 1997*a*).

Although these approaches enhance nitrogen use efficiency, they do not necessarily reduce N_2O emissions significantly. However, newly developed urease and nitrification inhibitors and coated-urea have the capacity to prevent loss of N, including the loss as N_2O, as well as increase crop yields (Freney 1997*a*).

Arable cropping

The above principles apply to all agricultural systems. In arable systems, soil structure (soil texture), nitrate, soluble and readily available C, and water content appear to be major factors that affect the N_2O/N_2 balance between N_2O diffusion into the atmosphere and its further reduction to N_2 gas (Weier 1998).

Nitrate concentration in the soil can be kept at low levels by applying N fertiliser to the crop when the crop needs it and in an amount that the crop can readily utilise, which is monitored through regular soil analysis for mineral N and crop monitoring for N, split applications, and applying in drip irrigation. Further reductions in N_2O emissions may be achieved by the use of cover crops during the fallow period to remove residual nitrate from the soil profile (Weier *et al.* 1998). However, these practices will increase costs of crop production.

Similar to flooded-rice soils, urease and nitrification inhibitors such as wax-coated calcium carbide (Keerthisinghe *et al.* 1993), etridiazole (Rochester *et al.* 1994), dicyandiamide, and 3,4-dimethylpyrazole phosphate (Weiske *et al.* 2001) have been found to reduce nitrate concentration in arable soils after the application of ammoniacal fertilisers, and thus reduce N_2O emissions. For example, Weiske *et al.* (2001) found that dicyandiamide and 3,4-dimethylpyrazole phosphate reduced N_2O emissions from the fertilised plots over a 3-year period by 29% and 45%, respectively. However, the cost-effectiveness of the latter 2 nitrification inhibitors needs to be further investigated, in terms of economics of use and N use efficiency but also N_2O emission reduction benefits.

The claimed benefits of 3,4-dimethylpyrazole phosphate in reducing CO_2 fluxes and increasing CH_4 oxidation (Weiske *et al.* 2001), if confirmed in further field experiments, could provide further impetus to the use of nitrification inhibitors. Similarly, the usefulness of urease inhibitors such as Agrotain (*n*-butyl-thiophosphoric triamide) also needs to be tested (McGuffog 2001), since they may have an added benefit of reducing N_2O emissions in the nitrite formation process.

Mitigating N_2O emissions from grazing systems

Dairying

Since N fertiliser use is likely to increase on dairy pastures, N fertiliser management practices must ensure that pastures efficiently use N. Therefore, strategies are required for

fertiliser management generally similar to that required for cropping, mentioned above, with a major significant difference. Unlike crops, most established pastures have an active extensive root system throughout the growing season. By monitoring pastures and soil mineral N, it is possible to arrive at the pasture N requirement in a given season commensurate with the pasture growth. Then N fertiliser application must be just enough to meet the pasture N requirement. Thus, mineral N in soil under pasture can be kept low, which may result in low N$_2$ and N$_2$O losses from irrigated pastures (Eckard *et al.* 2001). However, N fertiliser applications to dryland pastures with episodic and unreliable rainfall provide a challenge for efficient N management.

Waterlogged pastures can be better managed if flood irrigation is avoided and sprinkler irrigation is introduced for irrigation as well as for N fertiliser application. This also avoids hoof damage to soil structure and thus maintains oxygen diffusion into soil. However, the effect of sprinkler irrigation on N$_2$O emissions from fertiliser application as well as soil mineral N is unknown.

Efficient management of legumes in pasture may result in lower N fertiliser needs. However, with increasing rates of N fertiliser applications, legume N contribution is likely to decrease. Although the accompanying grass in the pasture theoretically should better use legume N, once the legume organic N is mineralised, the fate of mineral N is likely to be similar to that from fertiliser.

Extensive grazing

Generally, N$_2$O emissions from rangelands are about 0.2 kg N/ha.year. A number of options have been suggested to reduce the N$_2$O emissions from these areas. Williams (1994) suggested that increasing the efficiency of animal productivity per unit of N consumed might be the most economical way to reduce N$_2$O emissions from rangelands. These management options are likely to enhance the efficient resource management of rangelands as well. On the other hand, Howden *et al.* (1996) suggested that reducing sheep stock number and spring lamb production appears to be an option for reducing N$_2$O emissions from Australian grazing lands, although this approach is likely to reduce farm income by about 15% from reducing the stock number by 18% (Howden *et al.* 1996). However, these options could form an overall strategy for sustainable grazing systems.

Mitigating N$_2$O emissions from forestry systems

Nitrogen management for plantations and agroforestry systems follows similar principles to those of good N fertiliser management practices discussed above. That is, the rate of N fertiliser application and soil mineral N should meet the tree plantation N need and mineral N in soil should be kept low.

Land preparation for plantation and agroforestry should involve minimum soil disturbance or be restricted to a limited area to reduce organic N mineralisation. Barton *et al.* (1999) reported that improved drainage in forest soils reduced N losses from 40 kg N/ha.year to <1 kg N/ha.year. However, this may increase NO$_3^-$ losses in drainage waters. Understorey legumes could be utilised to provide N to trees and plantations, especially in the early years.

In summary, management practices for reducing N$_2$O emissions from arable cropping are aimed at the efficient use of N fertilisers by synchronising N supply and plant N demand. This may be achieved through regulating rate and time of fertiliser application, regulating mineral N release characteristics by using urease and nitrification inhibitors and

organic sources, and utilisation of residual mineral N by increasing crop intensity or using cover crops. These practices are also relevant to irrigated pastures and other intensively managed systems. Incorporating legumes into rotations, pastures, and the plantation understorey will reduce the need for N fertiliser applications and in the long-term may also reduce N_2O emissions on a national scale.

## Modelling of nitrous oxide emissions from soil

Since N_2O emissions from agroecosystems are sporadic, and temporally (diurnal, seasonal) and spatially variable, modelling of these systems is essential to provide an estimate of N_2O emissions over a number of seasons and areas for the National Greenhouse Gas Inventory.

Ecosystem models are necessary to simulate the N_2O emission from soil. These models must include the amount of N in soil and biomass, as well as the N fluxes (organic N, ammonium N, nitrate N), which contribute to the N_2O emissions from soil and biota. Also, ecosystem models are used to extend the results of spot measurement and short time experiments to regional and global estimates of N_2O emissions. As a first approximation, simple empirical models are used to estimate N_2O emissions from total nitrogen additions. For example, IPCC (1996) recommends a value of 1.25% of fertiliser N and animal manure N to estimate the total amount of N_2O emissions from these sources, whereas Taylor (1992) suggested net primary production as a basis of estimating N_2O emissions.

On the other hand, highly complex models, which explicitly simulate the biological, physical, and chemical processes governing N flows and fluxes in an ecosystem, are desirable for scenario simulations in terms of mitigation options but they require large data sets and measurements for parameterisation.

A large number of models have been developed to simulate N_2O fluxes from natural and managed ecosystems (Potter *et al.* 1996; Grant and Pattey 1999; Li *et al.*1992*a;* Parton *et al.* 2001). The more simplistic models were developed to simulate nitrogen gas fluxes (NO_x, N_2O, and N_2) at regional and global scales (Parton *et al.* 1996, Potter *et al.* 1996), by assuming that gaseous N loss is proportional to N mineralisation, with the proportion of N loss for each gas (N_2, N_2O, and NO_x) being a function of the soil WFPS (Fig. 3). The more process-oriented nitrogen gas flux models (Li *et al.* 1992*a*; Grant and Pattey 1999; Parton *et al.* 2001) were developed to simulate the effect of crop management practices and fertiliser N inputs on N gas fluxes. These process-oriented models simulate gas fluxes of N_2, N_2O, and NO_x from nitrification and denitrification and include detailed descriptions of the effect of soil temperature, NO_3^-, NH_4^+, water, and texture on N gas fluxes. The models use different approaches to simulate nitrification and denitrification. However, the theoretical basis of the models is similar and is based on extensive field and laboratory data (Firestone and Davidson 1989; Groffman 1991; Davidson and Schimel 1995). Some of the models simulate the dynamics of the microbes that are responsible for soil nitrification and denitrification, such as a process-based model, DNDC (Denitrification and Decomposition; Li *et al.* 1992*a*), whereas other models simulate nitrification and denitrification as a function of frequently measured and modelled variables such as soil water, temperature, NO_3^-, NH_4^+, and soil respiration, e.g. DAYCENT (Del Grosso *et al.* 2001*a*). Some of the highly mechanistic models simulate microbial growth rates, and solute and gas transport through the soil profile and aggregates (Smith 1980; Grant and Pattey 1999; Riley and Matson 2000).

A formal comparison of models that simulate N_2O gas fluxes was made by Frolking *et al.* (1998) and included a daily trace gas version of the CENTURY (a forerunner of DAYCENT), DNDC, CASA, and ExpertN models. The model results were compared with
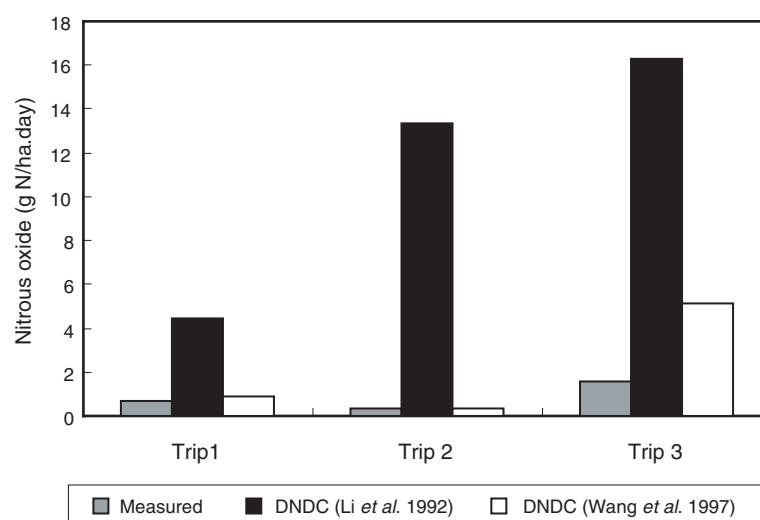
Fig. 5. Measured and simulated N$_2$O fluxes from a legume pasture in south-eastern Australia. DNDC model from Li *et al.* (1992*b*), and modified DNDC model by Wang *et al.* (1997) including changes in WFPS, temperature, organic C pools, and plant N uptake. (Adapted from Wang *et al.* 1997.)

observed daily N$_2$O gas fluxes from 4 different agroecosystem sites, which ranged from fertilised grasslands in Scotland to cropping systems in Germany and included sites with different soil textures and that were arid (Colorado) to subhumid (Scotland). The comparison demonstrated that models simulated the major differences in N$_2$O gas fluxes for the various sites fairly well. However, there were major differences in simulated N$_2$ and NO$_x$ gas fluxes (not measured at the sites used in the model comparison) among the models. The authors suggested that new model comparison efforts include sites where fluxes of N$_2$, NO$_x$, and N$_2$O are measured at the same time.

DNDC and DAYCENT (Li *et al.* 1992*a*; Del Grosso *et al.* 2001*a*, 2001*b*; Parton *et al.* 2001) have been used extensively to simulate the effect of management practice on N gas fluxes. Wang *et al.* (1997) simulated fluxes of CO$_2$ and N$_2$O by DNDC in a legume pasture system in south-eastern Australia in 1993. They had to modify DNDC by partitioning the total soil organic C into plant residue, microbial biomass, and humus using the constant proportions of 0.4:0.4:0.2 at the start of simulations, and the fraction of WFPS above which denitrification takes place from 0.4 in DNDC to 0.6 in the revised DNDC. Also, the soil surface temperature was used instead of the air temperature, since the former could be higher by as much as 20°C in summer. After these modifications and plant N uptake, they obtained a reasonable agreement between measured and simulated N$_2$O emission from a legume pasture (Fig. 5). This emphasises the need for verification and validation of imported models with parameterisation for Australian conditions.

Conceptually, the DAYCENT model is based on the assumption that the total gas emissions from soil are proportional to N cycling through the system and that soil gas diffusivity determines the relative amounts of the respective N gas species emitted from soil. The DAYCENT model includes submodels for soil organic matter decomposition, land surface parameters, plant productivity, and trace gas fluxes. The submodel for N$_2$O and N$_2$ fluxes from denitrification assumes that the N gas flux from denitrification is controlled
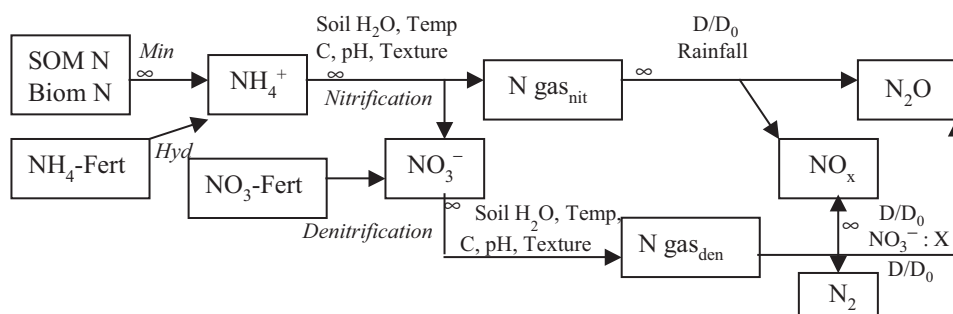
Fig. 6.   The N_2O, NO_x, and N_2 submodel of DAYCENT (adapted from Del Grosso *et al.* 2001*a*). The symbol ∞ indicates the process-controlling factor.
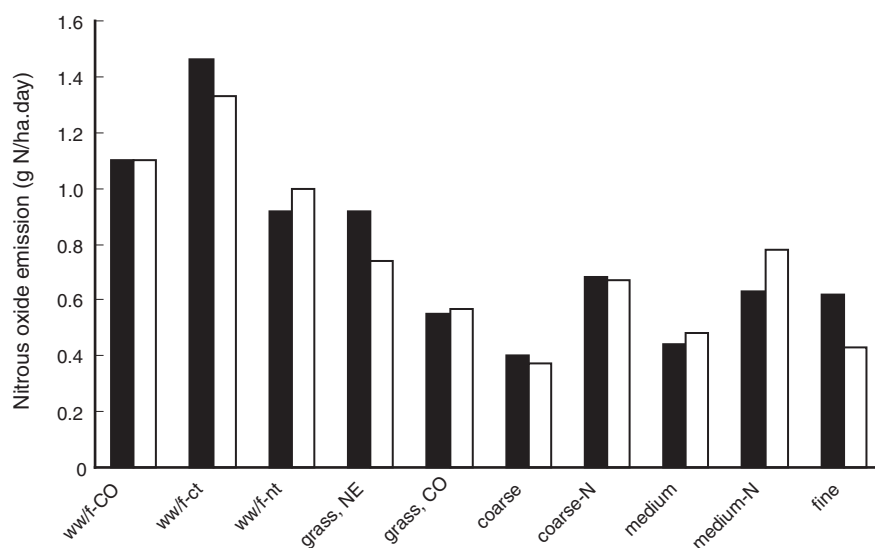
Fig. 7.   DAYCENT simulated (■) and observed (□) N_2O emissions from various farming systems, including conventional till (ct) and no-till (nt), soil textures (coarse, medium, and fine), and N application (10 kg N/ha) in Colorado, USA. (Source: Del Grosso *et al.* 2001*b*.)

by the most limiting factor among soil nitrate concentration, carbon substrate supply, and oxygen supply (Fig. 6).

The DAYCENT model simulated reasonably well the observed N_2O emissions from various farming systems, including conventional till and no-till, soil textures (coarse, medium, and fine), and low N application (10 kg N/ha) in an arid region (Fig. 7), similar to the moisture conditions in large parts of Australia. It should be noted that in a subhumid region in eastern Australia, N_2O emissions have been found to be higher under no-till and trash retention than under conventional till (Weier *et al.* 1998) due to high N applications, wetter regimes (greater WFPS), and a large soluble C source.

Although DAYCENT simulates the seasonal pattern of N_2O fluxes from grasslands reasonably well, it simulates these fluxes very poorly on a daily basis, and hence the significance of diurnal fluctuations in N_2O emissions due to changes in soil water and temperature. Also, soil mineral nitrogen appears to be consistently underestimated (Parton *et al.* 1998). Further improvements in simulation modelling are required, taking into
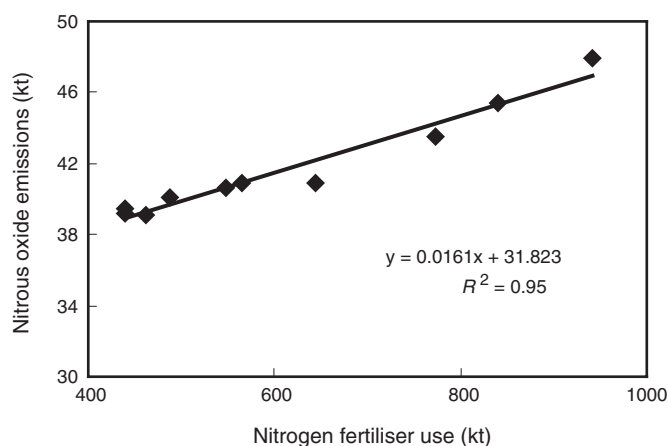
Fig. 8. Relationship between N$_2$O emissions (NGGI 1999) and nitrogen fertiliser used in Australian agriculture from 1990 to 1999 (McGuffog 2001).

account the soil texture differences, clay mineralogy, and integration of the 'hot spots' over time and field scale.

Summary and conclusions

There is a close relationship between the amount of nitrogen fertiliser used and the estimated N$_2$O emissions by NGGI (1999) (Fig. 8). The IPCC (1996) default emission factor is 1.25 kg of N$_2$O-N emitted for 100 kg of fertiliser N, while in Fig. 8, it appears to be 1.6 kg of N$_2$O-N per 100 kg of fertiliser N use.

Most N$_2$O emission measurements are of very short duration and over a limited area, and therefore do not measure the total N$_2$O emissions from either fertiliser or soils accurately. When N$_2$O emissions have been measured for over a year, about 2% of nitrogen fertiliser applied appears as N$_2$O in temperate environments (Cates and Keeney 1987; Mosier *et al.* 1996). No long-term N$_2$O emission measurements exist for the subtropical and tropical regions. Bouwman (1998) believes that the N$_2$O emission rates in these regions are likely to be higher than in the temperate regions. There is a need to measure these emissions for the Australian subtropical and tropical ecosystems.

Nitrous oxide emissions from legume-based pastures may be in the order of 1–2 kg N/ha, and similar amounts are estimated when these pastures are terminated for cropping. However, there is limited information on N$_2$O emissions during the fallowing period following the pasture phase. Similarly, few measurements are available to estimate the N$_2$O emissions from frequent fires in savannas. Since these areas are large, even modest N$_2$O emissions per unit area would have a large effect on the National Greenhouse Gas Inventory. Besides reducing the fire frequency of savannas, and ley pasture termination early in the season, limited management options are available to reduce N$_2$O emissions from these systems.

Management practices to improve nitrogen use efficiency and minimise N$_2$O emissions from nitrogen fertilisers include the following (Freney 1997*b*; Laegreid *et al.* 1999; McGuffog 2001).

(1) Apply fertiliser N at optimum rates by taking into account all N sources available to the crop/pasture from soil (ammonium and nitrate N in the soil at the time of crop sowing, and in-crop N mineralisation), and from other N sources such as manure or waste.

(2) Apply fertiliser N at the rate and time to meet crop/pasture needs and development stage, and when appropriate through split application.

(3) Avoid fertiliser N application outside the crop/pasture growing season, and especially prior to a clean fallow period. Avoid fallow periods if season or availability of irrigation permits.

(4) Provide fertiliser N application guide through crop/pasture monitoring and soil tests, and adjust fertiliser application rates and timing accordingly.

(5) Apply other nutrients if required so that nutrient supply to crop/pasture is balanced and N utilisation is optimised.

(6) Avoid surface application so that fertiliser N losses are minimised and plant utilisation maximised. Incorporate fertiliser N with soil; use band placement or point placement close to the plant roots.

(7) Monitor and adjust fertiliser application equipment to ensure the precision and amount of fertiliser applied, and control over appropriate spatial distribution (Global Positioning System/Geographical Information System) according to the information from yield monitors, crop/pasture monitors (including remote sensing), and soil tests.

(8) Fertiliser should be in a form (such as granulated) that can be applied evenly, conveniently, and cost-effectively. In irrigated agricultural systems, application in sprinkler/drip irrigation may be an effective option.

(9) Fertiliser may be formulated with urease and/or nitrification inhibitors or physical coatings to synchronise N release to that of crop/pasture growth needs so that, at any given time, a minimum amount of mineral N (ammonium and nitrate) is present in soil.

(10) Practice good crop/pasture management, disease control, and good soil management to optimise crop/pasture growth and hence efficient fertiliser N utilisation. Avoid or reduce cultivation early in the fallow period and retain plant residues to minimise mineralisation and nitrate accumulation during the fallow period.

(11) Use cover crops to utilise the residual mineral N following N-fertilised main crops or mineral N accumulation following legume-leys.

For manure management, the most effective practice is the early application and immediate incorporation of manure into soil to reduce the direct nitrous oxide emissions and secondary emissions from deposition of ammonia volatilised from manure and urine.

Perennial crop/pasture/tree systems mostly provide the optimum environment for reducing nitrous oxide emissions by using mineral nitrogen and water effectively to produce biomass. Management options for other agricultural systems should mimic the above to reduce nitrous oxide emissions from agricultural soils.

Prescribed burning of savannas primarily carried out in northern Australia continues to contribute to nitrous oxide and other oxides of nitrogen (NO, NO_2, etc.) emissions. Even light rainfall following burning increases nitrous oxide emissions. However, it is not known whether its elimination would reduce the annual amounts of nitrous oxide emissions, although the amounts of NO and NO_2 emissions immediately following the burning will be reduced.

Land clearing disturbs the soil and thus enhances nitrogen mineralisation. For the first few years after clearing, nitrous oxide emissions may be an order of magnitude greater than from undisturbed lands. Restriction on land clearing, therefore, will reduce nitrous oxide emissions, although the exact reductions are not known.

Emission factor uncertainties remain, since it is difficult to obtain definitive nitrous oxide flux values. Some of the reasons are: the techniques (high precision required for measuring atmospheric nitrous oxide concentration and the limitations of chamber and micrometeorogical methods), large spatial variability, the sporadic nature of nitrous oxide emissions, and seasonal and climate variability. Simulation modelling should complement these measurements over seasons and locations.

In addition to fertilisers, research is needed to evaluate various mitigation options:

● for cropping systems: (*i*) no-till systems, (*ii*) timing of plant residue incorporation, (*iii*) legume management practice, (*iv*) animal manure management and field applications, (*v*) nitrogen source (legume *v.* fertiliser nitrogen), (*vi*) crop combinations, (*vii*) crop/pasture mix and duration, (*viii*) salinity and sodicity effects, and (*ix*) soil type, season, and climate variability;

● for irrigated pastures: (*i*) optimum utilisation of pastures, thus ensuring plant biomass sink for nitrogen, (*ii*) hay or silage production when plant biomass is produced in excess, (*iii*) an optimum mix of grasses and legumes, (*iii*) animal waste management, (*iv*) soil type, season and climate variability;

● for extensive grazing lands: (*i*) optimum grazing management, cell or rotational grazing, and feed quality, (*ii*) regulating livestock numbers, (*iii*) fire frequency, (*iv*) soil type, season, and climate variability; and

● for forestry plantations: (*i*) soil disturbance, (*ii*) time of logging, (*iii*) legume *v.* fertiliser nitrogen, (*iv*) planting (species and techniques), (*iv*) salinity and sodicity.

In summary, a dedicated national research program is needed over a reasonable length of time combining chamber, mass balance, and micrometeorogical techniques using high precision analytical instruments, and covering a range of strategic activities in the agriculture sector, mentioned above.

We recognise that CO_2 emissions and sinks as well as CH_4 emissions and sinks interact strongly with N_2O emissions from soil. A comprehensive research program is needed to account for these interactions to arrive at cost-effective and efficient greenhouse gas mitigation management, policy, and legislation options.

Acknowledgments

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