Emission Sources of Nitrous Oxide from Australian Agricultural and Forest Lands and Mitigation Options

Ram Dalal, Weijin Wang, G. Philip Robertson, William J. Parton, C. Mike Myer and R. John Raison
The National Carbon Accounting System:

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• Reduces the scientific uncertainties that surround estimates of land based greenhouse gas emissions and sequestration in the Australian context.

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EMISSION SOURCES OF NITROUS OXIDE FROM AUSTRALIAN AGRICULTURAL AND FOREST LANDS AND MITIGATION OPTIONS

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#CSIRO Atmospheric Research
~CSIRO Forestry and Forest Products

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EXECUTIVE SUMMARY

Increases in the concentrations of greenhouse gases, carbon dioxide (CO₂), methane (CH₄), nitrous oxide (N₂O), and halocarbons in the atmosphere due to human activities are associated with global climate change. Carbon dioxide concentration in the atmosphere has increased by 30% and N₂O concentration has increased by 16% since 1750. The global mean surface temperature has increased by about 0.6°C in the last century. It is predicted to increase much faster this century if no action is taken to mitigate the emission rate of greenhouse gases.

Although atmospheric concentration of N₂O is much smaller (314 ppb in 1998) than carbon dioxide (365 ppm), its global warming potential (cumulative radiative forcing) is 296 times that of the latter in a 100-years time horizon. As a CO₂ equivalent (N₂O emission x 296), it contributes about 6% of the global warming effect due to all greenhouse gases.

MAGNITUDE OF NITROUS OXIDE EMISSIONS FROM AUSTRALIAN AGRICULTURE

Almost 80% of N₂O in the National Greenhouse Gas Inventory is produced by the agricultural sector; of this, 73% is emitted from agricultural soils and 24% from prescribed burning of savannas.

Australian sources of nitrous oxide (N₂O) emissions expressed in carbon dioxide (CO₂) equivalents and percent share of each sector in 1999.

<table>
<thead>
<tr>
<th>Source</th>
<th>CO₂ equivalent (Mt/yr)</th>
<th>CO₂ equivalent Percent share</th>
</tr>
</thead>
<tbody>
<tr>
<td>Agriculture</td>
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<td>Land Use Change and Forestry</td>
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<tr>
<td>Stationary Combustion</td>
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<td>3.4</td>
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<td>Transport and Fugitive</td>
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<tr>
<td>emissions from Fuel</td>
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<td>Industrial Plastics and Solvents, etc</td>
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<tr>
<td><strong>Total</strong></td>
<td><strong>28.31</strong></td>
<td><strong>100</strong></td>
</tr>
</tbody>
</table>

Estimates of N fertiliser use by crops and pastures in 2000 (Chudleigh and Simpson 2001).
Nitrous oxide emissions from agricultural soils come from nitrogen fertilisers (32%), soil disturbance (38%) and animal waste (30%). Since 1990, there has been a steady decline in pasture area, accompanied by decline in cattle and sheep numbers but increase in intensive livestock production. Therefore, \( \text{N}_2\text{O} \) emissions from soil disturbance and intensive livestock production have steadily increased. There has also been a fast increase in nitrogen fertiliser use for cereal production, accounting for 70% of the total fertiliser use in 2000 compared with just over 50% in 1990.

**NITROUS OXIDE EMISSIONS FROM AGRICULTURAL SOILS**

Nitrous oxide is primarily produced in soil by the activities of micro-organisms during nitrification of ammonium into nitrate, and denitrification of nitrate (and nitrite) into nitrogen gas.

The lack of oxygen or limited oxygen supply in soil to the active micro-organisms is the primary cause of \( \text{N}_2\text{O} \) production during denitrification. When oxygen supply is low due to slow oxygen diffusion through water-filled pores in saturated or waterlogged soils or due to high oxygen demand created by abundant carbon food source, micro-organisms utilise nitrite and nitrate in place of oxygen. Thus, nitrate and nitrite are denitrified to nitric oxide (NO), nitrous oxide, and nitrogen (\( \text{N}_2 \)) gases.

The proportion of \( \text{N}_2\text{O} \) to \( \text{N}_2 \) gas in the denitrification process depends on:

- nitrate and nitrite supply (generally higher \( \text{N}_2\text{O}/\text{N}_2 \) ratio at higher \( \text{NO}_2^- \) or \( \text{NO}_3^- \) concentrations);
- oxygen supply or water-filled pore space (higher \( \text{N}_2\text{O}/\text{N}_2 \) ratio at <60-80% water-filled pore space, but at saturation or above, it is almost exclusively \( \text{N}_2 \) gas emissions);
- temperature (\( \text{N}_2\text{O} \) production rate increases but the \( \text{N}_2\text{O}/\text{N}_2 \) ratio decreases with increasing temperature);
- pH and salinity (even very acidic soils, especially pH <4.5, also produce \( \text{N}_2\text{O} \)); and
- carbon substrate supply (moderate supply of carbon substrate increases \( \text{N}_2\text{O} \) production but high carbon substrate may lead to \( \text{N}_2 \) production due to anaerobic conditions).

Since these variables in agricultural systems change over space and time at different rates, \( \text{N}_2\text{O} \) production from soil is sporadic both in time and space. Therefore, it is a challenge to measure \( \text{N}_2\text{O} \) emissions from a given location and then scale up to regional and national scales.

Estimates of \( \text{N}_2\text{O} \) emissions from various agricultural systems vary widely. For example, in flooded rice in the Riverina Plains, \( \text{N}_2\text{O} \) emissions ranged from 0.02% to 1.4% of fertiliser N applied, whereas in irrigated sugarcane crops, 15.4% of fertiliser applied at 160 kg N/ha was lost over a 4-day period (12 t/ha CO\(_2\) equivalent). Nitrous oxide emissions from the fertilised dairy pasture soils in Victoria range from 6 to 11 kg \( \text{N}_2\text{O}-\text{N} \)/ha (2.9-5.3 t/ha CO\(_2\) equivalent) when fertiliser is applied at rates from 0 to 200 kg N/ha.

In arable cereal cropping, estimates of \( \text{N}_2\text{O} \) 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 almost unknown but probably exceed that from fertilisers due to frequent wetting and drying cycles over a longer period. In ley-cropping system, significant \( \text{N}_2\text{O} \) losses could occur, due to the accumulation of 70-150 kg nitrate-N/ha, from legume-based pastures. However, limited data exist to quantify \( \text{N}_2\text{O} \) losses in ley cropping system in Australia.

Extensive grazed pastures and rangelands contribute annually about 0.2 kg N/ha as \( \text{N}_2\text{O} \) (93 kg/ha per year CO\(_2\) equivalent). Unfertilised forestry systems may emit less but the fertilised plantations may emit more \( \text{N}_2\text{O} \) than the extensive grazed pastures. Australian data is sparse for the forestry systems.
Burning of biomass such as forest vegetation, savannah grass, and agricultural residues is an important source of nitrous oxide emissions. In 1999, about 24% of the total annual emissions of nitrous oxide from the agricultural sector were the result of direct emissions from prescribed burning of savannas. While combustion is clearly a major source of nitrous oxide emissions, the effect of fires extends beyond the direct contribution of emissions during combustion. Fire also influences processes of N₂O production within the soil by heating it, causing natural pyrolysis or thermal degradation to both the labile and intractable soil materials.

Overall, there is a need to examine the emission factors used in estimating emissions, for example 1.25% of fertiliser or animal excreted nitrogen appearing as N₂O (IPCC 1996). The impacts of an appropriate emission factor are at least four-fold: improve N₂O emissions estimates in the National Greenhouse Gas Inventory; identify the agricultural sectors that inefficiently utilise or dispose of N products (fertilisers, soil mineralisation, animal waste and other products); examine mitigation options to reduce N₂O emissions; and evaluate code of practice, and policy/legislation that encourages mitigation of N₂O emissions from agriculture.

**MITIGATION OF NITROUS OXIDE EMISSIONS**

The primary consideration for mitigating N₂O emissions from the agricultural sector is to match the supply of mineral N (ammonium, nitrite and nitrate) and keep it to a minimum commensurate with its spatial and temporal needs of crops/pastures/trees.

Management practices to minimise N₂O emissions from N fertilisers and legumes as well as improve N use by crop/pasture or reduce application rates include:

1. Apply fertiliser N at optimum rates by taking into account all N sources available to the crop/pasture from soil, and other N sources such as legume, manure or waste.

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

3. Avoid fertiliser N application outside the crop/pasture-growing season. Avoid fallow periods if season or availability of irrigation permits.

4. Provide fertiliser N application guide through crop/pasture monitoring, yield maps and soil tests.

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

6. Avoid surface application, incorporate or band place so that fertiliser N losses are minimised and plant utilisation maximised.

7. Monitor and adjust fertiliser application equipment to ensure the precision and the amount of fertiliser applied.


9. Fertilizer 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.

10. Fertilizer may be formulated with urease and/or nitrification inhibitors or physical coatings to mimic fertiliser N release to crop/pasture growth needs.

11. Practice good crop/pasture management, disease control and soil management to optimise crop/pasture growth.
12. Use non-legume 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 N$_2$O emissions and secondary emissions from deposition of ammonia volatilised from manure and urine.

Secondary considerations to reduce N$_2$O emissions include:

- Oxygen supply/soil water content (water-filled pore space, <40% increases nitrification but reduces N$_2$O loss, >90% increases N$_2$ loss); examples include arable cropping/pasture/forestry, and flooded rice. Improve oxygen diffusion in soil by eliminating the compacted layer.

- Carbon-substrate supply (readily available carbon creates ‘hot spots’ of microbial growth, and hence increases in N$_2$O emissions); restrict readily available supply - examples are addition or incorporation of biomass of high carbon:nitrogen ratio such as non-legumes rather than legume biomass.

- Soil organic matter management to manipulate carbon substrate and oxygen/water supply.

- Soil pH and salinity (salinity and high pH enhance the N$_2$O emissions due to the persistence of nitrite); soil amendments such as application of gypsum or crop residues of high carbon: nitrogen ratio reduces N$_2$O emissions.

- Eliminate limitations of other nutrients such as phosphorus, potassium or zinc.

Perennial crop/pasture/tree systems mostly provide optimum environment to reduce N$_2$O emissions by using mineral N and water effectively to produce biomass. Management options for other agricultural systems should mimic the above to reduce N$_2$O emissions from the agricultural soils.

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. Besides, ley pasture O$_2$ emissions during the fallowing period following the pasture phase.

Field burning of agricultural residues and the prescribed burning of tropical savanna and temperate grasslands contribute significantly to national emissions of N$_2$O. Direct emissions of N$_2$O into the atmosphere from field burning of agricultural residues are being reduced by the use of alternative management practices such as stubble retention and green cane harvesting. Minimising emissions from prescribed burning practices associated with pasture management will require practical management and monitoring protocols specifically designed to provide opportunities for local landholders to develop the expertise required to implement more appropriate pasture management to eliminate or reduce burning practices.

Land clearing disturbs the soil and thus enhances N mineralisation and N$_2$O emissions at least in the first few years. Restriction on land clearing, therefore, will reduce N$_2$O emissions although the exact reductions are not known.

Current models such as DNDC and DAYCENT can be used to simulate N$_2$O production from soil after parameterisation with the local data, appropriate modification and verification against the measured N$_2$O emissions under different management practices and integrated spatially and temporally.
POLICY CONSIDERATIONS AND RESEARCH OPPORTUNITIES

Nitrogen fertiliser use is likely to increase for crop/pasture production, since soil N supply is declining as improved pasture areas and soil fertility decrease, along with greater demand for product quality and supply.

Education is critical for the efficient use of fertilisers in crop/pasture production to reduce fertiliser waste as well as to reduce N\textsubscript{2}O emissions. Fertilizer Industry Federation of Australia (FIFA) should play a pivotal role in partnership with the relevant industry bodies in preparing Codes of Practice for Efficient Fertiliser Use (a current FIFA initiative: Guidelines for Developing Nutrient Management Codes of Practice).

Although, both State and Federal legislation exists to ensure quality assurance on fertiliser quality, policy consideration should be given to discourage fertiliser applications out of season, and in excess of crop demands. This will also require the elimination of price incentives by suppliers for out of crop season applications.

Incentives may be offered to producers of machinery and equipment for precise applications of fertilisers, when (at or near sowing or in-crop) and where (prescribed area in a paddock) it is needed. There is a limited scope to encourage the use of slow-release fertilisers to mimic plant needs due to higher costs of the amended fertilisers, which are rarely recovered by increased yields. Recently, it has been claimed that the application of 3,4-dimethylpyrazole phosphate with urea, reduced N\textsubscript{2}O emissions by 45% over a 3-year period in the field in Germany. There is a need to examine its effectiveness over a number of seasons, crop/pastures, and soils under warmer Australian conditions where other products have been ineffective.

There are a number of other areas of research such as N\textsubscript{2}O emissions estimates in ley cropping and intensive livestock systems (the emission factors, the activity data, coverage of measurements, spatial integration, and information on specific on-farm practices), as well as mitigation options for reducing N\textsubscript{2}O emissions from agriculture.

Emission factor uncertainties remain since it is difficult to obtain definitive N\textsubscript{2}O flux values. Some of the reasons are: the techniques - high precision required for measuring atmospheric N\textsubscript{2}O concentration and the limitations of chamber and micrometeorological methods, large spatial variability, the sporadic nature of N\textsubscript{2}O emissions, and seasonal and climate variability.

In addition to fertilisers, research is needed in evaluating 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 versus fertiliser nitrogen, (vi) crop combinations, (vii) crop/pasture mix and duration, (viii) salinity, 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, (iv) animal waste management, (v) 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 versus fertiliser nitrogen, (iv) planting- species and techniques.
In summary, a directed national research program is needed for a considerable duration, to cover sampling season and climate, and combined with chamber, mass balance and micrometeorological techniques using high precision analytical instruments, simulation modelling, covering a range of strategic activities in the agriculture sector.

Although it is not considered in this review, we recognise that since CO₂ emissions and sinks as well as CH₄ emissions and sinks interact strongly with N₂O emissions from soil, a comprehensive research program is needed to account for these interactions and to arrive at cost-effective and efficient greenhouse gas mitigation management, policy and legislation options.

ACKNOWLEDGMENTS

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Aerobic
Molecular oxygen being available for microbial growth, respiration or decomposition.

Ammonia volatilisation
Mass transfer of nitrogen as ammonia gas from soil to the atmosphere.

Aerobic
Molecular oxygen being available for microbial growth, respiration or decomposition.

Carbon dioxide
CO₂, the principal anthropogenic greenhouse gas, contributing 60% of the enhanced greenhouse effect.

Carbon dioxide equivalent
Radiative activity and atmospheric residence time of a greenhouse gas compared with carbon dioxide (global warming potential). Carbon dioxide equivalent for nitrous oxide: multiply molecular N as N₂O by 1.57, and then multiply by 296 (global warming potential) to obtain carbon dioxide equivalent (1 kg N₂O-N = 465 kg CO₂-equivalent).

Denitrification
Reduction of nitrogen oxides (usually nitrite and nitrate) to nitrogen oxides (NOₓ) with a lower oxidation state of nitrogen or molecular nitrogen (N₂) by microbial activity (denitrification) or by chemical reactions involving nitrite and reduction of nitrogen oxides by combustion (chemodenitrification). Nitrogen oxides are used by microorganisms as terminal electron acceptors in place of oxygen in anaerobic or low-oxygen respiratory metabolism.

Electron acceptor
A compound which accepts electrons during biotic or abiotic chemical reactions and is thereby reduced.

Field capacity
The content of water, on mass or volume basis, that remains in a soil 48-72 hours after full wetting when free drainage is negligible.

Global warming potential
Relative warming effect of a unit mass of the gas compared with the same mass of carbon dioxide over a specific period (see also carbon dioxide equivalent).

Greenhouse effect
The absorption of infrared heat waves radiated back from the earth surface by the air, principally by carbon dioxide, methane and nitrous oxide, and water vapour, thus trapping heat by the atmosphere much as does the glass in a greenhouse.

Methane
CH₄; contribute about 20% to global warming.

Nitrogen mineralisation
The conversion of an organic form of nitrogen to an inorganic form as a result of microbial activity.

Nitrate reduction (biological)
The biological process whereby nitrate is reduced by microorganisms to ammonium for cell synthesis (assimilatory nitrate reduction) or to nitrite by microorganisms using nitrate as the terminal electron acceptor in anaerobic respiration (dissimilatory nitrate reduction) (also see denitrification).

Nitrification
Biological oxidation of ammonium to nitrite and nitrate, or a biologically induced increase in the oxidation state of nitrogen.

Nitrogen fixation
Conversion of molecular nitrogen (N₂) to ammonia and subsequently to organic nitrogen.

NOₓ gases
Oxygen state of nitrogen (nitrogen oxides) in gaseous form; major NOₓ gases are nitrous oxide (N₂O), nitric oxide (NO), and nitrogen dioxide (NO₂).
**Nitrous oxide conversion**

$N_2O$ contributes 6% to global warming.

$1 \text{ g } N_2O-N/\text{ha}/\text{day} = 0.365 \text{ kg } N_2O-N/\text{ha}/\text{year}$

$= 0.574 \text{ kg } N_2O/\text{ha}/\text{year}$

$= 170 \text{ kg } CO_2 \text{ equivalent/ha/year}.$

**Nitrous oxide sinks**

The stratosphere is the known nitrous oxide sink. Nitrous oxide in the stratosphere reacts with photons and produces free oxygen radicals, which react with ozone to convert $N_2O$ into nitric oxide (NO) and nitrogen ($N_2$). This process depletes ozone in the stratosphere.

**Nitrous oxide sources**

The primary known source is from soil, primarily produced by microbial activity (see denitrification).

**ppb nitrous oxide**

Parts per billion or $1000^{th}$ of ppm;

$314 \text{ ppb } N_2O = 0.358 \text{ mg } N_2O-N/\text{m}^3.$

**Water-filled pore space (WFPS)**

The fraction of total porosity (total volume not occupied by solids) of soil that is filled with water. WFPS >100% in flooded soils.
# TABLE OF CONTENTS

<table>
<thead>
<tr>
<th>Section</th>
<th>Page No.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Executive Summary</td>
<td>iii</td>
</tr>
<tr>
<td>Acknowledgements</td>
<td>viii</td>
</tr>
<tr>
<td>Glossary of Terms</td>
<td>ix</td>
</tr>
<tr>
<td>1. Introduction</td>
<td>1</td>
</tr>
<tr>
<td>1.1 Sources and Sinks of Nitrous Oxide</td>
<td></td>
</tr>
<tr>
<td>2. N₂O Production in Soil</td>
<td>5</td>
</tr>
<tr>
<td>2.1 Processes Involved in N₂O Production in Soil</td>
<td>5</td>
</tr>
<tr>
<td>2.2 Factors Controlling N₂O Production in Soil</td>
<td>5</td>
</tr>
<tr>
<td>2.2.1 Moisture and Aeration</td>
<td>5</td>
</tr>
<tr>
<td>2.2.2 Temperature</td>
<td>6</td>
</tr>
<tr>
<td>2.2.3 Soluble and Readily Decomposable Carbon</td>
<td>7</td>
</tr>
<tr>
<td>2.2.4 Soil and Fertilizer Nitrogen</td>
<td>8</td>
</tr>
<tr>
<td>2.2.5 Soil pH and Salinity</td>
<td>9</td>
</tr>
<tr>
<td>2.2.6 Limitation of Nutrients Other Than Nitrogen</td>
<td>9</td>
</tr>
<tr>
<td>2.3 Methods and Their Limitations in Measuring N₂O Emissions from Soil</td>
<td>10</td>
</tr>
<tr>
<td>2.3.1 Chamber Methods</td>
<td>10</td>
</tr>
<tr>
<td>2.3.2 Micrometeorological Methods</td>
<td>13</td>
</tr>
<tr>
<td>2.3.3 Methods for Measuring N₂O Emissions During Fire</td>
<td>15</td>
</tr>
<tr>
<td>3. Modelling of Nitrous Oxide Emissions from Soil</td>
<td>16</td>
</tr>
<tr>
<td>4. N₂O Emissions from Various Ecosystems</td>
<td>19</td>
</tr>
<tr>
<td>4.1 N₂O Emissions from Agricultural Systems</td>
<td>19</td>
</tr>
<tr>
<td>4.1.1 Cropping Soils</td>
<td>19</td>
</tr>
<tr>
<td>4.1.2 Horticultural Soils</td>
<td>24</td>
</tr>
<tr>
<td>4.2 N₂O Emissions from Grazing Systems</td>
<td>24</td>
</tr>
<tr>
<td>4.2.1 Intensive Grazing</td>
<td>24</td>
</tr>
<tr>
<td>4.2.2 Extensive Grazing (improved pastures vs rangelands)</td>
<td>25</td>
</tr>
<tr>
<td>4.3 N₂O Emissions from Forestry Systems</td>
<td>26</td>
</tr>
<tr>
<td>4.4 N₂O Emissions from Other Systems</td>
<td>28</td>
</tr>
<tr>
<td>5. Mitigating N₂O Emissions from Various Ecosystems</td>
<td>28</td>
</tr>
<tr>
<td>5.1 Mitigating N₂O Emissions from Agricultural Systems</td>
<td>28</td>
</tr>
<tr>
<td>5.2 Mitigating N₂O Emissions from Grazing Systems</td>
<td>29</td>
</tr>
<tr>
<td>5.3 Mitigating N₂O Emissions from Forestry Systems</td>
<td>30</td>
</tr>
<tr>
<td>6. Summary and Conclusions</td>
<td>31</td>
</tr>
<tr>
<td>6.1 Policy Considerations and Research Opportunities</td>
<td>33</td>
</tr>
<tr>
<td>7. References</td>
<td>35</td>
</tr>
</tbody>
</table>
LIST OF TABLES

Table 1. Estimates of global warming potentials (GWP) of carbon dioxide, methane and nitrous oxide. 1
Table 2. Australian sources of nitrous oxide in 1999. 2
Table 3. Nitrous oxide emissions from Australian agriculture in 1999. 3
Table 4. Nitrous oxide emissions from agricultural soils in 1999. 4
Table 5. Estimates of nitrogen use by different crops. 4
Table 6. Comparison between the estimated enhanced emissions due to soil disturbance by in situ biomass burning and directly from smoke plume emissions. 7
Table 8. Estimates of mineral N (mostly nitrate-N) produced during 3-8 months following the termination of the pasture phase and calculated nitrous oxide emissions due to denitrification. 23

LIST OF FIGURES

Figure 1. Trends in N2O-N emissions from Australia from 1990 to 1999: (■) all sources and (♦) the agricultural lands (Australian Greenhouse Office 2001; Dalal et al. 2003). 2
Figure 2. Trends in nitrogen fertiliser use in the Australian agriculture (data from Chudleigh and Simpson 2001; Dalal et al. 2003). 3
Figure 3. A generalised relationship between water-filled pore space (WFPS) of soils and the relative fluxes of N2O (▲) and N2 (■) from nitrification and denitrification. N2O fluxes dominate between 50% and 80% of WFPS and N2 dominates above 80% WFPS (From Dalal et al. 2003). 6
Figure 4. A diagrammatic representation of the effect of the concentration of nitrate-N in soil on the relative N2O (▲) and N2 (■), and total denitrification losses (●) (redrawn from Mosier et al. 1983; Dalal et al. 2003). 8
Figure 5. Probability distribution for chamber-based N2O flux measurements for a conventionally managed cropping system 1991-1999 (Robertson et al. 2000). Distributions tend to be similarly skewed in even low-flux systems. 10
Figure 6. N2O and CO2 fluxes from an automated chamber located in a conventionally managed cropping system. Note the order of magnitude difference in N2O flux over the course of day 182-183 (Phillip Robertson, unpublished data). 11
Figure 7. CO2 emission (soil + plant respiration) and N2O emission from lucerne. Note the order of magnitude difference in N2O flux on 23 October following rainfall. (Source: Meyer et al. 2001). 12
Figure 8. Denitrification fluxes across 0.5 ha of a Michigan old field. Average rates vary from 18 to >90 µg m⁻² d⁻¹ in the light vs. dark areas, respectively. (From Robertson et al. (1988)). 13
Figure 9. Measured and simulated N2O fluxes from a legume pasture in south-east Australia. (DNDC model from Li et al. (1992b), 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; Dalal et al. 2003)). 17
Figure 10. The N2O, NOx and N2 submodel of DAYCENT (adapted from Del Grosso et al. 2001b; Dalal et al. 2003). 18
LIST OF FIGURES continued

Figure 11. DAYCENT simulated (red bar) and observed (blue bar) N$_2$O emissions from various farming systems, including conventional till (ct) and no-till (nt), soil textures (coarse, medium and fine), N application (10 kg N/ha), intensively cropped rye, crop rotations (G1, multiple cropping with FYM application and G2, multiple cropping with cattle slurry application) and irrigated barley and maize (Source: Del Grosso et al. 2001b). 20

Figure 12. Relationship between nitrous oxide emissions and nitrogen fertiliser used in Australian agriculture from 1990 to 1999 (from Dalal et al. 2003). 31

Figure 13. A schematic diagram of nitrogen uptake pattern for a cereal crop. 33
1. INTRODUCTION

Over the 20th century, the global mean surface temperature has increased by about 0.6 ± 0.2°C (IPCC, 2001). This warming effect has primarily resulted from the increase in the concentrations of greenhouse gases (CO₂, CH₄, N₂O, and halocarbons) in the atmosphere due to human activities. For example, the atmospheric concentration of CO₂ has increased from 280 ppm in 1750 to 365 ppm in 1998; and N₂O has increased from 270 ppb to 314 ppb during this period (IPCC, 2001). These gases absorb light in the infrared region, therefore trap thermal radiation emitted from the earth’s 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 of N₂O is 296 times that of CO₂ over a 100-year time horizon (Table 1); and in spite of its lower rate of concentration rise (0.8 ppb/year from 1990 to 1999) than that of CO₂ (1.5 ppm/year), N₂O contributes approximately 6% of the warming effect caused by the increase of the various greenhouse gases.

| Table 1. Estimates of global warming potentials (GWP) of carbon dioxide, methane and nitrous oxide. |
|---|---|
| Gas  | GWP |
| CO₂  | 1   |
| CH₄  | 23  |
| N₂O  | 296 |


1.1. SOURCES AND SINKS OF NITROUS OXIDE

Land and oceans are the principal natural sources of N₂O emissions, with soils contributing about 65% and oceans about 30% (IPCC, 2001). N₂O is very stable in the atmosphere, with a lifetime of about 114 yrs (IPCC, 2001). Probably the only significant process that removes N₂O is its reaction in the stratosphere with excited oxygen atoms formed by photolysis of ozone (Crutzen, 1981). Microorganisms in soil can reduce N₂O into N₂ under anaerobic conditions (Freney et al., 1978; Ryden, 1981). However, the significance of soil as a sink for N₂O remains uncertain and probably very small (Freney et al. 1978).

The Australian National Greenhouse Gas Inventory (NGGI) includes CO₂, CH₄ and N₂O gases. Annual emissions of N₂O exceeded 60 kilo tonnes of N (94 kt of N₂O) in 1999 (Figure 1), and nearly 80% was contributed by the Australian agriculture (Australian Greenhouse Office 2001). Considering the Global Warming Potential (GWP) of N₂O as 296 compared to CO₂ (Table 1), the CO₂-equivalent emissions from N₂O in 1999 were 28.31 Mt CO₂-e (Table 2), (22.27 Mt CO₂-e from the agricultural sector), which were about 6% of the total CO₂-e emissions from Australia. The rate of N₂O emissions increased by 21% from 1990 to 1999; however, a much steeper increase was recorded in the last 3 years than in the 1990-1996 period (Figure 1) (Australian Greenhouse Office 2001). Thus, the rate of increase for N₂O 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 N₂O emissions from Australian agriculture.
Table 2. Australian sources of nitrous oxide in 1999.

<table>
<thead>
<tr>
<th>Source</th>
<th>$N_2O$ kt N/yr</th>
<th>CO$_2$ equivalent Mt/yr</th>
</tr>
</thead>
<tbody>
<tr>
<td>Agriculture</td>
<td>47.92</td>
<td>22.27 (78.7%)$^B$</td>
</tr>
<tr>
<td>Land Use Change and Forestry</td>
<td>1.43</td>
<td>0.66 (2.3%)</td>
</tr>
<tr>
<td>Stationary Combustion</td>
<td>2.09</td>
<td>0.97 (3.4%)</td>
</tr>
<tr>
<td>Transport and Fugitive</td>
<td>8.09</td>
<td>3.76 (13.4%)</td>
</tr>
<tr>
<td>emissions from Fuel</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Industrial Plastics and</td>
<td>1.39</td>
<td>0.65 (2.2%)</td>
</tr>
<tr>
<td>Solvents, etc</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total</td>
<td>60.92</td>
<td>28.31 (100%)</td>
</tr>
</tbody>
</table>


$^B$ Percentage of the total $N_2O$ emissions from different Australian sectors.

Of the total emissions of $N_2O$ as CO$_2$-e of 28.31 Mt in 1999 (Table 2), almost 22.3 Mt CO$_2$-e was emitted from agricultural lands (16.17 Mt CO$_2$-e) and fire related activities (5.48 Mt CO$_2$-e prescribed woodland burning and 0.6 Mt CO$_2$-e from tree clearing) (Table 3) (Australian Greenhouse Office 2001). Soil and land-based activities, therefore, account for the main sources of $N_2O$ (about 80%) in Australia. However, large uncertainties exist in these estimates (Table 3). Uncertainties in estimates of direct emissions of $N_2O$ 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 $N_2O$ 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 4), and grasslands and savanna burning (Table 3).
Nitrogen fertiliser use for agricultural production has increased sharply in recent years (Figure 2). Nitrogen fertiliser use for cereal crops increased by 314% from 1987 to 1996 and it further increased by 11% from 1996 to 2000 (Table 5). Although much smaller in amount, N fertiliser use for oilseed crops such as canola increased by 390% from 1996 to 2000. Since 1996, cereal/oilseed crops accounted for more than 70% of the total N fertiliser use. For comparison, just over 50% of the total N fertilisers was used in the agricultural sector ten 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 6-7% each by 1996-2000. Overall, almost 90% increase in N₂O emissions from 1990 to 1999 was due to an increase in the rate of N fertiliser application (Australian Greenhouse Office 2001).

![Figure 2. Trends in nitrogen fertiliser use in the Australian agriculture](image)

Table 3. Nitrous oxide emissions from Australian agriculture in 1999.

<table>
<thead>
<tr>
<th>Source</th>
<th>N₂O kt N/yr</th>
<th>CO₂ equivalent Mt/yr</th>
<th>Uncertainty 95% confidence limit (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Agricultural Soils</td>
<td>34.78</td>
<td>16.17 (72.6%)</td>
<td>-56 to +120</td>
</tr>
<tr>
<td>Prescribed Burning of Savannas</td>
<td>11.80</td>
<td>5.48 (24.6%)</td>
<td>-68 to +129</td>
</tr>
<tr>
<td>Crop Residue Burning</td>
<td>0.17</td>
<td>0.08 (0.3%)</td>
<td>-39 to +50</td>
</tr>
<tr>
<td>Manure Management</td>
<td>1.17</td>
<td>0.54 (2.5%)</td>
<td>-7.6 to 22.8</td>
</tr>
<tr>
<td>Total</td>
<td>47.92</td>
<td>22.27 (100.0%)</td>
<td></td>
</tr>
</tbody>
</table>

B Percentage of the total N₂O emissions from different Australian activities.
Table 4. Nitrous oxide emissions from agricultural soils in 1999.

<table>
<thead>
<tr>
<th>Source</th>
<th>( \text{N}_2\text{O} )</th>
<th>( \text{CO}_2 ) equivalent</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>kt N/yr</td>
<td>Mt/yr</td>
</tr>
<tr>
<td>Fertilizer Use</td>
<td>11.14</td>
<td>5.18 (32.1%)</td>
</tr>
<tr>
<td>Crop/Pasture Soil Disturbance</td>
<td>13.38</td>
<td>6.22 (38.4%)</td>
</tr>
<tr>
<td>Animal Faeces and Urine</td>
<td>10.26</td>
<td>4.77 (29.5%)</td>
</tr>
<tr>
<td>Total</td>
<td>34.78</td>
<td>16.17 (100.0%)</td>
</tr>
</tbody>
</table>

*Australian Greenhouse Office (2001; Dalal *et al.* 2003).*

*Percentage of the total \( \text{N}_2\text{O} \) emissions from different Australian activities.*

There are numerous reasons given for the increase in N fertiliser use for cereals/oilseed crops in recent years (Table 5). Some of these reasons are: (i) decrease in pasture area by 6 million ha in the last 10 years, hence lower legume \( \text{N}_2\)-fixed supply of N to the following 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 impact of technological changes on crop yields and N demand (Angus 2001).

Table 5. Estimates of nitrogen use by different crops.

<table>
<thead>
<tr>
<th>Crop</th>
<th>Fertilizer N use (kt N)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>1987*</td>
</tr>
<tr>
<td>Cereals</td>
<td>195</td>
</tr>
<tr>
<td>Sugarcane</td>
<td>58</td>
</tr>
<tr>
<td>Cotton</td>
<td>27</td>
</tr>
<tr>
<td>Oilseed/Pulses</td>
<td>4</td>
</tr>
<tr>
<td>Horticulture</td>
<td>43</td>
</tr>
<tr>
<td>Pasture</td>
<td>44</td>
</tr>
<tr>
<td>Total</td>
<td>371</td>
</tr>
</tbody>
</table>

*Bellingham (1989).*

*Reuter (2001).*

*Chudleigh and Simpson (2001).*

*Dalal *et al.* 2003.*

Nitrous oxide emissions from soil disturbance (cultivation, agricultural activities, ley/cropping) showed a slight decline of 2.1% from 1990 to 1999 due to decrease in ley/cropping area, shifting to continuous cropping.

Animal faeces and urine account for an increase of 90% in the \( \text{N}_2\text{O} \) 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).

The burning of biomass such as forest vegetation, savanna grasslands, and agricultural residues is an important source of nitrous oxide emissions. 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 by recycling of P and K, which are generally low in the region’s soils although pastures get further depleted of N after every fire.

It was estimated in 1999 that 24% of the total annual emissions of nitrous oxide from the agricultural sector were the result of direct emissions from prescribed burning of savannas (Australian Greenhouse Office 2001). While combustion is clearly a major source of nitrous oxide emissions, the effect of fires extends beyond the direct contribution of emissions during combustion. Fires also cause substantial changes to the topsoil resulting in significant and persistent changes in the soil-to-atmosphere fluxes of nitrous oxide (Meyer *et al.* 1997), due to changes in soil N mineralisation (Raison 1979). Emissions of \( \text{N}_2\text{O} \) directly from savanna and grassland fires in 1999 were estimated to be 11.8 kt, while the estimated enhanced emissions due to soil disturbance by *in situ* biomass burning were in the range of 0.6-12.1 kt (NGGIC 1999). The estimates of \( \text{N}_2\text{O} \) emissions from 1990 to 1999 showed an increase of 38% from savanna fires although there is a large uncertainty in the estimates of total area affected by fires (Australian Greenhouse Office 2001). Such estimates highlight the significant contribution made by emissions due to soil N disturbance by *in situ* biomass burning.
2. N₂O PRODUCTION IN SOIL

2.1. PROCESSES INVOLVED IN N₂O PRODUCTION IN SOIL

Nitrous oxide is produced in soil by at least three microbial-mediated mechanisms: (i) during ammonium oxidation to nitrite (nitrification) (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 N₂O in soil (Tiedje 1994), although N₂O production by nitrification may sometimes be equally important (Granli and Bøckman 1994). The generalised Equation 1 below shows the first two mechanisms of nitrous oxide production in soil (Dalal et al. 2003).

The fourth mechanism implicated in N₂O production is the abiotic nitrite and nitrate reduction (involving organic matter and reductants such as ferrous iron) (Magalhaes and Chalk 1989) although its process mechanism is unclear and its overall contribution is uncertain (Venterea and Rolston 2000). It is favoured under acidic conditions and may occur under both aerobic and anaerobic conditions (Chalk and Smith 1983).

2.2 FACTORS CONTROLLING N₂O PRODUCTION IN SOIL

2.2.1 Moisture and Aeration

Nitrification increases with increasing water content up to 50 to 60% of water-filled pore space (WFPS) (Figure 3), provided it is not limited by other factors such as NH₄⁺ supply (Linn and Doran, 1984). The rate of N₂O production from nitrification is normally low below 40% WFPS, but increases rapidly with increasing water content up to 55–65% WFPS (Goodroad and Keeney, 1984). When soil moisture exceeds 60–70% WFPS (depending on other factors e.g. temperature and decomposable C), increase in water content hinders aeration (limits oxygen diffusion) and promotes denitrification (Figure 3). Thus, the rate of N₂O emission increases with increasing moisture, from air-dry to field capacity (Freney et al. 1979).

Many field studies showed that irrigation or rainfall results in high N₂O emissions, especially when N fertiliser is applied with irrigation or soon before the irrigation event (Mosier and Hutchinson, 1981; Freney et al., 1985; Hansen et al., 1993). Provided other factors remain constant, the water content for maximal N₂O emissions is usually such that both aerobic and anaerobic sites are abundant. Thus soil water contents within the range of 55 to 85% WFPS generally gives maximum N₂O emissions from denitrification and/or nitrification (Granli and Beckman, 1994). Higher water content favours denitrification towards N₂ formation. Thus, the N₂O/N₂ ratio decreases as the soil water content exceeds 75% WFPS (Davidson 1992; Weier et al., 1993). If the soil remains at waterlogged condition for a prolonged period, denitrification and thus N₂O production may cease because of lack of O₂ and NO₃⁻ production from nitrification.

\[
\text{SOM} \rightarrow (\text{Medium O}_2, \text{low H}_2\text{O}): \text{NO} \rightarrow \text{N}_2 \text{O} \quad \text{N}_2 \text{O} \rightarrow \text{N}_2 \ (\text{Low O}_2, \text{high H}_2\text{O})
\]

\[
\text{NH}_4^+ \leftrightarrow \text{NH}_2\text{OH} \leftrightarrow \text{NH}_2\text{ONO} \leftrightarrow \text{NO}_2 \leftrightarrow \text{NO}_3^- \quad (1)
\]

\[
\text{NH}_4^+ \text{Fertiliser} \quad \text{Nitrate Fertiliser}
\]

Equation 1
Therefore, alternate wet and dry cycles stimulate N mineralisation from organic matter, promote NO$_3^-$ accumulation during the dry period, and increases N$_2$O production during the wet period as long as it does not get waterlogged.

### 2.2.2 Temperature

Like other biological processes, nitrification and denitrification rates increase with increasing temperature within a certain range. The optimum soil temperature for nitrification is generally between 25°C and 35°C (Haynes, 1986; Granli and Bøckman, 1994). Higher temperature favours a higher ratio of N$_2$O/NO$_3^-$ from nitrification (Goodroad and Keeney, 1984).

Denitrification has been detected at temperatures <0°C, but usually becomes significant at >5°C (Granli and Bøckman, 1994). The reported optimum temperature for denitrification varies greatly, ranging from 30 to 65°C (Bremner and Shaw, 1958; Malhi et al., 1990; Aulakh et al., 1992; Tiedje, 1994).

As the soil temperature increases, N$_2$O emissions also increase, at least up to 37°C (Freney et al. 1979; Castaldi 2000). The ratio of N$_2$O/N$_2$ due to denitrification declines with increasing temperatures above 37°C (Keeney et al., 1979). The principal mechanism for gaseous N production at higher temperatures (>50°C) is probably chemodenitrification of NO$_3^-$ that can be rapidly produced by thermophilic nitrate respirers (Keeney et al., 1979), resulting in primarily N$_2$ gas emissions from soil.

Direct combustion remains the major source of trace gas emissions from biomass burning. Nitrogen species emissions from biomass burning are dependent upon the nitrogen content of the fuel and fire intensity and duration of flaming combustion vs. smouldering combustion (Lobert et al. 1990). Molecular nitrogen is the most significant species emitted, with the largest contribution coming from flaming combustion (Kuhlbusch et al. 1991). Lobert et al. (1990) found that

![Figure 3](from Dalar et al. 2003).

**Figure 3.** A generalised relationship between water-filled pore space (WFPS) of soils and the relative fluxes of N$_2$O (▲) and N$_2$ (■) from nitrification and denitrification. N$_2$O fluxes dominate between 50% and 80% of WFPS and N$_2$ dominates above 80% WFPS (From Dalal et al. 2003).
up to half of the biomass nitrogen can be converted to molecular nitrogen, \( \text{N}_2 \), through the process of pyrodenitrification. It is also during the hot flaming phase that \( \text{N}_2\text{O} \) is emitted (Lobert et al. 1990; Kuhlbusch et al. 1991), which comprises less than 1 percent of total N lost during fire.

Fire also influences processes of \( \text{N}_2\text{O} \) production within the soil by heating it; causing natural pyrolysis or thermal degradation to both the labile and intractable soil N material (Meyer et al. 1997). This in turn promotes the rate of mineralisation, effectively increasing the mineral nitrogen concentrations in the soil. Increased soil ammonium pools will then promote nitrification rates, leading to substantial increases in nitrate within the soil (Raison 1979) and increasing the likelihood of further nitrogen flow through denitrification (Meyer et al. 1997).

Changes to the fluxes of nitrous oxide between soil and the atmosphere appear to be proportional to fire intensity (Meyer et al. 1997). Studies show that medium to high intensity fires such as savannah and grassland fires may cause significant and persistent increases in nitrous oxide, while low temperature prescribed fires or the field burning of wheat residues do not appear to cause any significant change (Table 6). Uncertainty remains as to the effects of rainfall on nitrous oxide fluxes between soil and the atmosphere. Some studies have indicated that substantial increases in \( \text{N}_2\text{O} \) emissions occurred only after the soil was subjected to simulated rainfall events. Other studies, however, demonstrate that enhancement of \( \text{N}_2\text{O} \) emissions can be independent of rainfall events (Meyer et al. 1997). It appears that this variability in emission rates is related to the extent of the disturbance to soil organic N and its subsequent mineralisation following fire (Meyer et al. 1997).

### 2.2.3 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. Oxygen consumption in soils with high decomposable C contents is normally large, which may result in anaerobic microsites because of insufficient replenishment of air by diffusion. The occurrence of the anaerobic microsites is interactively dependent on the availability of organic C, moisture, soil texture and structure as well as temperature. As discussed above, simultaneous presence of anaerobic and aerobic sites creates a situation favourable for \( \text{N}_2\text{O} \) production.

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 (Dorland and Beauchamp, 1991; Aulakh et al., 1991). Soil disturbance such as drying-wetting and freezing-thawing can liberate more available C, and thus can greatly increase the rate of denitrification (Fotch and Verstraete, 1977). Although the ratio of \( \text{N}_2\text{O}/\text{N}_2 \) from

Table 6. Comparison between the estimated enhanced emissions due to soil disturbance by *in situ* biomass burning and directly from smoke plume emissions.

<table>
<thead>
<tr>
<th>Fire Category</th>
<th>Enhanced Soil Emissions (Gg)</th>
<th>Smoke plume Emissions (Gg)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>( \text{N}_2\text{O} )</td>
<td>( \text{N}_2\text{O} )</td>
</tr>
<tr>
<td></td>
<td>( \text{NO}_x )</td>
<td>( \text{NO}_x )</td>
</tr>
<tr>
<td>Savanna and Temperate Grassland</td>
<td>0.6-12.1</td>
<td>164</td>
</tr>
<tr>
<td>Wheat</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Coarse Grain</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Sugar Cane</td>
<td>0.005-0.1</td>
<td>1.3</td>
</tr>
<tr>
<td>Prescribed burning</td>
<td>0</td>
<td>?</td>
</tr>
</tbody>
</table>

(Source: Meyer et al. 1997)
denitrification decreases with increasing available C supply (Weier et al., 1993), the total amount of N₂O produced from denitrification may be enhanced by the addition of organic materials.

The relationship of organic C availability to nitrification is less straightforward. Addition of organic materials with high C/N ratios promotes microbial immobilisation of NH₄⁺ and hence competes for this substrate against nitrification. If the organic materials have low C/N ratios, however, the rates of nitrification and hence N₂O production are increased by supplying more NH₄⁺ from mineralisation. Decomposable organic materials increase the respiration rate of microorganisms and may induce O₂-limitation, which enhances N₂O production from nitrification.

In general, addition of degradable organic materials increases N₂O production in soils containing NO₃⁻ or applied with fertiliser NO₃⁻ (Murakami et al., 1987). High amount of N₂O can also be produced if materials containing degradable organic N (e.g. animal and green manures) is applied (Bremner and Blackmer, 1981; Goodroad et al., 1984).

### 2.2.4 Soil and Fertiliser Nitrogen

Generally, the rate of denitrification increases with increasing NO₃⁻ content in soil under conditions suitable for denitrification (e.g. high moisture). When other factors such as temperature and available organic C are limiting, however, changes in NO₃⁻ content can have little effect on denitrification rate. Under most circumstances, the presence of NO₃⁻ inhibits the rate of N₂O reduction to N₂ (Blackmer and Bremner, 1978), which results in higher N₂O/N₂ ratio at similar moisture and oxygen content. The mechanisms for this inhibitory effect may be: (i) NO₃⁻ is reduced to NO₂⁻ that inhibits N₂O reduction (Firestone et al., 1980); and (ii) NO₃⁻ is preferred over N₂O as an electron acceptor during denitrification (Cho and Sakdinan, 1978) (Figure 4).

However, flushes of N₂O production are observed

![Figure 4. A diagrammatic representation of the effect of the concentration of nitrate-N in soil on the relative N₂O (▲) and N₂ (●), and total denitrification losses (○) (redrawn from Mosier et al. 1983; Dalal et al. 2003).](image)
usually immediately after NO$_3^-$ addition. In a few hours or days, N$_2$O production decreases, and N$_2$ production increases, even when relatively high concentrations of NO$_3^-$ are still present (Rolston et al., 1978). This phenomenon was explained as the result of a lag period between the activation of NO$_3^-$ reductase and N$_2$O reductase enzymes (Letey et al., 1980).

Under normal field conditions, nitrification is limited by the formation of NH$_4^+$ from mineralisation. In fields applied with fertilisers containing NH$_4^+$, significant N$_2$O emissions have been observed (Brenner and Blackmer, 1981; Hutchinson and Brams, 1992). 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 (Wang and Rees, 1996), confirming that nitrification is a significant source of N$_2$O under aerobic conditions.

The rate of N$_2$O emission after fertiliser application is interactively influenced by the amount and type of fertiliser N, soil properties and the environmental conditions (Granli and Bøckman 1994). For example, use of NO$_3^-$ fertilisers in waterlogged fields or in soils rich in organic C, or NH$_4^+$ fertiliser applied to moist and partially aerated soils, generally results in large N$_2$O emissions.

### 2.2.5 Soil pH and Salinity

Soil pH is a secondary controller of denitrification by mainly affecting the nitrification process. Nitrification is sensitive to extremes in soil pH. The optimal pH for nitrification is approximately 7 to 8 (Fotch and Verstraete, 1977; Haynes, 1986). Laboratory incubations of soils added with NH$_4^+$ under aerobic conditions showed that N$_2$O production could increase by many times with increasing pH up to about 8 (Brenner and Blackmer, 1981; Minami and Fukushi, 1983; Wang and Rees, 1996). At higher pH (pH >8.2), nitrite accumulates in soil, and this is then reduced to N$_2$O since competitive biological oxidation of nitrite by *Nitrobacter* is prohibited (Chalk and Smith 1983).

The maximum rate of denitrification occurs at a pH range of about 7 to 8 (Brenner and Shaw, 1958; Bryan, 1981). Although NO$_3^-$ reduction was detected at a pH as low as 3.5 (Fotch and Verstraete, 1977), denitrification declines if soil acidity shifts towards lower pH. Because the reduction of N$_2$O is inhibited more than the reduction of NO$_3^-$ by acidic conditions, N$_2$O production is enhanced or even becomes dominant at pH <5.5 to 6.0 (Burford and Brenner, 1975; Weier and Gilliam, 1986). As the pH increases, denitrification products tend more, or completely, towards N$_2$ production (Fotch and Verstraete, 1977).

High salinity inhibits both nitrification and denitrification (Inubushi et al., 1999). N$_2$O reductase is susceptible to salt, which may result in N$_2$O accumulation from denitrification under saline conditions (Menyailo et al., 1997). Therefore, N$_2$O production from nitrification could be promoted by increased salt concentration (Low et al., 1997). The effects of soil salinity on nitrification, denitrification and N$_2$O production have been less studied than the effects of other regulating factors. More research is required on this topic.

### 2.2.6 Limitation of Nutrients Other Than Nitrogen

Limitations of other essential nutrients for plant growth limit the ability of plants to utilise ammonium and nitrate N and reduces the overall plant sink for mineral N absorption. Hall and Matson (1999) observed that N$_2$O emissions from 15 kg N/ha applications were much higher from a phosphorus-limited forest soil than the soil high in plant available phosphorus in a tropical forest. 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.
2.3 METHODS AND THEIR LIMITATIONS IN MEASURING N₂O EMISSIONS FROM SOIL

N₂O exchange between soil and the atmosphere is not difficult to measure but high spatial and temporal variability at local scales makes extrapolation even to small landscape units very problematic. High spatial and temporal variability also hinders efforts to identify biological sources and controls of fluxes. The nitrous oxide gas can be measured by infrared analysers (Denmead 1979), and gas chromatography using electron capture detectors (Kammann et al. 2001), tunable diode-laser, infrared detectors, and molecular sieves (0.5 nm) (Van Cleemput et al. 1992).

There are two main strategies for measuring N₂O flux in situ: chambers placed on the soil surface for short periods of time and micrometeorological approaches. Each has strengths and weaknesses.

2.3.1 Chamber Methods

Chamber methods represent the most accessible techniques for measuring N₂O fluxes (Mosier 1989, Hutchinson and Livingston 1993, Holland et al. 1999). They have been instrumental for documenting differences in fluxes among different ecosystems and management regimes, but tend not to capture spatial and temporal variability very well.

In their simplest configuration, chambers are simply open-bottom cylinders or boxes placed on the soil surface for a period of time, usually 1-2 hours, 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. Generally chambers are fitted with a rubber septum through which gases are withdrawn for analysis, and most chamber designs include a small vent to buffer against abrupt changes in barometric pressure over the installation period (see Holland

Figure 5. Probability distribution for chamber-based N₂O flux measurements for a conventionally managed cropping system 1991-1999 (Robertson et al. 2000). Distributions tend to be similarly skewed in even low-flux systems.
et al. 1999) and reduce microclimate within the chamber. 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.

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. The temporal distribution of N\textsubscript{2}O fluxes tends to be log-normally skewed in most ecosystems (Fig. 5). Annual flux estimates are almost always based on many near-zero values averaged with a few number of values 1 – 3 orders of magnitude higher. In one recent study (Wagner-Riddle et al. 1996a), as much as 40% of the annual emissions of N\textsubscript{2}O from one system occurred as a single event.

Automated chambers (e.g. Brumme and Beese 1992, Wang et al. 1997, Ambus and Robertson 1998, Weitz et al. 1999, Galle et al. 2002) 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 changes in fluxes from specific environments (Fig. 6 and 7), and should be especially useful for parameterising quantitative models of N\textsubscript{2}O flux. However, automated chambers are expensive to maintain and similar to static chambers, perhaps more importantly, measure fluxes from relatively small areas, typically <0.25 m\textsuperscript{2}. This can be a significant limitation for whole-ecosystem estimates of N\textsubscript{2}O flux because the short-range spatial variability of fluxes tends to be high (Fig. 8). Automated chambers also tend to be inoperable in heavy snow, and fluxes of trace gases such as N\textsubscript{2}O through snow cover may be a significant part of annual budgets in some environments (Wagner-Riddle 1996b).

![Figure 6. N\textsubscript{2}O and CO\textsubscript{2} fluxes from an automated chamber located in a conventionally managed cropping system. Note the order of magnitude difference in N\textsubscript{2}O flux over the course of day 182-183 (Phillip Robertson, unpublished data).](image)
Figure 7. CO$_2$ emission (soil + plant respiration) and N$_2$O emission from lucerne. Note the order of magnitude difference in N$_2$O flux on 23 October following rainfall. (Source: Meyer et al. 2001).
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. We know that CO$_2$ fluxes can be variable even in monoculture crops such as wheat systems (Aiken et al. 1991). And studies of the spatial variability of denitrification, the most important source of N$_2$O in many ecosystems, suggest equally high variability for N$_2$O flux. For example, Folorunso and Rolston (1984) found coefficients of variation >500% for denitrification fluxes across a fallow vegetable field in central California. Robertson et al. (1988) found spatial variability log-normally skewed across a mid-Michigan old cultivated field. As for temporal variability, noted above, they found the average whole-site flux dominated by a relatively few autocorrelated locations at which fluxes were on average 5-fold greater than fluxes elsewhere in the field (Figure 8). Overcoming this variability with very large chambers (e.g. Christensen et al. 1996) is usually impractical and leads to a different set of compromising assumptions such as uniformity of air mixing and longer equilibration periods that may affect microclimate within chamber.

Capturing temporal and spatial variability for N$_2$O gas fluxes is thus very important to accurately quantify ecosystem-wide fluxes. It is less important for documenting differences among ecosystems at this point, although once most of the within-site variability in ecosystems is captured, the cross-site comparisons with better precision can be made, and more subtle patterns may emerge. But at least for now, one of the major problems facing biogeochemists attempting to assemble landscape to global N$_2$O and other trace gases budgets remains the same problem as in the 1980’s (Robertson et al. 1989), that is, how to accurately quantify the flux from a single ecosystem.

2.3.2 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...
Micrometeorological systems for N$_2$O 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 (Christensen et al. 1996, Mosier et al. 1996; Leuning et al. 1999; Denmead et al. 2000) and sufficient sensitivity to detect fluxes in the systems examined (Wagner-Riddle et al. 1996a, 1997). Very recent advances in FTIR (Griffith and Galle 2000) 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).

Leuning et al. (1997) and Denmead et al. (2000), for example, measured micrometeorological N$_2$O emissions from grazed pastures at Wagga Wagga, and found general agreement with fluxes calculated using NGGIC (1996) and IPCC (1996) methodologies (Table 7). Amongst the micrometeorological approaches used, the mass-balance approach appears to be most precise since N$_2$O concentrations are larger and thus more easily measured due to higher stocking densities than in the normal field situation. On the other hand, application of flux-gradient approach is difficult due to low N$_2$O concentrations over a large region. Although both temporal and spatial variability in N$_2$O emission estimates by micrometeorological approaches are high, these estimates are considerably larger than those estimated from NGGIC algorithms (N$_2$O emission factors of 0.4% for urine and 1.25% for dung) but closer to those estimated from IPCC (1996) algorithms, which employ N$_2$O emission factor of 2% for both urine and dung N (Denmead et al. 2000).

<table>
<thead>
<tr>
<th>Methodology</th>
<th>N$_2$O-N emission$^A$ (kg N/ha.year)</th>
</tr>
</thead>
<tbody>
<tr>
<td>NGGIC (1996)</td>
<td>1.0</td>
</tr>
<tr>
<td>IPCC (1996)</td>
<td>1.8</td>
</tr>
<tr>
<td>Micrometeorological:</td>
<td></td>
</tr>
<tr>
<td>Flux-gradient</td>
<td>2.2 ± 1.2</td>
</tr>
<tr>
<td>Convective boundary layer</td>
<td>1.8 ± 1.4</td>
</tr>
<tr>
<td>Mass balance: $\varepsilon = 0.115$</td>
<td>6.4</td>
</tr>
<tr>
<td>Mass Balance: $\varepsilon = 0.039^B$</td>
<td>2.7</td>
</tr>
</tbody>
</table>

$^A$ Calculated from Denmead et al. (2000).

$^B$ Excludes N$_2$O emissions following rainfall. (from Dalal et al. 2003).

The major limitations of N$_2$O emission measurements by micrometeorological methods include a high deployment cost and technical limitations associated with weather conditions. The deployment cost is related to the technical expertise required and instrumentation, costs for which are several orders of magnitude higher than those for chambers. Nitrous oxide measurement campaigns using micrometeorological approaches thus tend to focus on a single ecosystem type, and continuous measurements for more than a few weeks at a time are rare. Weather limitations are related to boundary layer conditions and air turbulence; in general, fluxes are not measurable at night or during periods of rainfall.

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 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.

2.3.3 Methods for Measuring N$_2$O Emissions During Fire

Emissions of nitrogen containing compounds from the burning of biomass can be measured using a variety of techniques. Such techniques include real time in situ measurements of N$_2$O made over large prescribed fires, the collection of grab-bottle samples, and analysis of small laboratory biomass test fires.

Real time in situ measurements of N$_2$O made over large prescribed fires involves collecting air samples above the fire, which are then subjected to a gas chromatogram with Ni detection (Cofer et al. 1991). Alternatively, air can be collected in stainless steel grab-bottles and analysed 4-8 hours after collection.

The bulk of evidence suggests that concentrations of N$_2$O in grab bottles will be stable for several hours after collection (Cofer et al. 1991).

Small laboratory biomass test fires require small scale burning apparatus built to simulate open fires. An inherent limitation in measuring N$_2$O emissions using these test fires is that only direct combustion emissions of N$_2$O are measured. Combustion itself, however, is not the only source of N$_2$O; significant and persistent increases in N$_2$O also result from soil disturbance by in situ biomass burning. Thus, N$_2$O measured through laboratory biomass test fires and not over the soil during field collections do not constitute the total sum of combustion and soil processes (Cofer et al. 1991). Laboratory biomass test fires also vary from field collections in that field collections typically involve sampling aged smoke plumes, thereby allowing combustion products more opportunity to undergo heterogeneous N$_2$O producing reactions on aerosols (Cofer et al. 1991).
Since $N_2O$ emissions from agroecosystems are sporadic, and temporally (diurnal, season) and spatially variable, modelling of these systems is essential to provide estimates of $N_2O$ emissions over a number of seasons and areas for the National Greenhouse Gas Inventory.

Ecosystem models are necessary to simulate $N_2O$ emissions 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, the ecosystem models are used to extend the results of spot measurement and short time experiments to the 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 (Li et al. 1992a, Riley and Matson 2000, Parton et al. 2001, Potter et al. 1996, Grant and Pattey 1999). The simpler models were developed to simulate nitrogen gas fluxes ($NO_x$, $N_2O$, and $N_2$) at regional and global scales (Parton et al. 1988, Potter et al. 1996), by assuming that gaseous $N$ loss is proportional to $N$ mineralization, with the proportion of $N$ loss for each gas ($N_2$, $N_2O$, and $NO_x$) being a function of the soil water filled pore space (Fig. 3). The more process oriented nitrogen gas flux models (Li et al. 1992b, Grant and Pattey 1999, Parton et al. 2001) were developed to simulate the impact of crop management practices and fertiliser $N$ inputs on $N$ gas fluxes. These process-oriented models all simulate gas fluxes of $N_2$, $N_2O$ and $NO_x$ from nitrification and denitrification and include detailed descriptions of the impact 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 bases of the models are similar and are based on extensive field and laboratory data (Firestone and Davidson 1989, Davidson and Schimel 1995, Groffman 1991). 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. 1992a), while 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 (Del Grasso et al. 2001a). 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).

A formal comparison of models that simulate $N_2O$ gas fluxes was done by Frolking et al. (1998) and included the forerunner of the DAYCENT, DNDC, CASA and NITRO models. The model results were compared with observed daily $N_2O$ gas fluxes from four different agroecosystem sites, which ranged from fertilised grasslands in Scotland to cropping systems in Germany and included sites with different soil textures. The comparison demonstrated that models simulated the major differences in $N_2O$ 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 $N$, $NO_x$ and $N_2O$ are measured at the same time.
DNDC and the DAYCENT (Li et al. 1992a, 1992b, Del Grasso et al. 2001a, 2001b, Parton et al. 2001) have been used extensively to simulate the impact of management practice on N gas fluxes. The DAYCENT model has been used to simulate the effect of the dominant management practice on net greenhouse gas fluxes (CO₂, N₂O and CH₄) for dryland and irrigated regions in the US Great Plains and the Corn Belt regions (Del Grasso et al. 2001a, 2001b). The model results suggest that adopting no-tillage practices for the Corn Belt region and reducing the frequency of fallow in the dryland regions of the US Great Plains will substantially reduce greenhouse emissions for these regions. DNDC has been used to simulate N₂O gas fluxes for current agricultural practices in US (Li et al. 1992b) and is capable of simulating the impact of new management practices that have the potential to reduce agricultural greenhouse gas fluxes.

Wang et al. (1997) simulated fluxes of CO₂ and N₂O by DNDC in a legume pasture system in south-east 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. Further, within DNDC, they had to modify the fraction of water-filled pore space above which denitrification takes place from 0.4 to 0.6. Another modification was to use the soil surface temperature instead of the air temperature since the former could be higher by as much as 20°C in summer. After these modifications, they obtained a reasonable agreement between measured and simulated N₂O emissions from a legume pasture (Fig. 9). This emphasizes the verification and validation of imported models with parameterisation of the Australian conditions.

Figure 9. Measured and simulated N₂O fluxes from a legume pasture in south-east Australia. (DNDC model from Li et al. (1992b), 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; Dalal et al. 2003)).
Conceptually, DAYCENT model is based on the assumption that the total gas emissions from soil are proportional to nitrogen cycling through the system and 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 N₂O and N₂ fluxes from denitrification assumes that N gas flux from denitrification is controlled by the most limiting factor among soil nitrate concentration, carbon substrate supply, and oxygen supply (Fig. 11).

The DAYCENT simulated reasonably well the observed N₂O emissions from various farming systems, including conventional till and no-till, soil textures (coarse, medium and fine), N application (10 kg N/ha), intensively cropped rye (Scotland), crop rotations (Germany) and irrigated barley and maize in Colorado (Fig. 11).

Although DAYCENT simulates seasonal pattern of N₂O fluxes from grasslands reasonably well, it simulates these fluxes very poorly on a daily basis, and hence the significance of diurnal fluctuations in N₂O 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 is required, taking into account the soil texture differences, clay mineralogy, and integration of the ‘hot spots’ over time and field scale.
4. N₂O EMISSIONS FROM VARIOUS ECOSYSTEMS

4.1 N₂O EMISSIONS FROM AGRICULTURAL SYSTEMS

4.1.1 Cropping Soils

**Flooded rice cultivation.** Rice is primarily grown in the Riverina area of New South Wales and Ord River irrigation area in Western Australia. By world standards, Australian rice yields are high (>10t/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 N₂ gas (Figure 3). Often, the limited supply of soluble and readily available C, low nitrate concentration, and competing crop demand for N limits the total gaseous N losses from flooded rice systems (Rennenberg et al. 1992).

For example, application of ammonium sulfate at 80 kg N/ha to flooded rice resulted in only 0.1% as N₂O loss (Freney et al. 1981). It is estimated that N₂O 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 N₂O dissolved and retained in water and not measured (Heincke and Kaupenjohann 1999). Thus, total N₂O 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 N₂O if mineral N (ammonium and/or nitrate) is also present in soil. For example, over an 18-day period after flooding, from 0.8% to 1.4% of nitrate nitrogen present initially in soil was lost as N₂O (Denmead et al. 1979a).

Secondary emissions of N₂O could occur from the deposition of NH₃ volatilised from flooded rice fields after the application of ammoniacal fertilisers since ammonia has a short lifetime in the atmosphere. In flooded rice, NH₃ volatilisation losses can account for from 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 the N fertiliser is used for sugarcane production (Table 5). 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 N₂O-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 N₂O 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 N₂O from the cultivated and no-till soil, respectively, over a 4-day period after the fertiliser application and irrigation. Although N₂ is the major product of denitrification loss of fertiliser N from fine-textured soils, N₂O becomes a major gaseous N loss when soil nitrate concentrations are high (Weier et al. 1996) (Fig 4).

Split application of the N fertiliser may reduce the rapid N₂O losses. Split application of urea and ammonium sulfate to sugarcane soil at 80% WFPS and 100% WFPS results in lower N₂O emissions initially compared with full application of urea although total N₂O emissions over a given season are not significantly different (Weier 1999).

Application of sugarcane trash (10t/ha) to soil fertilised with KNO₃ or urea at the rate of 160 kg N/ha, and followed by 50 mm of irrigation, increased N₂O emissions, and CO₂ respiration, especially from KNO₃; the mean values were 13.3 kg N₂O-N/ha.day for the trash-retained compared with 11.3 kg N₂O-N/ha.day without trash retention (Weier 2000).

Nitrous oxide emissions from N added to sugarcane soils of different texture differed. For example, more
Figure 11. DAYCENT simulated (red bar) and observed (blue bar) N$_2$O emissions from various farming systems, including conventional till (ct) and no-till (nt), soil textures (coarse, medium and fine), N application (10 kg N/ha), intensively cropped rye, crop rotations (G1, multiple cropping with FYM application and G2, multiple cropping with cattle slurry application) and irrigated barley and maize (Source: Del Grosso et al. 2001b).
gaseous N was lost from a fine-textured soil than from a coarse-textured soil; almost 3.2-19.6% of the applied N was lost from a fine-textured soil as compared to <1% from a coarse-textured soil, with 45% to 78% appearing as N$_2$O gas. The N loss was closely related to the soil nitrate concentration and the soil water content (Weier et al. 1996).

**Field burning of sugar cane.** The burning of sugar cane crops immediately before harvest continues to contribute to agricultural and national emissions of N$_2$O. This contribution is decreasing, however, with the rapid introduction of green cane mechanical harvesting in which sugar cane crops are only burnt once every three or four years at the end of the sowing/ratoon cycle (Commonwealth of Australia 2000). This change in harvesting practice is evident in N$_2$O emission estimates. The National Greenhouse Gas Inventory 2000 reported that although N$_2$O emissions from field burning of agricultural residues increased by 37.2% from 1990 to 2000, a 26.8% decline in emissions from sugar cane burning occurred due to changes in harvesting practice (Australian Greenhouse Office 2002). Low to medium intensity fires associated with the field burning of sugar cane can also produce enhanced emissions of N$_2$O due to soil disturbance by in situ biomass burning (Meyer et al. 1997). In 1990 these enhanced N$_2$O emissions were estimated to be between 0.005 and 0.1 kt (Meyer et al. 1997).

**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 rain-fed cotton, often in excess of 100 kg N/ha. Also, cotton is grown in warmer conditions than sugarcane. It is, therefore, expected that N$_2$O 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 carry 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 large part of N supply to cereals, while 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 the Australian soils, have resulted in increasing use of N fertilisers for cereal/oilseed cropping since grain legumes only occupy 2 million ha compared to 6 million ha reduction in improved pastures (Angus 2001). The combined N$_2$O emissions from cropping and ley farming soils are in the order of 24.5 kt N/yr (11.9 Mt CO$_2$-e) in 1999, and 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$_2$O from fertiliser as well as soil mineral N varies considerably. Application of urea to ten soils resulted in N loss as N$_2$O, 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 et al. 1987; Magalhaes and Chalk 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 emissions 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 (Stefanson 1973; Burford and 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 practised 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, thus potentially affecting the \( \text{N}_2\text{O} \) emissions significantly different from that of conventional cultivation. For example, addition of wheat straw at the rate of 10.5 t/ha to a Vertisol doubled the rate of loss of denitrification of applied urea. The \( \text{N}_2\text{O} \) emissions were about 0.5% in Vertisol and 3.0% in Alfisol of applied N when straw was added, and remained higher than 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 \( \text{N}_2\text{O} \) (Bronson and Fillery 1998). Soil temperature effects on \( \text{N}_2\text{O} \) emissions are noticed when seasonal effects on \( \text{N}_2\text{O} \) emissions from cropping soils are studied. Decreasing temperature from 30°C to 5°C slowed the rate of \( \text{N}_2\text{O} \) emissions from a Vertisol and an Alfisol; but the \( \text{N}_2\text{O}/\text{N}_2 \) ratio increased (Avalakki et al. 1995). However, in subtropical environments, the temperature effects appear to be less marked. For example, almost 75% of \( \text{N}_2\text{O} \) emissions occurred in equal amounts in spring and summer under black gram (\textit{Vigna mungo}) in subtropical Queensland. About 88% of the variation in \( \text{N}_2\text{O} \) emissions are found to be associated with monthly rainfall and nitrate concentration (Weier et al. 1991). The microbial population of \( \text{N}_2\text{O} \) producers appears to be controlled by the soil water content under a black gram (\textit{Vigna mungo}) crop (Weier et al.1993b).

Emissions of \( \text{N}_2\text{O} \) 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 \( \text{N}_2\text{O} \) 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 almost equal amount of N as \( \text{N}_2\text{O} \) was lost from ammonium initially and then from nitrate after the second irrigation (Freney et al. 1985). Furthermore, there may be regional differences in \( \text{N}_2\text{O} \) emission patterns since in Western and Southern Australia major rainfall events occur in winter while there is summer-dominant rainfall in northern Australia.

Field burning of wheat and coarse grain stubble. The field burning of agricultural residues, notably for wheat crops, continues to be a small but significant contributor to agricultural and national emissions of \( \text{N}_2\text{O} \). In 1998 it was estimated that 25% of the stubble from wheat crop was burnt annually (NGGIC 1999). Burning of agricultural residues in Australia is, however, becoming less common with the increasing adoption of stubble retention as a form of land management (Commonwealth of Australia 2000). Despite this reduction in field burning, CO\(_2\)-e emissions of CH\(_4\) and \( \text{N}_2\text{O} \) from the burning of agricultural residues, estimated to be 0.4 Mt in 2000, represent a 37.2% increase since 1990 (Australian Greenhouse Office 2002). A 36% increase in the area of land sown for crops since 1990 may have contributed to the increase in emissions over the period 1990 to 2000 (Australian Greenhouse Office 2002). The fire intensity classification is low for field burning of wheat residues and coarse grains, indicating that no enhanced emissions of \( \text{N}_2\text{O} \) due to soil disturbance by \textit{in situ} biomass burning is likely (Meyer et al. 1997).

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, and gained N benefits from the mineralisation of legume N after termination of the pasture phase.
Under the pasture phase, N$_2$O emissions from grazed pastures near Wagga Wagga, NSW 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 N/ha.year (Denmead et al. 2000). Since most non-dairy pastures in Australia are not N fertilised, the N$_2$O emissions 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 ley pasture containing legumes for the cropping phase mineralises a substantial amount of organic N into mineral N (Table 8), and also produces soluble organic C from litter and root decomposition. In Table 8, a conservative estimate of N$_2$O emissions from denitrification following the termination of the pasture phase indicates about similar or higher losses to that during the pasture phase (Galbally et al. 1994). However, Smith et al. (1988) measured N$_2$O emission rate of 30 kg/ha.year from a field sown to irrigated sunflower crop following 8 years of lucerne pasture. Furthermore, waterlogging of these soils after organic N mineralisation has 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 emissions during nitrification in Table 8. 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 pasture will double the estimated N$_2$O emissions, that is, 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 lowering the water table in the cropping regions, where salinity may become a problem. Lucerne is very effective in fixing a substantial amount of atmospheric N$_2$, often exceeding 100 kg N/ha/year (Peoples and Baldock 2001, Table 8). However, after termination of a lucerne phase, nitrate builds up rather rapidly, especially in the subtropical environment (Hossain et al. 1996). When the soil experiences wetting and drying cycles during the summer months, the 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 early autumn. It has the implications for the termination

### Table 8. Estimates of mineral N (mostly nitrate-N) produced during 3-8 months following the termination of the pasture phase and calculated nitrous oxide emissions due to denitrification.

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

$^A$ Percent 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).

(from Dalal et al. 2003)
time for a legume phase, especially in subtropical and tropical regions, and management of legume residues in other environments. Unfortunately, data are lacking to confirm these statements.

Similarly, the extent of N\textsubscript{2}O emissions during and following a pulse crop in the Australian cereal belt is unknown.

### 4.1.2 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\textsubscript{2}O emissions are likely to be high due to high fertiliser rates. Also, many horticultural soils, especially in eastern Australia, are acidic, and therefore, N\textsubscript{2}O may be the dominant denitrification product (Ryden and Lund 1980). On banana plantations, loss rates for N\textsubscript{2}O are estimated to range from 1.3% to 2.9% of applied N (Bouwman 1998). In vegetable crops, N\textsubscript{2}O emissions ranged from <2% to >10% of fertiliser N applied (Ryden and Lund 1980), thus they are much higher than the grain cropping systems.

### 4.2 N\textsubscript{2}O EMISSIONS FROM GRAZING SYSTEMS

#### 4.2.1 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 between 450 and 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), while in New South Wales and Queensland about 100-150 kg N/ha/year are applied to subtropical pastures and up to 300 kg N/ha/year on 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 (2.8-5.1 t/ha of CO\textsubscript{2}-e) from 0 and 200 kg N/ha N applications, respectively (Eckard et al. 2001). At two intensively grazed pastures in New Zealand, N\textsubscript{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 impact on emission through hoof damage in 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 the 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\textsubscript{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\textsubscript{2}O emissions to total denitrification losses are not known. Addition of water to a grass sward as small as 5 mm of irrigation increased nitrous oxide emission rates markedly. Nitrous oxide was produced near the soil surface, and accompanied by the net increase in soil nitrate (Denmead et al. 1979b). 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. 1979b).

The population of N\textsubscript{2}O producers appears to be controlled by the soil water content (Weier et al. 1993). In the USA, almost 37% of fertiliser N (about 240 kg N/ha) was lost from irrigated pastures (Barton et al. 1999). Studies on N\textsubscript{2}O...
emissions from irrigated pastures are required to estimate N$_2$O losses, both in the temperate and subtropical regions.

Another source of N$_2$O 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$_2$O when applied to soil although overall impact on total N$_2$O emissions from agriculture is small (Australian Greenhouse Office 2001).

4.2.2 Extensive Grazing (Improved Pastures vs 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$_2$O emissions are likely to be low in most rangelands; the values reported range from 0 to 1.1 kg N/ha/year, with a median value of 0.2 kg N/ha/year (Galbally et al. 1992; Leuning et al. 1999). Although the rates of N$_2$O emissions from rangeland soils may be low, considering the extent of the total rangeland area (>400 million ha), it is important to measure N$_2$O emissions in the field over an extended period, since small amounts of rainfalls may trigger significant nitrification activity and hence N$_2$O production following a prolonged dry period. For tropical savannas the rates could be much higher, but limited data are available.

Introduction of exotic species may result in increased pasture productivity and thus increase soluble C even in deeper layers. For example, in a green panic (Panicum maximum) pasture, significant amounts of N$_2$O emissions were measured at 50-55 cm depth in the field (Weier et al. 1991; Weier et al. 1993b). Soil temperatures in rangelands vary considerably not only between seasons but also diurnally. The N$_2$O emissions show a diurnal cycle, varying with the temperature of the surface soil. Peak nitrous oxide emission rates occur in the afternoon and minimum near sunrise. Seasonal variation shows the largest emissions of N$_2$O in spring (Denmead et al. 1979b). Almost 75% of N$_2$O emissions occur in spring (55%) and summer (20%). These are associated with CO$_2$ concentrations due to microbial respiration (Weier et al. 1991), and hence intense oxygen demand, resulting in partial anaerobiosis.

Prescribed burning of savanna and temperate grasslands. The burning of tropical savanna and temperate grasslands continues to contribute significantly to agricultural and national emissions of N$_2$O. In Australia, savanna and temperate grassland ecosystems are burnt either anthropogenically or as a result of wildfire. The prescribed burning of tropical savannas occurs for a variety of reasons including pasture management, fuel reduction and traditional Aboriginal fires (Galbally and Meyer 2003). In 1999 it was estimated that 24% of the total annual emissions of nitrous oxide from the agricultural sector were the result of direct emissions from prescribed burning of savannas, carried out primarily in Northern Australia (Australian Greenhouse Office 2001).

Direct combustion remains as the major source of N$_2$O emissions from savanna and grassland fires. However, the relationship between soil nitrogen cycling and fire intensity indicates that medium to high intensity fires, such as savanna and grassland fires, may also cause significant and persistent increases in N$_2$O due to soil disturbance by in situ biomass burning (Meyer et al. 1997). Emissions of N$_2$O directly from savanna and grassland fires in 1999 were estimated to be 11.8 kt, while the estimated enhanced emissions due to soil disturbance by in situ biomass burning were in the range of 0.6-12.1kt (Australian Greenhouse Office 2001). Such estimates highlight the significant contribution made by emissions due to soil disturbance by in situ biomass burning.

The nitrous oxide emissions from the urine patches could be significant in both improved pastures and rangelands (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 needs to be examined after strategic field measurements.

4.3 \textbf{N}_2\text{O} \textbf{EMISSIONS FROM FORESTRY SYSTEMS}

Whilst a broad picture of \text{N}_2\text{O} emission patterns and controls is emerging in forests in the northern hemisphere (e.g. Brumme \textit{et al.} 1999), very few direct measures have been made in Australia. There are two main sources of \text{N}_2\text{O} emissions from forests – fire (both prescribed burning and wildfire) and soil mineral – N transformations, specifically nitrification and de-nitrification (e.g. Robertson \textit{et al.} 1987).

For tropical forests the median rate of \text{N}_2\text{O} emissions is reported to be 1.7 kg N/ha/year (Bouwman 1998). Although Kiese \textit{et al.} (2002) also recorded similar rates (<1.8 kg N\textsubscript{2}O-N/ha/year) in the dry season in three tropical rainforest sites in northeast Queensland, in the wet season, \text{N}_2\text{O} emission rates were 4 to 10 times higher than in the dry season. Assuming on average 3 months of wet season, the annual rate of \text{N}_2\text{O} emission is estimated to vary from 3.2 to 6.7 kg N/ha/year from the northeast Queensland rainforests.

Nitrous oxide emissions from mangrove forests and mangrove sediments could be substantial because mangrove forests are characterised by high N turnover rates and high C production, as well as receiving nutrients from runoff from inland agricultural areas (Corredor \textit{et al.} 1999). However, there have been no measurements of emissions from Australian mangrove forests.

In Australian forests, the following management practices may have important consequences for emissions of \text{N}_2\text{O} to the atmosphere:

\begin{itemize}
  \item Fire management, which determines directly emissions due to prescribed burning (low-intensity fuel reduction burns, and higher intensity slash burns) and indirectly the extent and intensity wildfire and thus emissions due to wildfire.
  \item Harvesting (often associated with slash burning) which can increase rates of soil nitrification.
  \item Aforestation of grazed pastures which affects the nature of the soil N cycle.
  \item N fertiliser (or effluent) application to plantation forests.
\end{itemize}

A brief discussion of each of these follows.

Most Australian native forests are N limited, and there is typically a tight link between ammonium production and uptake in the soil (Connell \textit{et al.} 1995; Attiwill and Adams, 1996). As a consequence, rates of nitrification are typically very low. Low and slowly growing nitrifying populations in such forest soils appears to be the main cause of slow nitrification (e.g. Vitousek \textit{et al.} 1982; Khanna \textit{et al.} 1991), and the addition of inoculum can be sufficient to induce nitrification without changing any other soil characteristics (Bauhus \textit{et al.} 1993).

Soil heating during high intensity fire results in an immediate increase in soil NH\textsubscript{4}-N concentration (e.g. Raison 1979; Walker \textit{et al.} 1986;) and subsequent increased mineralization of fire-modified soil organic matter (Raison \textit{et al.} 1990). Both of these processes provide a greater pool of soil NH\textsubscript{4}-N after fire, and hence opportunity for initiation or increase in nitrification (e.g. Tiedemann \textit{et al.} 1978; Raison 1979; Ellis \textit{et al.} 1982; Weston and Attiwill 1996). In the field, nitrification was induced (it did not occur in unburnt soil) under ashbeds following intensive harvest and slash burning of coastal Victorian eucalypt forest growing on low fertility soils (Raison \textit{et al.} 1991). However, significant nitrification rates did not occur until ~ 9-18 months after fire, by which time recolonising vegetation exists to take up much of the nitrate, thus reducing the risks of denitrification. On higher fertility soils under mountain ash (\textit{E. regnans}) forest, nitrification is greatly increased following stand-replacing wildfire as well as after clearfelling and slashburning (Adams and Attiwill 1986; Weston and Attiwill 1996). Under these situations
nitrification rates and nitrate accumulation are much greater, and more prolonged (up to 1 year) as is the potential for N$_2$O losses.

Low-intensity prescribed burning is widely applied as a management tool for fuel reduction in Australian native forests. Too frequent application of such fire can result in depletion of soil N stocks and lowered rates of soil N mineralization (see Raison et al. 1993 for discussion of evidence and mechanisms). Application of low-intensity prescribed fire however probably has little effect on N$_2$O emissions from the forest soils, which typically have rather low rates of mineralization and negligible or very low rates of nitrification. Meyer et al. (1997) concluded that low-intensity prescribed burns applied to forests used for either wood production or for conservation, may not change the flux of N$_2$O between the soil and the atmosphere. However, this conclusion is based on a very limited set of observations.

Fires do result in direct emission of N$_2$O as a consequence of combustion processes (Galbally et al. 1992; Prather et al. 2001). The total direct combustion emissions of N$_2$O from fires (prescribed burns and wildfire) in Australia in 1997 was estimated to be 291 kt CO$_2$-e (NGGI, 1999).

The effects of intense wildfire on nitrification rates in forests which are not killed, but which re-sprout via epicormic growth has been little studied. Such forest response is the most widespread in the Australian landscape (stand replacing fires are confined to ash-type forests such as E. regnans). Khanna and Raison (1986) found that high intensity (crown defoliating) fire in E. pauciflora forest did not stimulate nitrification even under ashbeds. Generally the uptake of NH$_4^+$ by the recovering vegetation would be expected to limit the potential for prolonged nitrification and thus N$_2$O loss.

Grazed pastures are often significant sources of N$_2$O emissions (e.g. Yamulki et al. 2001) resulting from high N inputs to the soil from legumes, fertiliser and animal excreta, and consequent rapid rates of nitrification which create potential for N$_2$O formation and loss. Following afforestation, rates of N input and soil N mineralization decline. For example, O’Connell et al. (2003) have found that inputs of C – rich litter result in a rapid decline in soil N mineralization rates within a few years after E. globulus plantations are established on improved pastures in the SW of WA.

Afforestation will also reduce CH$_4$ emissions by grazing animals, and also creates a potential for forest soil to become an enhanced sink (by oxidization) for CH$_4$, (e.g. MacDonald et al. 1997).

Thus, there is the potential for afforestation of grazed pastures, which is currently occurring at rates of 60-100,000 ha per annum in Australia, to have a major beneficial effect on non-CO$_2$ GHG emissions. It is thus important to quantify the effects of such land-use change on total (carbon storage plus change in non-CO$_2$ emissions) GHG balance, and to enhance our understanding of the processes affecting such change.

N fertilisers are used to establish plantations, and sometimes are applied to stimulate growth at mid-rotation. Annual denitrification rates for plantation forest soils vary, depending on stand age, site preparation, fertilisation, and wastewater application. Denitrification rates are likely to be higher after clear felling and N fertilisation than under natural forest but are probably lower than for grazed 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, the exact measurements of N$_2$O emissions or N$_2$O/N$_2$ ratio are rare under Australian forests and plantation soils.

In a study of denitrification losses following the application of treated sewage effluent to young plantations, Smith and Bond (1999) concluded that losses were very small on a freely draining soil. However, measurements of N$_2$O emissions were not made.

Pu et al. (2001) measured significant emission of N$_2$O following harvest of hoop pine plantations in south-east Queensland. Losses were greatest under
windrows of decaying slash, and the authors suggested that significant rates of denitrification may last for 2 years after harvest. It is well known that final harvest increases soil N mineralization, including nitrification, in plantation forests (e.g. Smethurst and Nambiar 1990) and that sites are vulnerable to N losses during the inter-rotation period. However, little is known of N₂O emission rates during this period, and further studies are needed to quantify this.

4.4 N₂O EMISSIONS FROM OTHER SYSTEMS
Termite mounds emit significant amounts of nitrous oxide during spring and summer (Khalil et al. 1990; French et al. 1997). Exact contribution of termite mounds to N₂O emissions is not known since it is calculated from the estimates of CH₄ production as a ratio of CH₄/N₂O of 20:1 on a mass basis (Galbally et al. 1992).

5. MITIGATING N₂O EMISSIONS IN VARIOUS ECOSYSTEMS

5.1 MITIGATING N₂O EMISSIONS FROM AGRICULTURAL SYSTEMS
Flooded rice cropping. In waterlogged or flooded soils, where aeration is restricted, less N₂O is emitted into the atmosphere because more N₂O is converted into N₂ 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 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 fertiliser-induced loss of nitrous oxide only in a calcareous soil, which accumulated nitrite in the fertiliser band (Magalhaes et al. 1984). Wax-coated calcium carbide reduces significantly the rate of N₂O emission while nitrapyrin is much less effective in a flooded rice system (Keerthisinghe et al. 1993). Denitrification and N₂O losses of urea from flooded rice systems are further reduced when urea is deep placed as compared to surface broadcast application (Keerthisinghe et al. 1993).

The strategies that increase the efficiency of N fertiliser use will reduce N₂O emissions (Aulakh et al. 1992; Mosier et al. 1996). These strategies include: fertiliser 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 the soil, and the use of slow-release fertilisers, and urease and nitrification inhibitors (Freney 1997a).

Although these approaches enhance nitrogen use efficiency, they do not necessarily reduce N₂O emissions significantly. However, newly developed urease and nitrification inhibitors and coated-urea have the capacity to prevent loss of N, including the loss as N₂O, as well as increase crop yields.
**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$_2$O:N$_2$ balance between N$_2$O 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 crop when crop needs it and in the amount the crop can readily utilise it through regular soil analysis for mineral N and crop monitoring for N, split applications, and applying in drip irrigation. Further reductions in N$_2$O 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 that in flooded-rice soils, urease and nitrification inhibitors such as wax-coated calcium carbide (Keerthisinghe et al. 1993), etridiazole, dicyandiamide and 3,4-dimethylpyrazole phosphate (Weiske et al. 2001) have been found to reduce nitrate concentration in arable soils from the application of ammoniacal fertilisers, and thus reducing N$_2$O emissions. For example, Weiske et al. (2001) found that dicyandiamide and 3,4-dimethylpyrazole phosphate reduced N$_2$O emissions from the fertilised plots over a 3-year period by 29% and 45%, respectively. However, the cost-effectiveness of the latter two nitrification inhibitors needs to be further investigated, both in terms of economics of use and N use efficiency and also N$_2$O 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 for the use of nitrification inhibitors. Similarly, the usefulness of urease inhibitors such as Agrotain (n-butyl-thiophosphoric triamide) also need to be tested (McGuffog 2001) since it may have an added benefit of reducing N$_2$O emissions in the nitrite formation process as well.

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 nitrous oxide mitigation alone (Phipps and Hall 1994).

Increasing nitrogen use efficiency should reduce gaseous losses of N, including that of nitrous oxide.

Mitigating N$_2$O emissions arising from field burning of sugar cane, wheat stubble, and coarse grain stubble involve the use of alternative management practices, such as the adoption of stubble retention, which reduces direct N$_2$O emissions into the atmosphere by minimising the field burning of agricultural residues (CSIRO, 2000). The practice of crop residue retention is increasing in both the Australian cereal belt and in the sugar cane industry, with the adoption of techniques such as green cane harvesting and trash blanketing.

### 5.2 MITIGATING N$_2$O EMISSIONS FROM GRAZING SYSTEMS

**Dairying.** Since N fertiliser use is likely to increase on dairy pastures, the N fertiliser management practices must ensure that pastures use N efficiently. Therefore, generally similar strategies are required for fertiliser management to that required for cropping, mentioned above, with a major significant difference. Unlike crops, most established pastures have active extensive root systems 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 in commensurate with the pasture growth. Then N fertiliser application must be made 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, nitrogen 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 N fertiliser application. This also avoids the hoof damage to soil structure and thus maintains oxygen diffusion into soil. However, the effect of sprinkler irrigation on N₂O emissions from fertiliser application as well as soil mineral N is unknown. Barton et al. (1999) reported that improved drainage in forest soils reduced N losses from 40 kg N/ha/yr to <1 kg N/ha/yr. However, this may increase NO₃ losses in drainage waters.

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, N₂O emissions from rangelands are about 0.2 kg N/ha/year. A number of options have been suggested to reduce the N₂O 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 N₂O emissions from rangelands. These management options are likely to enhance the resource management of rangelands as well. On the other hand, Howden et al. (1996) suggested that reducing sheep stock number and spring lamb production appear to be two options for reducing nitrous oxide emissions from the Australian grazing lands, although this approach is likely to reduce farm income by about 15% by reducing the stock number by 18% (Howden et al. 1996). However, these options could form an overall strategy for sustainable grazing systems.

Minimising emissions from those prescribed burning practices associated with pasture management will require practical management and monitoring protocols specifically designed to provide opportunities for local landholders to develop the expertise required to implement more appropriate pasture management and burning practices (Australian Greenhouse Office 2001).

It should not be assumed, however, that stopping anthropogenic fires would lead to a reduction in greenhouse gas emissions, as these would be replaced by natural wildfires. Fire will continue to play a major role in many rangeland ecosystems since specific flora and fauna are fire dependent and removing fires will result in a loss of biodiversity.

### 5.3 MITIGATING N₂O EMISSIONS FROM FORESTRY SYSTEMS

Nitrogen management for plantations and agroforestry systems follows similar principles to that 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 that mineral N in soil is kept low.

Land preparation for plantation and agroforestry should involve minimum soil disturbance or restricted to a limited area to reduce organic N mineralisation. Understorey legumes could be utilised to provide N to trees and plantations, especially in the early years.

Prescribed fires in forestry management appears to release negligible or low amounts of N₂O (Meyer et al. 1997) due to low flame intensity (Kuhlbusch et al. 1991), and probably it is an appropriate practice for not only controlling wild fires but also reducing N₂O emissions from forestry systems.
6. SUMMARY AND CONCLUSIONS

Within the NGGI, there is a close relationship between the amount of nitrogen fertiliser used and the estimated \( N_2O \) emissions by NGGIC (1999) (Fig. 12). The IPCC (1996) default emissions factor is 1.25 kg of \( N_2O \)-N emitted for 100 kg of fertiliser N, while in Figure 12, it appears to be 1.6 kg of \( N_2O \)-N per 100 kg of fertiliser N use, assuming \( N_2O \) emissions from other sources has remained constant over the last 10 years.

Most \( N_2O \) emissions measurements are of very short duration and over a limited area, and therefore do not measure the total \( N_2O \) emissions from either fertiliser or soils accurately. When \( N_2O \) emissions have been measured for over a year, about 2% of nitrogen fertiliser applied appears as \( N_2O \) in temperate environments (Cates et al. 1987; Mosier et al. 1996). No long-term \( N_2O \) emission measurements exist for the subtropical and tropical regions. Bouwman (1998) believes that the \( N_2O \) emission rates in these regions are likely to be higher than 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 \( N_2O \) emissions during the fallowing period following the pasture phase. Similarly, a few measurements are available to estimate the \( N_2O \) emissions from frequent fires in savannas. Since these areas are large, even modest \( N_2O \) emissions per unit area would have a large impact on 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 \( N_2O \) emissions from these systems.

Management practices to improve nitrogen use efficiency and minimise \( N_2O \) emissions from nitrogen fertilisers include the following (Freney 1997b; Laegreid et al. 1999; McGuffog 2001):

![Graph showing the relationship between nitrous oxide emissions and nitrogen fertiliser use in Australian agriculture from 1990 to 1999](from Dalal et al. 2003).
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 other N sources such as manure or waste.

2. Apply fertiliser N at the rate and time to meet crop/pasture needs (Fig. 13), 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 nutrients 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; apply 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. Fertilizers 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. Fertilizer may be formulated with urease and/or nitrification inhibitors or physical coatings to synchronise fertiliser N release to that of crop/pasture growth needs so that at any given time 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.

Secondary considerations to reduce nitrous oxide emissions include:

- Oxygen supply/soil water content (water-filled pore space <40% increases nitrification but reduces nitrous oxide loss, >90% increases denitrification as a nitrogen gas loss); examples include arable cropping/pasture/forestry, and flooded rice. Improve oxygen diffusion in soil by eliminating the compacted layer.

- Carbon substrate supply (readily available carbon) creates ‘hot spots’ of microbial growth, and hence nitrous oxide emissions; examples are addition or incorporation of biomass of high carbon: nitrogen ratio such as non-legumes rather than legume biomass.

- Soil organic matter management to manipulate carbon substrate and oxygen/water supply.
Soil pH and salinity (salinity and alkaline pH enhance the nitrous oxide emissions due to the persistence of nitrites); soil amendments such as application of gypsum or crop residues of high carbon: nitrogen ratio reduces nitrous oxide emissions.

Perennial crop/pasture/tree systems mostly provide optimum environment to reduce 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 the agricultural soils.

Prescribed burning of savanna primarily carried out in northern Australia continues to contribute to nitrous oxide and other oxides of nitrogen (NO, NO₂, etc) emissions. Even light rainfall following burning increases nitrous oxide emissions. However, it is not known whether elimination of prescribed burning would reduce the annual amounts of nitrous oxide emissions although the amounts of NO and NO₂ emissions will be reduced but increase the frequency of wild fires. 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 undisturbed lands. Restriction of land clearing, therefore, will reduce nitrous oxide emissions although the exact reductions are not known.

6.1 POLICY CONSIDERATIONS AND RESEARCH OPPORTUNITIES

Nitrogen fertiliser use is likely to increase for crop/pasture production, since soil N supply is declining as improved pasture areas decrease. Also crop/livestock production will intensify due to marketing pressure, and greater control over product quality and supply.

Education is critical in the efficient use of fertilisers for crop/pasture production to reduce fertiliser waste as well as reduce nitrous oxide emissions. Fertilizer Industry Federation of Australia (FIFA)
should play a pivotal role in partnership with the relevant industry bodies in preparing Codes of Practice for Efficient Fertiliser Use (a current FIFA initiative - Cracking the Nutrient Code: Guidelines for Developing Nutrient Management Codes of Practice).

Although, both State and Federal legislations exist for fertiliser quality assurance, consideration should be given to discourage fertiliser applications out of season, and in excess of crop demands. This will also require the elimination of price incentives by suppliers for out of crop season applications.

Incentives may be offered to producers for machinery and equipment for precise applications of fertilisers, when (at or near sowing or in-crop) and where (prescribed area in a paddock) it is needed. There is a limited scope to encourage the use of slow-release fertilisers to mimic plant needs due to higher costs of the amended fertilisers, which are rarely recovered by increased yields. Recently, it has been claimed that the application of 3,4-dimethylpyrazole phosphate with urea, reduced nitrous oxide emissions by 45% over a 3-year period in the field in Germany. We need to examine its effectiveness under Australian conditions.

There are a number of other areas of research on nitrous oxide emissions, both for improving National Greenhouse Gas Inventory (reducing the uncertainties in the emission factors, the activity data, lack of coverage of measurements, spatial aggregation, and lack of information on specific on-farm practices), as well as mitigation options for reducing nitrous oxide emissions from agriculture.

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 micrometeorological 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 versus 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 versus fertiliser nitrogen, (iii) 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 micrometeorological techniques using high precision analytical instruments, and covering a range of strategic activities in the agriculture sector, mentioned above.

We recognise that carbon dioxide emissions and sinks as well as methane emissions and sinks interact strongly with nitrous oxide 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.
The Kyoto Protocol, Articles 3.3 and 3.4, provides an opportunity to reduce nitrous oxide emissions from agricultural lands. Admittedly, designing policy to reduce N$_2$O emissions from agriculture is not a simple one (Hinchy et al. 1998) since N forms an integral component of any sustainable agricultural systems. If appropriate price signals were to be passed to producers about abatement options, policy measures have to be applied at the farm level. However, monitoring and enforcement costs per unit of emission abatement are likely to be high (Hinchy et al. 1998).

Policy and regulatory arrangements need to be put in place to support greenhouse mitigation options for nitrous oxide. These may include:

(i) Mechanisms for measuring, monitoring and verifying nitrous oxide emissions and the compliance with the contractual arrangements;

(ii) Mechanism to integrate nitrous oxide emissions and abatement options with other greenhouse gases abatement programs such as carbon sinks, and natural resource management;

(iii) Devise framework for allocating mandatory carbon sinks to offset nitrous oxides as a trade off;

(iv) Encourage the national and international marketing for trading in the offsetting of nitrous oxide emissions;

(v) Encourage research and development on the impact of abatement options, and trading arrangements on the environment, profitability and sustainability of the agricultural industries; and economic and social welfare of the rural communities and Australian community.

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