SNAP CODE:

100100
100101
100102
100103
100104
100105

SOURCE ACTIVITY TITLE:

CULTURES WITH FERTILISERS
Permanent Crops
Arable Land Crops
Rice Field
Market Gardening
Grassland

NOSE CODE:

110.01.01
110.01.02
110.01.03
110.01.04
110.01.05

NFR CODE:

4 D 1
4 C

1 ACTIVITIES INCLUDED

This chapter considers the emissions of ammonia (NH_3), nitrous oxide (N_2O), other oxides of nitrogen (NO_x) and volatile organic compounds (VOCs) from agricultural soils to which nitrogen (N)-containing fertilisers are applied. Emissions from growing and decomposing fertilised plants are also included. Emissions following application of animal manures to land are considered in SNAP code 100500, Manure Management. However, losses of NH_3 from N-fertilisers applied to grass grazed by livestock, are difficult to distinguish from subsequent NH_3 emissions from urine patches produced by grazing animals (Black et al. 1984). In consequence NH_3 losses from grazed grassland, (but not following manure spreading), are dealt with in this chapter. Persistent Organic Pollutants are dealt with separately under SNAP code 100600, Use of Pesticides.

Emissions of N gases from fertilised agricultural land are generally related closely to the amount and, in some cases, type of fertiliser -N applied.

This chapter is a development of chapter ag100100 (EEA 1996) which considered only emissions of NH_3.

Sub-code 100106, Fallows, is not included, as N-fertiliser will not be applied to such land. In most cases there is insufficient evidence to discriminate between the sub-codes when estimating emissions. However there is some evidence that NH_3 emissions from grassland

and rice fields are significantly different to NH_3 emissions from the other sub-codes. Separate NH_3 emission factors are therefore used for 100103 and 100105.

2 CONTRIBUTIONS TO TOTAL EMISSIONS

Fertilised agricultural land typically contributes approximately one tenth of the total source strength for European emissions of NH_3, N_2O and NO.

The major source of NH_3 in Europe is volatilisation from livestock excreta. While 10-20% of agricultural NH_3 emissions have been estimated to derive from the volatilisation of N-fertilisers and from fertilised crops (Asman 1992; ECETOC 1994). This range may be compared with the estimate of the CORINAIR94 inventory (ETCAE, 1997) shown in Table 2.1. CORINAIR estimated a contribution of *c.* 5% to the total which is probably an underestimate compared with more recent values.

The CORINAIR inventory estimated that, via soil processes, fertilisers and crops contributed *c.* 20% of N_2O emissions on a European scale. Soils and crops are considered to be a net sink for most NO_x compounds. However NO may be released from soils during nitrification following mineralisation of N from incorporated crop residues and soil organic matter. Only NO emissions are therefore discussed. No figures were provided by CORINAIR for NO emissions. Estimates of NO emissions are very uncertain, but soils may contribute c. 8% of total European emissions. On a hot summer day this fraction may increase to *c.* 27% (Stohl *et al.* 1996).

Table 2.1: % Contribution of total emissions of the CORINAIR94 inventory
(28 countries)

Source-activity	**SNAP-code**								
		SO_2	NO_x	NMVOC	CH_4	CO	CO_2	N_2O	NH_3
Cultures with Fertilisers	100100	-	3.0	0.4	4.3	-	-	23.1	5.3

0 = emissions are reported, but the exact value is below the rounding limit (0.1 per cent)
\- = no emissions are reported

As can be seen from Table 2.1, VOCs from this SNAP Code represent < 1% of current total emission estimates, and do not therefore require a methodology for calculation. However given current uncertainties over the magnitude of VOC emissions from agricultural crops, some information is given in this chapter, in order to provide background information, and to highlight current uncertainties.

3 GENERAL

3.1 Description

3.1.1 Ammonia

The best information on NH_3 emissions from cultures with fertilisers concerns the direct emissions following fertiliser-N application. The evidence for direct emissions from, and uptake by, plant foliage is also good, though estimates of net emissions are much more uncertain. Although estimates of the component emissions from crop foliage have been made (ECETOC 1994), it is often difficult to separate the direct fertiliser and plant emissions in practice, since both are a function of fertiliser-N supply, and in many experiments total emissions were measured. General reviews and estimates of NH_3 from these sources have been provided by Asman (1992), ECETOC (1994) and Sutton et al. (1995b).

Emissions of NH_3 from mineral fertilisers depend on the type of N-fertiliser applied, soil type (especially soil pH), meteorological conditions and time of application in relation to crop canopy development. In particular, the type of N-fertiliser applied has a great effect on the magnitude of emissions (Whitehead and Raistrick 1990). Emissions are largest from urea fertiliser because it hydrolyses in the soil and releases NH_3. Emissions from ammonium sulphate (AS) may also be large, but these are very dependent on soil pH, with larger emissions from calcareous soils. Other fertilisers, such as ammonium nitrate (AN), are more neutral in pH and produce much smaller emissions. These are often difficult to distinguish in measurements from plant-atmosphere fluxes. Fertilisers containing only nitrate (NO_3^-) will not emit NH_3 directly, but may increase NH_3 emissions by fertilised crops.

The estimates of NH_3 emission from decomposing crop residues are also extremely uncertain, and emissions from this source are likely to be very variable. The limited experimental work (Whitehead and Lockyer 1989) found only emission from grass foliage with a high N content where much N-fertiliser had been applied, and was restricted to laboratory measurements which may overestimate emission. Recent measurements have also indicated significant NH_3 emissions from decomposing brassica leaves (including oilseed rape, cauliflower), especially after cutting (Sutton et al. 1996; R Harrison, ADAS, UK, pers. comm.).

Depending on the interpretation of results, emissions from growing vegetation and from decomposing grass herbage may be treated as an additional emission. Or they may be included together with soil emissions as a single emission factor. The time scale over which the emission estimates are made is important to note. Fertiliser emissions are largest in the days after application, but in some instances (e.g. urea applied in dry conditions resulting in a slow hydrolysis), fertiliser emission may proceed for over a month after application (Sutton et al. 1995a). For background emissions (other than initial fertiliser losses) during the plant growing period, most of the emission occurs indirectly from the foliage. However, as well as being influenced by air concentration and environmental conditions, both emission and deposition occur on diurnal cycles. It has been suggested that for some arable ecosystems, on an annual basis, foliar emission may balance dry deposition to the same vegetation (Sutton et al. 1995a). Foliar emissions are expected to be larger from annual cereal crops than from fertilised agricultural grassland, since much of the emission may occur during the grain ripening and vegetation senescence phase (Schjørring 1991). In contrast, where agricultural grassland, or other crops, are cut and left in the field for extended periods, decomposition may result in emissions of similar magnitude. Emissions from this source are extremely uncertain, and probably vary greatly from year to year depending on environmental conditions and success of harvests.

Emissions of NH_3 from grazed grassland have been shown to increase with increasing fertiliser-N application (Jarvis et al. 1989; Bussink 1992). Moreover the proportion of NH_3 emitted increases with increasing fertiliser-N.

Particularly large emissions (*c.* 30% of N applied), have been measured following application of urea and AS to flooded rice fields.

3.1.2 Nitrous Oxide

In soil N_2O is produced predominantly by two microbial processes: nitrification (the oxidation of ammonium (NH_4^+) to nitrate (NO_3^-) and denitrification (the reduction of NO_3^- to gaseous forms of N, ultimately N_2O and N_2). The rate of N_2O production is primarily dependent on the availability of mineral N in the soil (e.g. Bouwman 1996) Maximum N_2O emissions are generally observed within 2 to 3 weeks of N-fertiliser application. The magnitude of the emissions depend on the rate and form of fertiliser applied, the crop type and the soil temperature and soil moisture content. However, it is not possible to derive emission factors for different fertilisers or soil types from existing data (Bouwman 1996). Therefore, the IPCC method defines only one emission factor for all types of N input.

Following the IPCC Methodology (IPCC/OECD 1997), N_2O emissions from agricultural soils may be calculated as the sum of :

i. direct soil emissions (1.25% of N inputs are emitted as N_2O-N); (where N inputs are from fertilisers, biological N fixation and crop residues). See IPCC Worksheet 4-5, sheet 1.

ii direct N_2O emissions from cultivation of histosols (IPCC Worksheet 4-5, sheet 2).

iii. direct soil emissions (2% of N inputs) from grazing animals (IPCC Worksheet 4-5, sheet 3).

iv. indirect emissions following deposition of NH_3 and NO_x (1% of N deposited as NH_3 and NO_x is subsequently re-emitted as N_2O), or leaching or run off (2.5 % of N leached or run off, IPCC Worksheet 4-5, sheets 4 and 5).

Prior to estimation of direct N_2O emissions, fertiliser N inputs are reduced by 10%, and excretal N returns by 20%, to allow for N lost as NH_3. Emissions from animal waste management and manure spreading are included in SNAP Code 100500, Manure Management.

Direct emissions from cultures with fertilisers include emissions which are induced by N input (fertiliser, excretal N deposited during grazing, biological N fixation and crop residues). In addition, cultivation of organic soils (histosols) is regarded as a direct source of N_2O. The magnitude of direct N_2O emissions varies with a range of soil and environmental factors. Application of N-fertiliser to, or incorporation of N-rich crop residues into, moisture-retentive soils produces greater N_2O emissions than application to free-draining soils (Skiba et al. 1992). Application to or incorporation into warm soils is also likely to lead to greater emissions than from soils which are cold. However, recent studies showed, that the largest N_2O emissions occur during thawing of frozen soils (Müller et al. 1997), and the total emissions between November and February were 50% of the total annual flux (Kaiser et al. 1997). Rapid crop growth, and demand for NO_3^--N, will reduce N_2O emissions by reducing the pool of mineral N available for denitrification (Yamulki et al. 1995). Increased exudation of C from plants may also increase denitrification.

These soil and environmental factors also influence the magnitude of indirect N_2O emissions following atmospheric deposition of NH_3 and NO_x.

3.1.3 Nitric Oxide

Nitric oxide (NO) may be emitted either as a consequence of nitrification, or denitrification. In cultures with fertilisers, where pH is likely to be maintained above 5.0, nitrification is considered to be the dominant pathway of NO emission (Remde & Conrad 1991; Skiba et al. 1997). The main determinant of NO production in agricultural soils is mineral N concentration (Skiba et al. 1997). This is increased by N-fertiliser application, excretal N deposited during grazing, crop residue incorporation and cultivation.

Current data on NO emissions in relation to fertiliser-N use were reviewed by Skiba et al. (1997). Losses ranged from 0.003 to 11% of applied fertiliser-N, with a geometric mean emission of 0.3% applied N. In view of the sparse and skewed nature of the data, this estimate is proposed in preference to that of Yienger & Levy (1995) who used an arithmetic mean of 2.5% loss of fertiliser-N to estimate NO emissions.

Activities such as tillage and incorporation were considered to increase NO emissions by a factor of 4 (Skiba et al. 1997). Thus a knowledge of the N concentration and mineralisation rate of crop residues could provide an estimate of soil NH_4^+ on which to base an emission estimate. A knowledge of soil N content could also allow an estimate to be made of NO emissions following cultivation.

3.1.4 Volatile Organic Compounds

Volatile Organic Compounds (VOCs) are defined as “all those organic compounds, other than methane, which can produce photochemical oxidants by reaction with nitrogen oxides in the presence of sunlight”.(ref to be added)

Three categories of sources may be distinguished:

1. activities that emit VOCs by combustion or evaporation;
2. land clearing, including burning;
3. Biogenic processes.

The primary sources in the agricultural sector are:

1. burning stubble and other plant wastes;
2. The use of organic solvents in pesticide production;
3. Anaerobic degradation of livestock feed and animal excreta.

These 3 major sources are dealt with elsewhere. Stubble burning in SNAP code 100300, emissions from burning other crop residues in SNAP code 090700 (Open Burning of

Agricultural Wastes). Emissions from livestock manures are included in SNAP Code 100500, Manure Management.

The emission of some VOCs may be of benefit to plants, e.g. to attract pollinating insects, while others may be waste products or a means of losing surplus energy (Hewitt and Street 1992). Ethene emission has been observed to increase when plants are under stress. Factors that can influence the emission of VOCs include temperature and light intensity, plant growth stage, water stress, air pollution and senescence (Hewitt and Street 1992). Emissions of VOCs from plants have usually been associated with woodlands, which predominantly emit isoprenes and terpenes (König et al. 1996). Hewitt and Street (1992) took qualitative measurements of the major grass and crop species in the UK (except for barley, *Hordeum vulgare*). The only crop species producing any significant emissions was Blackcurrant (*Ribes nigrum*). However these workers cautioned against classifying plants as 'non-emitters' on the basis of limited measurements, as plant growth stage had been shown to be an important factor in emission. The role of the soil as a source or sink of VOCs requires investigation

Hewitt and Street (1992) concluded that trees are the main emitters of non-methane hydrocarbons (NMHCs). Other plants, and crops are negligible in comparison. König et al. (1996) noted that in earlier studies NMHCs had been regarded as the major component of VOC emissions. However König et al. (1996) found oxygenated VOCs to be the major VOC emissions from cereals. In that study emissions were not invariably greater from trees than from agricultural crops. Total emissions of NMHCs per unit of dry matter ($\mu g\ g^{-1}\ ha^{-1}$) were : wheat, 0.01; beech (*Fagus sylvatica*), 0.08-0.20; hornbeam (*Carpinus betulis*), 0.16; rye,0.22-0.27; oilseed rape, 0.20-0.32; birch (*Betula pendula*), 0.56; and oak (*Quercus petrae*), 0.78. Emissions were increased during flowering for rape, but not for rye.

3.2 Definitions

Animal Manures. Animal excreta deposited in houses and on yards, collected, either with bedding or without, to be applied to land.

Livestock excreta. Livestock excreta deposited at any time, including while grazing.

Fertilised agricultural grassland. Grassland, to be used for either cutting grass or conservation, grazing or both, to which synthetic N-fertilisers have been applied.

Crop residues. The unharvested parts of crops that are left on the field and ultimately incorporated into the soil.

3.3 Controls

3.3.1 Ammonia

Emissions of NH_3 from crops have not generally been seen as a major option for control, primarily because the emissions from animal husbandry are much larger and therefore provide greater scope for reducing total emissions.

However, there may be potential for reducing crop emissions by switching from urea to other N-fertilisers. Urea has been considered to contribute approximately 50% of the fertiliser emissions in western Europe (ECETOC 1994) because of its greater NH_3 volatilisation rate. A potentially effective control of fertiliser emissions would, therefore, be to use alternative N-fertilisers with smaller NH_3 emissions. A further possibility is to add urease regulators/inhibitors to urea fertiliser which are expected to reduce emissions. Costs of these measures would include the differential price of more expensive fertilisers or of inhibitors. However, it should also be noted that urease inhibitors may have other undesirable environmental effects which need to be assessed before these are recommended. The most effective inhibitors have been toxic heavy metals (eg Hg and Cu), and phenolic compounds which may have a general toxicity to micro-organisms (Fenn and Hossner 1985).

As NH_3 emissions are strongly a function of N supply, another potential control is to use cultivars or crop species which require less N. Use of less N-demanding species and cultivars will generally reduce total produce yields, the costing of which may be difficult because of the close link to produce supply and market values. However, it may be appropriate to consider reduction in fertiliser-N inputs where these have an additional benefit for other environmental effects, such as reducing NO_3^- leaching.

Emissions may also be reduced by placing the fertiliser granule into the soil at the same depth as the seed (*c.* 7-8 cm). This will only be applicable for crops sown in the spring (apart from grass reseeds in autumn). When urea is used care needs to be taken to ensure the fertiliser is not placed so close to the seed that germination is inhibited. The NH_3 emissions from placement of fertilisers have been estimated to be negligible (assuming that N supply is dimensioned correctly). Deep placement of fertiliser granules is a common technology and has been used for many years in Finland (Aura 1967).

Incorporation of fertiliser-N prior to rice planting, or delaying application until panicle initiation, have been shown to reduce NH_3 emissions from rice fields (Humphreys et al. 1988). These are already standard practices in the USA (Bacon et al. 1988). Freney et al. (1988) warned that measures to reduce NH_3 emissions from rice cultivation may not reduce total N losses in soils with large nitrification or denitrification rates.

Reducing fertiliser-N applications to grassland may be effective in reducing NH_3 emissions per unit area. However this will reduce milk yield and liveweight gain, unless extra forage is made available. The provision of such extra forages may increase NH_3 losses, either at grazing or in buildings, by increasing excretal returns.

It should be noted that few of these changes have so far been applied by countries as measures to limit NH_3 emissions, and further work would be required to provide a detailed evaluation of all these possibilities.

3.3.2 Nitrous Oxide

Since emission of N_2O is a consequence of nitrification as well as denitrification (e.g. Klemendtsson et al.1988; Arcara et al. 1990), the greatest potential for reducing N_2O emissions will come from reducing mineral N (NH_4^+ and NO_3^-) concentrations in soils. This may be done most simply by reducing inputs of fertiliser-N. As stated in section 3.2.1, such a measure will have a significant cost if it reduces crop yields. However some scope may remain for better matching of N-fertiliser to crop requirement. This may be done by avoiding N applications in excess of the economic optimum. Such applications are sometimes made as an insurance against under-fertilising. To reduce the likelihood of excessive N fertilisation care should be taken, when deciding N-fertiliser applications, to make full allowance for N available in the soil from previous crop residues, organic manure applications and mineralisation of soil organic matter. Careful timing of fertiliser-N application may also reduce N_2O emissions. Crops and soil microbes may be regarded as being in direct competition for the mineral N in soil (Recous et al. 1988). If large amounts of N-fertiliser are applied to the soil at times when plant growth is limited, and when soil moisture and temperature are favourable, denitrifying and nitrifying bacteria may use the mineral N as a substrate. However if the N-fertiliser is applied to an actively-growing crop, much of it may be taken up quickly by the crop, and removed from sites of bacterial activity. Timing of fertiliser-N application to coincide with crop demand is already recommended in many countries to reduce NO_3^- leaching. Preliminary results from a UK study (S Ellis pers. comm.) suggest this strategy may also be effective in reducing N_2O emissions.

The effect of fertiliser type on the emission of N_2O is inconclusive, and is therefore not included in the IPCC methodology. However, some studies suggest, that fertiliser - induced emissions of N_2O may be greater with the use of anhydrous ammonia and organic N fertilisers (Bouwman, 1996).

Timing the incorporation of crop residues, to avoid incorporating when soils are poorly aerated, may also give some reduction in N_2O emissions (Flessa and Beese 1995). The relative importance of nitrification and denitrification need to be defined, if correct soil management is to be applied. Mitigation options to reduce N_2O emissions from agricultural soils are further discussed by Mosier et al. (1996).

3.3.3 Nitric Oxide

Since, in temperate climates, NO emissions are considered to be predominantly a consequence of nitrification, the use of urea fertiliser may produce larger NO emissions than equivalent amounts of N applied as AN. While at present there is insufficient data to discriminate between fertiliser-N sources (Skiba et al. 1997), the substitution of AN for urea to reduce NH_3 emissions, may also give some reduction in NO emissions.

Currently the only other possible option is to ensure that applications of N-fertiliser are no greater than is needed for optimum crop yield, by making full allowance for the N supplied by crop residues, organic manures, previous N-fertiliser applications, and mineralisation of soil N. Fertiliser application should also be timed to match crop demand.

3.3.4 Volatile Organic Compounds

No potential controls have been proposed for VOC emissions from fertilised crops.

4 SIMPLER METHODOLOGY

4.1 Ammonia

Noting the interdependence of direct fertiliser emissions and subsequent emissions from foliage and decomposing material of fertilised vegetation, the emissions are treated here as a single integrated term. These are estimated as % losses of the fertiliser-N use for each of the main fertiliser categories. In the simple methodology the % N emissions factors are taken to be the same for all countries. Soil type and climate are expected to affect emissions and an approach is given in the detailed methodology (section 5) to account for this.
The emission factors for the simple methodology are provided in Table 4.1. These are based largely on the estimates of Asman (1992), ECETOC (1994) and Sutton et al. (1995b). The combined fertiliser-plant emission factors are smaller than the totals of ECETOC (1994), since in the original estimates of ECETOC their emissions factors referred to just fertiliser losses, while they provided an additional emission from indirect foliar emissions (not shown in Table 4.1). In contrast the estimates here are larger than the estimates of Sutton et al. (1995b).

It should be noted that the estimates published by Buijsman et al. (1987) are now considered to be out of date and overestimate NH_3 emissions.

To calculate NH_3 emissions from fertilised cultures in a country, the use of each fertiliser type (expressed as mass of fertiliser-N used per year), is multiplied by the appropriate emission factor, and the emissions for the different fertiliser types summed. A simple spreadsheet for this calculation is provided in the detailed methodology (see section 5). Emissions of NH_3-N from fertilisers applied to grass cut for hay or silage may be calculated using the same factors as for arable and other crops. Separate emission factors are used for NH_3 emission from grazed grassland.

Table 4.1: Simpler methodology estimates of total NH_3 emissions from cultures due to fertiliser volatilisation, foliar emissions and decomposing vegetation (second column). The estimates are compared with other literature values. Values are kg NH_3-N volatilized per kg of N in fertilisers applied (100* NH_3-N/fertiliser N).

Fertiliser type	**Present simpler methodology to apply**	**Asman (1992) (Europe)**	**ECETOC (1994) Group II European countries)**	**Sutton et al. (1995) (UK)**
Estimates from	fertiliser and plants	fertiliser	fertiliser	fertiliser and plants
Ammonium sulphate	.08	.08	.10	-
Ammonium nitrate	.02	.02	.02	.01
Calcium ammonium nitrate	.02	.02	.02	.01
Anhydrous ammonia	.04	.01	.04	-
Urea	.15	.15	.15	.10
Combined ammonium phosphates (generally di-ammonium phosphate)		.04	.05	-
Mono-ammonium phosphate	.02			
Di-ammonium phosphate	.05			
Other complex NK, NPK fert	.02	.02.5-.04	.02	.025
Nitrogen solution (mixed urea and ammonium nitrate)	.08	-	.08	-

To estimate NH_3 emissions from grazed grassland, Jarvis and Bussink (1990) proposed a function of the form

$$V = a\,b^N . \qquad (1)$$

where V = NH_3-N emission
N = fertiliser-N application (kg ha^{-1})

This function was subsequently updated and used to calculate grazing emissions for the UK (Pain et al. 1997),where:

$$V = -2.27 + 0.0683N \qquad (2)$$

To estimate NH_3 emissions from grazed grassland, b will be equal to the average annual fertiliser-N application to grazed grass. This approach estimates both emissions from fertiliser-N, and subsequent emissions from excreta deposited during grazing. In the studies from which these data were derived, AN or calcium ammonium nitrate were the sources of fertiliser-N

Thus the above equation may underestimate losses from grassland when urea is applied. No distinction is made between emissions from cattle and sheep. However in general less fertiliser-N is applied to grass grazed by sheep, usually <100 kg ha^{-1} yr^{1}, and so in practice emissions from sheep grazing are not likely to be greatly overestimated.

Equation (2) was derived almost entirely from measurements of NH_3 emissions in N-W Europe. The relationship may not give accurate estimates of emissions from grazing in drier, or warmer climates.

4.2 Nitrous Oxide

The methodology used here follows the approach taken by IPCC/OECD (1997) in order to maintain consistency between the different emission inventories. Emissions arising from manure application are dealt with in SNAP Code 100500, Manure Management. The IPCC detailed methodology requires the following input data:

1. Total synthetic fertiliser-N applied

2. Total N content of fertilised crop residues, biological N_2 fixation and excreta from grazing animals returned to the soil.

3. Area of organic soils (histosols) cultivated in the country.

4. Estimate of atmospheric emission of NH_3 and NO_x.

5. Estimate of N lost from soils by leaching and runoff.

These input data can be estimated from FAO data (see IPCC/OECD 1997), see Table 4.2.

Table 4.2: Summary of IPCC source categories (IPCC Guidelines for National Greenhouse Gas Inventories, Volume 2: Workbook, 1997) to be reported as CORINAIR sub-sectors for agriculture.

CORINAIR SUB-SECTOR (snap code)	**IPCC N_2O SOURCE (IPCC WORKBOOK WORKSHEET**
Cultures with/without fertilisers (100100.100200)	- Direct soil emission due to N-inputs excluding manure (4-5, sheet 1, excluding animal waste Faw) - Direct soil emissions due to histosol cultivation (4-5, sheet 2) - Direct soil emissions from grazing animals; pasture, range & paddock (4-5, sheet 3) - Indirect emissions due NH_3 and NO_x emissions from synthetic fertiliser use and grazing animals (4-5, sheet 4, excluding animal waste used as fertiliser) - Indirect emissions due N leaching/runoff from synthetic fertiliser use and grazing animals (4-5, sheet 5, excluding animal waste used as fertiliser
Manure Management (100500)	- Manure management: 6 waste management systems (4-1, sheet 2, excluding pastue, range & paddock) - Direct soil emissions due to manure N-inputs excluding grazing animals (4-5, sheet 1, row for animal waste Faw only) - Indirect emissions due NH_3 and NO_x emissions from animal waste excluding grazing animals (4-5, sheet 4, animal waste used as fertiliser only) - Indirect emissions due N leaching/runoff from animal waste excluding grazing animals (4-5, sheet 5, animal waste used as fertiliser only)

The default emission factors for the above are given in Table 4.3. More detail may be obtained from IPCC Worksheet 4-5, sheets 1-5.

The N_2O emission may be calculated as :

FN_2O-N = 0.0125 * 'net' N inputs 0.02* 'net' N inputs from grazing + 0.01 * (NH_3 + NOx-N emissions to atmosphere + 0.025 * N (leached and/or run off) (3)
where units are generally expressed as kg N ha^{-1} yr^{-1}.

In the case of organic soils (histosols) an additional emission of 5 kg N ha^{-1} yr^{-1} is added (Table 4.3).

The IPCC guidelines estimate direct soil emissions as a fraction of N inputs to soils, excluding NH_3 emissions. This 'net' N input is calculated for N-fertilisers as :

'net' N input (kg N ha^{-1}) = fertiliser-N input (kg N ha^{-1}) * 0.9 (4)

To calculate the subsequent N_2O emission, this 'net' N input is multiplied by the emission factor 0.0125 kg N_2O-N per kg 'net' N input.

FN_2O (fertiliser) = 'net' fertiliser-N input (kg N ha^{-1}) * 0.0125 N_2O-N kg N input^{-1} (5)

For excretal N deposited during grazing, the 'net' N input is calculated as :

'net' input (kg N ha^{-1}) = Excretal-N input (kg N ha^{-1}) * 0.8 (6)

An estimate of excretal-N deposited during grazing is given for each major livestock class in SNAP Code 100500, Manure Management.

To calculate N_2O emissions from N deposited during grazing, the 'net' N input is multiplied by the emission factor 0.02 kg N_2O-N per kg 'net' grazing N input

FN_2O (grazing) = 'net' grazing N input (kg N ha^{-1}) * 0.02 kg N_2O-N kg N input^{-1} .. (7)

Table 4.3: Default emission factors for N_2O emissions from cultures with fertiliser (IPCC/OECD, 1995)

Source of N_2O	**Emissions Factor**
Direct soil emissions	
N inputs (fertiliser,biological N fixation and crop residues)[#]	0.0125 kg N_2O-N/kg 'net' N input^{-1}
cultivation of histosols	5 kg N_2O-N ha^{-1} yr^{-1}
Excretal N deposited during grazing†	0.02 kg N_2O-N kg 'net' N input^{-1}
Indirect emissions	
Emission of NH_3 and NO_x	0.010 kg N_2O-Nkg NH_3-N and NO_x-N emitted^{-1}
N leaching and runoff	0.025 kg N_2O/kg N leached or lost by run off^{-1}

[#] Manure N inputs are dealt with in SNAP Code 100500, Manure Management

The IPCC estimates were developed for wide area estimates to be integrated up to the global scale. As such a number of simplifications had to be made to suit the availability of data. A key approximation on this scale is that atmospheric N deposition equated to atmospheric emissions of NH_3 and NO_x. On a European scale, and particularly within the Convention on Long Range Transboundary Air Pollution, this might be considered an unacceptable simplification, given the different locations of source and sink areas. In this case atmospheric N deposition estimates should be taken from national estimates or EMEP (e.g. Barrett & Berge 1996).

It should be recognised that although N_2O emissions as a function of atmospheric N inputs will be significant, for fertilised agricultural land these will generally be small compared with emissions resulting from fertiliser-N inputs. This term should nevertheless be included to maintain consistency across different land-uses.

4.3 Nitric Oxide

A simple methodology similar to that described by the IPCC for N_2O flux estimates is applied for estimating NO emissions, i.e. emissions that arise directly as a result of N-fertiliser application. To estimate NO emissions, the total usage of mineral N-fertiliser in the country is required. Emissions of NO-N are calculated as 0.3% of total fertiliser-N applied (Skiba et al. 1997). This 0.3% value was derived from a much smaller data set compared to the equivalent value used for the N_2O fertiliser induced emission calculation. The certainty in the 0.3% value can therefore be greatly increased by further rigorous intensive measurements.

4.4 VOCs

An approach for estimating the emissions from grassland and crop is given in Chapter 110400, Natural Grassland. The same approach may be adopted here. It is important to use the same land use data as for other emitted species, and to avoid double counting. If the method given in Chapter 110400 is used, emissions from fertilised agricultural land must be reported under Chapter 100100.

5 DETAILED METHODOLOGY

5.1 Ammonia

5.1.1 Emissions from crops and cut grassland

To provide a more detailed methodology it is desirable to distinguish between the different climates and soil types for different countries. The justification for this is well established, as crop emissions are known to be larger in warmer climates (Fenn and Hossner 1985), while soil emissions (direct fertiliser losses) generally increase at higher soil pH (e.g. Whitehead and Raistrick 1990). Given the need to generalize, only a broad scale approach is possible to apply these known differences in inventories. A first attempt has been applied by ECETOC (1994), and is used as the basis for the present classification. Countries are categorized into 3 types:

Group I Warm temperate countries with a large proportion of calcareous soils (e.g. Greece, Spain).

Group II Temperate and warm-temperate countries with some calcareous soils (or managed with soil pH >7), but with large areas of acidic soils (e.g. Italy, France, UK, Eire, Portugal, Belgium, Netherlands, Luxembourg.

Group III Temperate and cool-temperate countries with largely acidic soils (e.g. Nordic countries, Germany, Switzerland, Austria).

The countries listed in brackets are as assigned by ECETOC (1994), which restricted its coverage to western Europe. Other UNECE countries may be added to this classification. Here the main extension would be that countries with subtropical and continental climates (eg Eastern Mediterranean, Southern Steppe) would be expected to fall into Group I.

In respect of urea, a greater NH_3 loss on calcareous soils may not be justified. While NH_3 losses from AS and AN have been found to increase markedly with increasing pH (e.g. Whitehead and Raistrick 1990), NH_3 loss from urea is less dependent on initial soil pH, since hydrolysis of urea will increase pH immediately around the fertiliser granule to *c.* 9.2 (Fenn 1988). Moreover reaction with calcium ions reduces the volatilisation potential of $(NH_4)_2CO_3$ produced by urea hydrolysis (Fenn and Hossner 1985). In contrast to other N-fertilisers, NH_3 loss from urea did not increase consistently with pH, and was not greater on a calcareous soil (Whitehead and Raistrick 1990). This was considered due to differences in cation exchange capacity (CEC). Whitehead and Raistrick (1993) also found losses of NH_3 from cattle urine were no greater on calcareous than on non-calcareous soils. The best correlation with NH_3 loss was with CEC. Gezgin and Bayrakli (1995) measured NH_3 losses from urea, AS and AN on calcareous soils in Turkey. Losses from AS (*c*. 16%) and AN (*c.* 5%) were greater than those measured on non-calcareous soils by Somner and Jensen (1994), which were <5% and <2% respectively. However losses from urea at *c*. 8% were less than those measured by Sommer and Jensen (1994). Nevertheless a greater emission factor for urea in Group 1 countries is justified by the greater temperatures. The large proportion of calcareous soils will however increase NH_3 losses from AS (Fleisher et al. 1987).

Values of emission estimates for the more detailed methodology are provided in Table 5.1. A simple spreadsheet is provided for calculating culture NH_3 emissions in Table 9.1.

5.1.2 Emissions from grazed grassland

Van der Weerden and Jarvis (1997) reviewed data from field measurements of NH_3 loss following application of N fertilisers to grassland and arable land. They concluded that NH_3 losses from urea are greater by a factor of 2 on grassland. This greater potential for NH_3 loss had been attributed to greater urease activity in grassland soils by O'Toole and Morgan (1985). This will give the following NH_3 losses (kg NH_3-N kg fertiliser -N^{-1} applied) for urea:

Arable land	0.16 (Zone I), 0.12 (Zones II and III);
Grassland	0.30 (Zone I), 0.25 (Zones II and III).

These coefficients are for estimating emissions only from grass that is cut. The method for calculating NH_3 emissions from grazed grassland is given in Section 4.1

Table 5.1: Detailed methodology estimates of total NH_3 emissions from cultures due to N fertiliser volatilisation, foliar emissions and decomposing vegetation. Values are kg NH_3-N volatilised per kg fertiliser-N applied (100 * NH_3-N/fertiliser-N^{-1})

Fertiliser type	Group I	Group II	Group III
Ammonium sulphate	.15	.10	.05
Ammonium nitrate	.03	.02	.01
Calcium ammonium nitrate	.03	.02	.01
Anhydrous ammonia	.04	.04	.04
Urea	.20	.15	.15
Combined ammonium phosphates (generally di-ammonium phosphate)	.05	.05	.05
Other complex NK, NPK fertilisers	.03	.02	.01
Nitrogen solutions (mixed urea and ammonium nitrate)	.08	.08	.08

5.1.3 Emissions from Rice Fields

Losses of NH_3 following applications of N-fertiliser to flooded rice soils are considered to be potentially greater than from other cropping systems (Fenn and Hossner 1985). Vlek and Crasswell (1979) measured NH_3 losses of up to 50% of urea-N applied, and up to 60% of AS-N. More recent studies of NH_3 emissions following urea application to flooded soils, have measured losses in the range *c*. 8-56% of urea-N, with a median loss of 30%. Many of these studies used micrometeorological techniques. Patel et al. (1989), in a review, concluded that earlier studies using chambers, had overestimated NH_3 losses in the field.

A loss coefficient of 0.30 NH_3-kg N kg Urea-N applied^{-1} should therefore be used to calculate NH_3 emissions from flooded rice fields.

As noted above, application of fertiliser-N to flooded rice fields is not standard practice in all countries. When application is either to the soil before flooding, or when the crop is at panicle initiation, losses will be significantly less. In such cases, emissions should be calculated using the coefficients given in Table 4.2.

No data was found on NH_3 losses following AN application to rice fields. In the absence of more specific data, emissions should be calculated from Table 4.2. Fillerey and De Datta (1987), found NH_3-N losses of 37% from AS applied to flooded rice fields. We propose using the same loss coefficients for AS as for urea, when estimating NH_3 losses from rice field.

5.2 Nitrous Oxide

No more detailed methodology is proposed for estimating emissions of N_2O. However countries may use their own estimates for any step in the IPCC method if this will increase precision. In particular countries are encouraged to estimate NH_3 losses using the methods described in this chapter, rather than the IPCC default values.

5.3 Nitric Oxide

A more detailed methodology, based on the soil temperature and the land use type has been developed by Williams et al. (1992).

$$F_{NO} = A * \exp(0.071 * T_s) \tag{8}$$

where

F_{NO} = NO flux, (ng Nm^{-2} s^{-1})
Ts = Soil temperature, (degrees Celsius)
A = Experimentally derived constant for the land use types of grasslands and pasture, forests and urban trees, and the individual agricultural categories

To improve this approach, N input and soil moisture contents (Meixner, 1994) need to be included in the equation. However at this stage, there is no sufficiently robust analysis to use as a basis for a more detailed methodology. This methodology produces much greater estimates of NO emission than are given by the simple methodology, and the authors conclude it is not appropriate to use at this stage. Details of this method are given in chapter 110300.

5.4 VOCs

See section 4.4.

6 RELEVANT ACTIVITY STATISTICS

Information is required on the annual consumption of major N- fertiliser types by each country. This may be obtained from IFA (1992) as well as from national agricultural censuses. Information may also be required on the amounts and N concentrations of crop residues returned to the soil. This may be obtained from statistics on crop production. The amounts of N deposited by animals while grazing are given in SNAP Code 100500, Manure Management. The area of organic soils (histosols) under cultivation is also useful. Finally information is needed on emissions of NH_3 and NO_x.

Where spatially disaggregated inventories of fertilised culture emissions are required (section 12), information on the spatial distribution of different crop types and average N-fertiliser inputs to each crop type may be used. In the absence of data on the use of different fertilisers for crop types, the average N-fertiliser inputs to crops may be combined with the average NH_3 emission factor for a country estimated according to Tables 9.1, 9.2 and 9.3 as: Total $NH_3/N_2O/NO$ emission/total N-fertiliser consumption. All the activity statistics required are summarised in Table 9.4.

7 POINT SOURCE CRITERIA

Ammonia, N_2O, NO and VOC emissions from cultures with fertilisers should be treated as area sources.

8 EMISSION FACTORS, QUALITY CODES AND REFERENCES

The emission factors for NH_3, N_2O and NO losses from cultures with fertilisers are treated as kg of N per kg N applied as fertiliser or, in the case of crop residues incorporated into the soil, and excretal-N deposited during grazing, of the N returned in these forms. Full details of calculations are provided in sections 5 and 6.

9 SPECIES PROFILES

9.1 VOCs

As with forest NMVOC emissions, biogenic emissions from grasslands consist of a wide variety of species, including isoprene, monoterpenes, (alpha-pinene, limonene, etc.), and ‘other’ VOC. The ‘other’ VOC (OVOC) species consist of a large number of oxygenated compounds (alcohols, aldehydes, etc.), and have proven difficult to quantify in atmospheric samples. Progress in quantification of OVOC from European vegetation has been made recently (König et al. 1996) although many more measurement data will be required before reliable attempts to inventory specific OVOC can be made.

Table 9.1: Spreadsheet for calculating NH_3 emissions from cultures with fertilisers according to either the simpler or more detailed methodologies

Group of country (for detailed methodology)			
Column	A	B	C
Fertiliser type	N emissions (from Table 1 or 2) kg NH_3-N kg N applied yr^{-1}	Fertiliser use kg N yr^{-1} (see section 6)	Ammonia emissions kg NH_3 yr^{-1} (A*B*17/14)
Ammonium sulphate			
Ammonium nitrate			
Calcium ammonium nitrate			
Anhydrous ammonia			
Urea			
Mono-ammonium phosphate			
Di-ammonium phosphate			
Other complex NK, NPK fertilisers			
Nitrogen solution (mixed urea and ammonium nitrate)			
Legumes (see section 5)			
Total ammonia emissions in kg NH_3 per year			

Table 9.2: Spreadsheet for calculating nitrous oxide emissions from cultures with fertilisers according to either the simpler or more detailed methodologies

	A	B	
N input	N emission kg N_2O-N kg ‘net’ N applied^{-1}	N input, kg N yr^{-1}	N_2O emission kg N_2O yr^{-1} (A * B * 44/28)
Fertiliser- N	0.0125		
Crop residues N	0.0125		
Excretal- N deposited during grazing	0.020		
Emission of NH_3	0.010		
Emission of NO_x	0.010		
N lost by leaching or runoff	0.025		
	kg ha^{-1}	Area (ha)	
Cultivation of histosols	5		

Table 9.3: Spreadsheet for calculating nitric oxide emissions from cultures with fertilisers according to the simple methodology

	A	B	
N input	N emission kg NO-N kg N applied^{-1}	Fertiliser-N use kg N yr^{-1}	Nitric Oxide emission kg NO-N yr^{-1}
			(A * B * 30/14)
Fertiliser-N	0.003		

This spreadsheet is much less detailed because there are few data on NO emissions from soils.

Table 9.4: Summary of activity statistics which may be required for the simple and detailed methodologies.

Activity Statistic		Source
Synthetic Fertiliser-N use	By type of fertiliser for arable and grassland	IFA (1992)
Amount and N concentration of fertilised crop residues	By crop type	FAO, See IPCC/OECD (1997)
Amount and N concentration of animal excreta deposited during grazing	By Livestock type	SNAP Code 100500, Manure Management
Area of cultivated histosols		FAO, See IPCC/OECD (1997)
Atmospheric emissions of NH_3 and NO_x		ETCAE, (1997)
N lost from soils by leaching and runoff		FAO, See IPCC/OECD, (1997)

10 UNCERTAINTY ESTIMATES

10.1 Ammonia

Although the processes governing the emission of NH_3 from fertilisers and crops are reasonably well understood, the interactions of many biological, chemical and environmental factors make quantitative estimates of emission rather uncertain. The main uncertainty lies in the generalization of emissions factors, rather than the areas of crops under cultivation which is probably accurate in most countries to better than +/- 10%. For example, the ranges of uncertainty for Group II emission factors in Table 4.1 are probably wider than the figures given as emission factors for Groups I and III, which are included in order to avoid bias between countries with different conditions. Overall emissions are probably no better than +/- 50% (a factor of 3).

10.2 Nitrous Oxide

The processes controlling the emission of N_2O from soils are reasonably well understood. Their interactions, and the partitioning between nitrification and denitrification, and hence estimates of emission, have not yet been accurately modelled. Application of fertiliser-N, in most countries, can probably be estimated to within $\pm$10%. However the magnitude of crop residues, and their N contents are only likely to be accurate to within $\pm$25%. Wet deposition of N may be estimated to $\pm$ 20%, but dry deposition of NH_3 to no more than $\pm$50% (UKRGIAN 1994). Most importantly fertiliser-N induced N_2O emissions can be estimated with a certainty of a factor of 9 (range 0.25 - 2.25% of N inputs, Bouwman 1996). The importance of winter time N_2O emissions caused by freezing and thawing has only recently been discovered and has not been included in any of the estimates. This will create a further large uncertainty.There will be even greater uncertainty with regard to indirect emissions of N_2O since there are fewer measurements available of N_2O formation from atmospheric deposition, or leached N.

10.3 Nitric Oxide

Much less information is available on factors determining losses of NO from soils. While application of fertiliser-N may be estimated with an accuracy of $\pm$10%, other factors such as returns of N in crop residues and soil N contents may be estimated to within $\pm$25%. However, the greatest uncertainty is over emission factors. Using data from essentially the same body of published work, Yienger and Levy (1995) and Skiba et al. (1997) arrived at mean emission factors almost an order of magnitude different, suggesting an uncertainty of a factor of 10.

10.4 Volatile Organic Compounds

Biogenic VOC emissions for the UK were summarized by Hewitt and Street (1992). These ranged from 38-211 kt yr^{-1} total VOCs. Emissions from woodlands were estimated to be 72% of total biogenic emissions by Anastasi et al. (1991). Thus between *c.* 10 and 59 kt yr^{-1} appear to be of agricultural origin. This compares with the CORINAIR 94 estimate of only 2 kt yr^{-1} for SNAP Code 100100 or <2% of emission from agriculture and forestry. Thus the range of emissions may be uncertain by a factor of 30. However the estimate for agriculture by Anastasi et al. (1991) was recognised as likely to be too large.

Hewitt and Street (1992) concluded that only *c.* 700 plant species, mainly from N. America, had been investigated as isoprene or monoterprene emitters. Few of these were agricultural crops, and quantitative data was available for only a few species. Many measurements had been made at temperatures higher than those prevailing in N and W Europe.

11 WEAKEST ASPECTS/PRIORITY AREAS FOR IMPROVEMENT IN CURRENT METHODOLOGY

11.1 Ammonia

A major criticism of the present estimates is their reliance on simple fixed (%) emission factors, given in relation to amounts of N applied. A first attempt to account for broad scale differences between countries (based on climate and soil type) has been included here (detailed methodology) but it is very much an empirical interpretation of the available data. More work needs to be done in the development of mechanistic process-based models for predicting NH_3 emissions from N-fertilisers and the foliage of fertilised crops, which take into account the known physicochemical equilibria as well as interactions with biological processes to predict net fluxes. It is well established that NH_3 may be exchanged with the soil surface and with leaves via stomata and cuticular absorption/desorption as well as with decomposing leaves, and future work needs to quantify the interactions and exchange cycles between these different components. Studies of NH_3 emissions from grazed grassland have largely been carried out in NW-Europe. Data is needed on emissions in warmer and drier climates.

The current estimates are limited to net emission of NH_3 over the year, and as such integrate both periods of emission from cultures and deposition to them on both diurnal and seasonal scales. Further work is needed in quantifying the temporal variability in emissions as well as the integration of emitting surfaces and depositing surfaces for development of atmospheric models.

11.2 Nitrous Oxide

Current estimates of N_2O emissions are also limited by the use of fixed emission factors. More work needs to be done in the development of process-based models that will allow greater discrimination to be made between soils with different moisture regimes, and between areas of different climate. Emissions of N_2O per kg fertiliser-N applied, have been shown to be greater from grasslands, particularly from grazed grasslands, than from arable crops (Velthof et al. 1996). This is caused by a combination of: soil compaction caused by the grazing animals; localised very high N and C inputs from animal excreta; and in cooler climates grasslands tend to be located on the wetter soils, less suitable for arable crop production. In addition the question of differentiating between fertiliser-N sources needs to be addressed. Emissions from organic N-fertilisers (Bouwman 1996; McTaggart et al. 1997) and anhydrous NH_3 have been shown to be larger than from solid mineral-N fertilisers. Evidence for significantly different emissions according to N source needs to be evaluated.

Estimates of indirect emissions of N_2O are dependent on accurate estimates of NH_3 and N_2O emissions and N leaching and runoff. As long as uncertainties in these estimates are large, then so too will be estimates of indirect N_2O emissions. Losses of N by leaching and run off are not necessarily directly proportional to N inputs, but depend on crop, soil, climate etc. It would be useful therefore to be more specific in this regard.

11.3 Nitric oxide

Reliance on a simple fixed (%) emission factor in relation to the amount of N-fertiliser applied is the major weakness. No account is taken of potentially large differences between climatic regimes. Nor is any account taken of the potential contribution from non-fertiliser sources of N, and from the mineralisation (and hence nitrification) of N already present in soils. There are not enough data available to discuss the effect of grazing on NO emissions, but the localised very high N and C inputs caused by animal excreta are likely to stimulate NO production.

11.4 Volatile Organic Compounds

Lack of measurements of VOC emissions from agricultural vegetation is a considerable weakness. It is unknown whether emissions are related to fertiliser-N inputs.

12 SPATIAL DISAGGREGATION CRITERIA FOR AREA SOURCES

12.1 Ammonia

The simplest approach to spatially disaggregate the emissions from cultures with fertilisers is to scale these by the distribution of total arable and fertilised grassland. In a more detailed approach census data on the distribution of different crop types may be combined with characteristic fertiliser inputs to each crop type, together with the overall fertiliser emissions factor estimated from Table 4.2. Where the average N-fertiliser application to crops is derived from similar national data as the fertiliser consumption, there should be a reasonable agreement between the mapped and national total emission. However, caution is required and spatially disaggregated estimates may need to be corrected.

Caution is also required to account for the possible double counting of fertiliser/foliar emissions from grazed grassland, noted in the detailed methodology (section 5). If this effect is to be treated in spatially disaggregating emissions, it may be considered that the emissions from grazed grass, where this is supplied, fertilised with AN or complex fertilisers, are already included in the grazing emissions. In this case, land-use maps of grazed grassland would be required, in a similar way to the distribution to crop types, but here a reduced emission factor applied to account for only emissions from ‘high emissions’ fertilisers. This reduced overall emissions factor may be found by completing a version of Table 4.2 for grazed grassland, not including emissions from AN, calcium AN or other complex fertilisers. Dividing the total NH_3 emission by total N-fertiliser use, multiplied by 17/14, provides an ‘average’ % N volatilised for grazed grassland. This can then be applied with the mapped distribution of grazed grassland. Where only the distribution of total grassland is available, estimates would need to be made of the fraction that is grazed, while account of the temporal overlap of grazing and culture emission should also be taken.

12.2 Nitrous Oxide

Direct emissions may be spatially disaggregated using census data on the distribution of different crops together with mean fertiliser-N inputs to those crops. Data on the distribution of organic soils (histosols) may also be included to improve spatial disaggregation.

Information on climatic variations (temperature and rainfall) within the country should be included. At present no standard model to do this is available. Dividing the country/region into 3 to 4 climatic groups and slightly modifying the IPCC methodology, by increasing and reducing the various emission factors for the individual regions may be a way forward.

Indirect emissions may also be spatially disaggregated if spatial data is available for N deposition and also for N leaching and runoff.

12.3 Nitric Oxide

Emissions due to N-fertiliser application may be spatially disaggregated using census data on the distribution of different crops, together with mean fertiliser-N inputs to those crops, and climatic information as outlined in section 12.2.

12.4 Volatile Organic Compounds

In the absence of specific data for VOC emissions from different agricultural crops, there appears to be little scope at present for spatially disaggregating VOC emissions.

13 TEMPORAL DISAGGREGATION CRITERIA

13.1 Ammonia

As noted in section 11, little information is available to generalize on temporally disaggregating NH_3 emissions from fertilisers and crops. Most of the direct emission from N-fertiliser occurs within a month of application and, for some countries, agricultural statistics may be available on the timing of these applications. Further crop emissions may occur particularly during senescence of crop plants, and may account for *c.* 1-3 kg NH_3- N per ha emission. A major uncertainty with fertiliser, foliar, grazing and decomposing vegetation emissions, is that losses are expected to vary greatly from year to year depending on agricultural and environmental conditions.

13.2 Nitrous Oxide

Most of the direct emission from fertiliser-N occurs with a few weeks of application (Bouwman 1996), and statistics on the timing of fertiliser-N application should be available, for some countries at least. Some data may also be available on the timing of crop residue incorporation. However, until process-based models have been developed and validated it will not be possible to take account of fluxes of N_2O emission that take place when soil mineral N concentrations, soil water regimes and soil temperature combine to produce favourable conditions for denitrification.

Data will be available, for some countries at least on the temporal variation in N deposition and N leaching and runoff.

As for NH_3, losses may vary greatly from year to year, depending upon weather conditions.

13.3 Nitric Oxide

Losses of NO take place mainly as a consequence of nitrification. Peaks in NO emission are therefore likely following application of NH_4^+-based N-fertilisers, incorporation of crop residues and tillage of soils. Data on all these should be available, for some countries at least. At present, however there are insufficient data on NO emissions to quantify these effects. Ultimately, as the mechanisms of NO production become better understood, climatic data may also be utilised to assess when soil and weather conditions are favourable for nitrification, and hence NO production. In common with NH_3 and N_2O, emissions may vary greatly from year to year, depending upon weather conditions.

13.4 Volatile Organic Compounds

Emissions of VOCs are likely to differ according to crop type, crop growth stage, soil type, cultivation and weather conditions. Some temporal disaggregation may be possible, if seasonal variations in emissions by non-agricultural plants, can be assumed to be valid for fertilised crops.

14 ADDITIONAL COMMENTS

Where more detailed methodologies than those described here are used by countries, a detailed description should be given of the methodology used, and comparison made with the results of the methodology described here.

15 SUPPLEMENTARY DOCUMENTS

The main supplementary documentation required for applying the estimates in this chapter are details of national N-fertiliser consumption and areas of fertilised grassland that are grazed. In addition estimates of the amounts and N concentrations of crop residues returned to the soil, together with the area of cultivated organic soils (histosols) will be required. The approximate timing of soil cultivation, including crop residue incorporation will also be useful. Finally information on deposition of NH_3 and NO_x, together with estimates of N leaching and runoff are needed. Where disaggregated estimates are to be made, details on N application rates to crops and spatially disaggregated crop distribution are needed.

16 VERIFICATION PROCEDURES

There are no direct methods to evaluate total inventory estimates of NH_3 emissions from croplands, and verification is dependent on laboratory and micrometeorological field studies of emissions from example situations. In particular, many studies have focused on laboratory measurements and there is a need to provide long-term field measurements using micrometeorological techniques to estimates NH_3 fluxes over a range of crop types in different climates.

Emissions of N_2O, NO and VOCs cannot be verified except by field studies of emissions from example situations. There is a need to obtain long-term field measurements to estimate fluxes over a range of crop types and climates.

17 REFERENCES

Anastasi, C., Hopkinson, L., Simpson, V. J. 1991. Natural Hydrocarbon emissions in the United kingdom. Atmospheric Environment **25A**, 1403-1408.

Arcara, P. G., Gregori, E., Miclaus, N., Cornellini, F. 1990. Denitrificazione e perdite di azoto (N_2O) da suoli coltivati a frumentofertiliszati con urea, nitrato ammonico e letame. Rivista di Agronomia **24 (1),** 19-27.

Asman, W. A. H. 1992 Ammonia emission in Europe: updated emission and emission variations. RIVM report 228471008. RIVM, Bilthoven, Netherlands.

Aura, E. 1967. Effect of the placement of fertiliser on the development of spring wheat. Journal of the Scientific Agricultural Society of Finland **39**, 148-155.

Bacon, P. E., Hoult, E. H., Lewin, L. G., McGarity, J. W. 1988. Ammonia volatilization from drill sown rice bays. Fertiliser Research **16 (3)**,257-272.

Barrett, K., Berge, E. 1996. Transboundary air pollution in Europe. Part 1: Estimated dispersion of acidifying agents and of near surface ozone. EMEP/MSC-W Status report 1996. Norwegian Meteorogological Institute, Blindern, Oslo, Norway.

Bouwman, A. F. 1996. Direct emission of nitrous oxide from agricultural soils. Nutrient Cycling in Agroecosystems **46**, 53-70.

Buijsman, E., Maas, H. F. M., Asman, W. A. H. 1987. Anthropogenic NH_3 emissions in Europe. Atmospheric Environment **21**, 1009-1022.

Bussink, D. W. 1992. Ammonia volatilization from grassland receiving nitrogen fertiliser and rotationally grazed by dairy cattle. Fertiliser Research **33**, 257-265.

ECETOC 1994. Ammonia emissions to air in western Europe. Technical Report 62. European Centre for Ecotoxicology and Toxicology of Chemicals, Avenue E Van Nieuwenhuyse 4, Brussels.

EEA 1996. Atmospheric Emission Inventory guidebook. (Ed G. McInnes). European Environment Agency, Kongens Nytorv 6, Copenhagen.

ETCAE 1997. CORINAIR 94 Summary Report - European Emission Inventory for Air Pollutants. European Environment Agency, Kongens Nytorv 6, Copenhagen.

Fenn, L. B. 1988. Effects of initial soil calcium content on ammonia losses from surface-applied urea and calcium-urea. Fertiliser Research **16**, 207-216.

Fenn, L. B., Hossner, L. R. 1985. Ammonia volatilization from ammonium or ammonium-forming fertilisers. Advances in Soil Science **1**, 123-169.

Fillery, I. R. P., De Datta, S. K. 1986. Ammonia volatilization from ammonia sources applied to rice fields: 1 Methodology, ammonia fluxes and N-15 loss. Soil Science Society of America, Journal **50(1)**, 80-85.

Fleisher, Z., Kenig, A., Ravina, I., Hagin, J. 1987. Model of ammonia Volatilization from calcareous soils. Plant and Soil **103**, 205-212.

Flessa, H., Beese, F. 1995. Effects of sugar beet residues on soil redox potential and N_2O emission. Soil Science Society of America Journal **59**, 1044-1051.

Freney, J. R., Trevitt, A. C. F., Muirhead, W. A., Denmead, O. T., Simpson, J. R.,

Obcemea, W. N. 1988. Effect of water depth on ammonia loss from lowland rice. Fertiliser Research **16 (2)**,97-108.

Gezgin, S., Bayrakli, F. 1995. Ammonia volatilization from ammonium sulphate, ammonium nitrate, and urea surface applied to winter wheat on a calcareous soil. Journal of Plant Nutrition **18 (11)**, 2483-2494.

Hewitt, C. N., Street, R. A. 1992. A qualitative assessment of the emission of non-methane hydrocarbons from the biosphere to the atmosphere in the U.K.: present knowledge and uncertainties. Atmospheric Environment **26A**,3069-3077.

Holtan-Hartwig L., Bøckman O. C., 1994. Ammonia exchange between crops and air. Norwegian Journal of Agricultural Science, Supplement No. 14. 41 pp.

Humphreys, E., Freney, J. R., Muirhead, W. A., Denmead, O. T., Simpson, J. R., Leuning, R., Trevitt, A. C. F., Obcemea, W. N., Wetselaar, R., Cai G-X. 1988. Lloss of ammonia after application of urea at different times todry-seeded irrigated rice. Fertiliser Research **16 (1)**, 47-58.

IPCC/OECD 1997. Revised 1997 IPPC guidelines for national greenhouse gas inventories. OECD, 2 rue André Pascal, Paris.

IFA, 1992. World fertiliser consumption statistics No. 24. 1988/89 to 1989/90. International Fertiliser Industry Association Limited, Paris.

Jarvis, S. C., Bussink, D. W. 1990. Nitrogen losses from grazed swards by ammonia volatilization. Proceedings of the 13th General Meeting of the European Grassland Federation, June 25-29 1990, Banska Bystrica, Czecheslovakia.

Jarvis, S. C., Hatch, D. J., Lockyer, D. R. 1989. Ammonia fluxes from grazed grassland : annual losses from cattle production systems and their relation to nitrogen inputs. Journal of Agricultural Science, Cambridge **113**, 99-108.

Kaiser, E.A., Kohrs, K., Kuecke, M., Schnug, E., Munch, J.C., Heinemeyer, O. 1997. N_2O emissions from different arable crops-importance of N fertilisation levels. In: Proceedings of the 7th International N_2O workshop, Cologne,April 1997.

Klemendtson, L., Svensson, B. H., Rosswall, T. 1988. Relationships between soil moisture content and nitrous oxide production during nitrification and denitrification. Biology and Fertility of Soils **6**, 106-111.

König, G., Brunda, M., Puxbaum, H., Hewitt, C. N., Duckham, S.C., Rudolph, J. 1995. Relative contribution of oxygenated hydrocarbons to the total biogenic VOC emissions of selected mid-European agricultural and natural plant species. Atmospheric Environment **29(8),** 861-874.

McTaggart, I. P., Douglas, J. T., Clayton H., Smith, K. A. (1997) N_2O emissions from slurry and mineral N fertiliser applied to grassland. pp 201-209 In: Gaseous Nitrogen Emissions from Grasslands (Eds S. C. Jarvis, B. F. Pain). CAB International.

Meixner, F.X. 1994. Surface exchange of odd nitrogen oxides. Nova Acta Leopoldina NF70 **288**, 299-348.

Moisier, A. R., Duxbury, J. M., Freney, J. R., Heinemeyer, O., Minami, K. 1996. Nitrous oxide emissions from agricultural fields - assessment, measurement and mitigation. Plant and Soil **181**, 95-108

MÜller, C., Kammann, C., Burger, S., Ottow, J. C. G., Grunhage, l., Jager, H. J. 1997. Nitrous oxide emission from frozen and during thawing of frozen grassland soil. In: Proceedings of the 7th International N_2O workshop, Cologne,April 1997.

O'Toole, P., Morgan, M. A. 1988. efficiency of fertiliser urea: the Irish experience. pp.191-206 In: Jenkinson, D. S. Smith, K. A. (Editors) Nitrogen Efficiency in Agricultural Soils. Elsevier Applied Science, London.

Pain, B. F., Weerden, T. Van der., Jarvis, S. C., Chambers, B. J., Smith, K. A., Demmers, T. G. M., Phillips, V. R. 1997. Ammonia Emissions Inventory for the UK. Institute of Grassland and Environmental Research, (IGER), Okehampton, UK.

Patel, S. K., Panda, D., Mohanty, S. K 1989. Relative ammonia loss from urea-based fertilisers applied to rice under different hydrological situations. Fertiliser Research **19(2)**,113-120.

Recous, S., Fresnau, C., Faurie, G., Mary, B. 1988. The fate of labelled ^{15}N urea and ammonium nitrate applied to a winter wheat crop. I. Nitrogen transformations in the soil. Plant and Soil **112**, 205-214.

Remede, A., Conrad, R. 1991. Role of nitrification and denitrification for NO metabolism in soils. Biogeochemistry **12**, 189-205

Ryden, J. C., Whitehead, D. C., Lockyer, D. R., Thompson, R. B., Skinner, J. H., Garwood, E.A. 1987. Ammonia emissions from grassland and livestock production systems in the UK. Environmental Pollution **48**,173-184.

Schøjrrring, J. K. 1991. Ammonia emissions from the foliage of growing plants, pp 267-292 In: Trace gas emissions by plants. (Eds. T. D. Sharkey, E. A. Holland and H. A. Mooney),. Academic Press, San Diego.

Skiba, U., Hargreaves, K. J., Fowler, D., Smith, K. A. 1992. Fluxes of nitric and nitrous oxides from agricultural soils in a cool temperate climate. Atmospheric Environment **26 (14)**, 2477-2488.

Skiba, U., Fowler, D., Smith, K. A. 1997. Nitric oxide emissions from agricultural soils in temperate and tropical climates: Sources, controls and mitigation options. Nutrient Cycling in Agroecosystems **48**, 75-90.

Sommer, S. G., Jensen, C. 1994. Ammonia volatilization from urea and ammoniacal fertilisers surface applied to winter wheat and grassland. Fertiliser Research **37**, 85-92.

Stohl, A., Williams, E., Wotawa, G., Kronup-Kolb, H. 1996. A European Inventory of soil nitric oxide emissions and the effect of these emissions on the photochemical formation of ozone. Atmospheric Environment **30(22)**, 3741-3755.

Sutton, M. A., Burkhardt, J. K., Geurin, D., Fowler, D. 1995a. Measurement and modelling of ammonia exchange over arable croplands. pp 71-80 In: Heij, G. J. and J. W. Erisman (Editors) Acid rain research, do we have enough answers? Studies in Environmental Science 64, Elsevier Science BV.

Sutton, M. A., Pitairn, C. E. R., Fowler, D. 1993. The exchange of ammonia between the atmosphere and plant communities. Advances in Ecological Research **24**, 301-393.

Sutton M. A., Place C. J., Eager M., Fowler D., Smith R. I. 1995b. Assessment of the magnitude of ammonia emissions in the United Kingdom. Atmospheric Environment **29**, 1393-1411.

Sutton, M. A., Nemitz, E., Fowler, D., Wyers, G. P., Otjes, R., San Jose, R., Moreno, J., Schjørring, L. K., Husted, S., Meixner, F. X., Ammann, C., Neftel, A., Gut, A. 1996. The EXAMINE Project: Exchange of Atmospheric Ammonia with European Ecosystems. pp 155-161. In : Proceedings of the EUROTRAC Symposium ‘96. (Eds P. M. Borrell, P. Borrell, T Cvitaš, K. Kelly & W. Seiler). Computational Mechanics Publications, Southampton, UK.

UKRGIAN, 1994. Impacts of Nitrogen deposition in Terrestrial Ecosystems (INDITE). Report of the United Kingdom Review Group on Impacts of Atmospheric Nitrogen. 110 pp. Department of the Environment, London.

Velthof, G. L., Brader, A. B., Oenema, O. 1996. Seasonal variations in nitrous oxide losses from managed grasslands in the Netherlands, Plant and Soil **181**, 263-274.

Vlek, P. L. G., Stumpe, J. M. 1978. Effects of solution chemistry and environmental conditions on ammonia volatilization losses from aqueous systems. Soil Science Society of America Journal **42**, 416-421

Weerden, T. J. van der, Jarvis, S.C. 1997. Ammonia emission factors for N-fertilisers applied to two contrasting grassland soils. Atmospheric Pollution **95(2)**,205-211.

Whitehead, D. C., Lockyer, D. R. 1989. Decomposing grass herbage as a source of ammonia in the atmosphere. Atmospheric Environment **23**, 1867-1869.

Whitehead D. C., Raistrick, N. 1990. Ammonia volatilization from five nitrogen compounds used as fertilisers following surface application to soils of differing characteristics. Journal of Soil Science **41**, 387-394.

Whitehead, D. C., Raistrick, N. 1993. The volatilization of ammonia from cattle urine applied to soils as influenced by soil properties. Plant and Soil **148**, 43-51.

Williams, E. J., Guenther, A., Fehsenfeld, F. C. 1992. An inventory of nitric oxide emissions from soils in the United States. Journal of Geophysical Research **97**, 7511-7519.

Yamulki, S., Goulding, K. W. T., Webster, C. P., Harrison, R. M. 1995. Studies on NO and N_2O fluxes from a wheat field. Atmospheric Environment **29(14),** 1627-1635.

Yienger, J. J., Levy, H. 1995. Empirical model of the global soil-biogenic NO_x emissions. Journal of Geophysical Research. **100**, 11447 - 11464.

18 BIBLIOGRAPHY

See in particular: Asman (1992), Bouwman (1996) ECETOC (1994), Holtan-Hartwig and Bøckman (1994), IPPC/OECD (1995), Skiba et al. (1997) and Sutton et al. (1995b).

19 RELEASE VERSION, DATE AND SOURCE

Version: 3.1

Date: 3 February 1999

Updated by:

J Webb (Lead author)
ADAS Wolverhampton
UK

Mark Sutton
Institute of Terrestrial Ecology
UK

Carolien Kroeze
Wageningen Institute for Environmental and Climate Change
The Netherlands

Ute Skiba
Institute of Terrestrial Ecology
UK

Wilfried Winiwarter
Osterreichisches
AUSTRIA

Brian Pain
Institute of Grassland and Environmental Research
UK

Guiseppi Bonazzi
C. R. P. A.
ITALY

Ingrid Svedinger
Swedish Environmental Protection Agency
SWEDEN

David Simpson
EMEP MSC-W
SWEDEN

Original author:

Mark Sutton
Institute of Terrestrial Ecology
UK

20 POINT OF ENQUIRY

Any comments on this chapter or enquiries should be directed to:

Pierre Cellier

INRA Unite de Recherche "Environment et Grandes Cultures"
78850 Thiverval-Grignon
France

Tel: +33 1 30815532
Fax: +33 1 30815563
Email: cellier@bcgn.grignon.inra.fr