slurry was emitted as N_2O from deep-litter systems for fattening pigs during a 4-month measuring period. Large N_2O losses may also occur during aerobic treatment of pig slurry. Burton et al. (1993) found that approximately 0.05 kg N/kg nitrogen in slurry was lost as N_2O during a four-day treatment period.

Losses of N_2O from large lagoons with pig slurry in south-eastern United States were negligible (Harper, personal communication). Concentration profiles of N_2O above the lagoon suggest that absorption of N_2O from the atmosphere by the pig slurry in the lagoon may occur.

Losses of N_2O from muck heaps were measured by Sibbesen and Lind (1993). They arrived at a tentative estimate of 8 10^{-3} kg/kg nitrogen in the dung heap/yr. Recent results indicate that losses were much larger during warm periods, as in the summer of 1995, than during colder conditions (Lind, personal communication, 1995).

In a study on total N_2O emission from The Netherlands, Kroeze (1994) included N_2O emissions from stables. She distinguished four classes, kg N_2O-N/kg nitrogen in the waste: class 1 (<0.002) for anaerobic storage of waste; Class 2 (0.002-0.0125) for none of the types of storage in the Netherlands; Class 3 (0.0125-0.025) for biological treatment of calf veal manure and Class 4 (>0.025) for deep-litter stables, nitric-acid-treated slurry.

Jarvis and Pain (1994) and Bakken et al. (1994) made inventories of N_2O emissions from livestock holdings, but did not include N_2O emissions from animals and animal waste storage. Other studies (Bouwman et al., in press; Mosier et al., 1995b; Flessa et al., submitted) assign an emission factor to nitrogen from animal slurry. No explicit distinction was made between N_2O emission from wastes during storage and treatment, and N_2O emission from wastes after deposition or spreading onto the land.

In conclusion, there is very limited information available on N_2O emissions from animals and from animal waste during storage and treatment. Moreover, there is a wide range in estimated losses, when expressed in N_2O N/kg nitrogen in the waste. Losses from animal waste during storage range from <10^{-4} kg N for slurries to >0.15 kg N/kg nitrogen in the pig waste in deep-litter stables. Although N_2O production may be affected by waste spreading and waste processing, available data are too scarce to base a new method on.

C) N_2O from animal grazing

A brief summary of estimates of N_2O emissions derived from dung and urine deposits of grazing animals is compiled in Appendix A, Table A-1. The N_2O emission is expressed as gN/kg of the nitrogen in urine and/or dung. Two types of studies may be distinguished. The first type focuses on emissions from a well-defined urine and/or dung patch. A control treatment is generally included, to facilitate the calculation of urine and dung derived emissions. The grazed grassland is the focus in the second type of studies. Grazing derived emissions can be obtained properly when a non-grazing treatment is included. For the purpose of this compilation we consider that grazing derived emissions are similar to 'dung and urine derived' emissions. This may not be completely true, because grazing animals have also other effects than deposition of dung and urine, for example, compaction of the soil by trampling, increased turnover of nitrogen from stubble and roots, etc.

The duration of the studies ranged from 1 week up to 2 years. Though the bulk of the N_2O will be lost shortly after deposition in the field, significant amounts may still be released from the urine and dung even after a couple of weeks after deposition. Hence, short-term studies may underestimate the total N_2O losses from animal excrements (Van Cleemput et al., 1994; Velthof et al., submitted a;b).

Grazing derived emissions range from 0.002 to 0.098 kg N_2O–N/kg of nitrogen excreted (Koops et al., 1995; Muller et al., 1995a;b). The lower estimates are from well-drained unfertilised grassland soils in New Zealand. Carran et al. (1995) examined five plots, ranging from well drained high-fertile plots to poorly drained low-fertility plots. They did not provide grazing derived emissions, probably because they did not include non-grazing treatments. However, they provided data on annual dry matter yields. Assuming a mean grazing efficiency in their grazing trials of 80 per cent, for both bulls and sheep, a nitrogen content in the clover/grass mixture of 30 g/kg dry weight, and a low background emission (20 per cent of the grazing trials), the grazing derived N_2O emissions are likely in the range of 0.002-0.01 kg N_2O–N/kg nitrogen excreted, which is rather low.

Large grazing derived emissions, induced by livestock nitrogen excretion, were obtained on drained peat soils in the Netherlands. These intensively managed grassland on peat soils have also a large background emission and a large fertiliser derived emission (Velthof and Oenema, 1995).

Nearly all data pertain to temperate areas, with intensively managed grassland. The nitrogen contents of dung and especially urine are higher from this intensively managed grassland than from less intensively managed (sub)tropical grasslands. The fraction of easily hydrolysable nitrogen, i.e., urea and uric acid, is much smaller in dung and urine from animals fed with a low nitrogen content ration than from animals fed with a high nitrogen content ration. This difference will probably result in a different emissions factor. Unfortunately, data are lacking to sustain this hypothesis.

Differences in climate, i.e., rainfall and temperature patterns, may also have a significant effect. Moist and warm environments facilitate the ammonification of organically bound nitrogen in urine and dung, and subsequently, nitrification and denitrification. As a result, the effect of a relatively low ratio of easily hydrolysable nitrogen versus total nitrogen in urine and dung in low-intensity managed tropical pastures may be compensated for by the effects of temperature and moisture, to some extent. However, this is speculative since little data exist to substantiate this proposition.

Nitrogen losses as N_2O are probably lower in arid and semiarid regions and in colder climates such as Sweden. Mosier and Parton (1985) found that during the course of a year per kg of urea–N from simulated urine patches, only 0.006 kg N_2O–N was emitted as N_2O. They did find in later studies that N_2O emissions remained detectably higher 10 years after the urea had been applied to the semi-arid shortgrass prairie (Mosier et al., 1991).

An overall reasonable average emission factor for animal waste excreted in pastures is 0.02 kg N_2O–N/kg of nitrogen excreted. This emission factor is likely applicable for all regions of the world and for all types of animals.

METHODOLOGY FOR ESTIMATING N_2O FROM ANIMAL PRODUCTION

As already discussed there are three potential sources of N_2O emissions related to animal production. These are (a) animals themselves, (b) animal wastes during storage and treatment, (c) dung and urine deposited by free-range grazing animals. N_2O emissions emitted directly from animals are not reported here. Emissions from manure applied to agricultural soils from stables (e.g., daily spreading) and from grazing animals (pasture range and paddock) are considered to be emissions from agricultural soils. N_2O emissions from other animal waste management systems (AWMS) are not directly attributable to soils and are reported under Manure Management (Section 4.2). Emissions from manure used as fuel should be reported in the Energy Chapter.

Nonetheless, all sources of N_2O from animal production and agricultural soils are described here as part of the N-cycle. Caution must be applied in reporting N_2O emissions under the appropriate source categories defined above.

EQUATION 2

$$N_2O_{ANIMALS} = N_2O(_{AWMS}) = \Sigma[\ N_{(T)} \text{ x } Nex_{(T)} \text{ x } AWMS_{(T)} \text{ x } EF_{3(AWMS)}]$$

where:

$N_2O_{ANIMALS}$ = N_2O emissions from animal production in a country (kg N/yr)

$N_2O_{(AWMS)}$ = N_2O emissions from Animal Waste Management Systems in the country (kg N/yr);
= $[N_{(T=1)} \text{ x } Nex_{(T=1)} \text{ x } AWMS_{(T=1)} \text{ x } EF_{3(AWMS)}] + \ldots$
$+ [N_{(T=TMAX)} \text{ x } Nex_{(T=TMAX)} \text{ x } AWMS_{(T=TMAX)} \text{ x } EF_{3(AWMS)}]$;

$N_{(T)}$ = number of animals of type T in the country;

$Nex_{(T)}$ = N excretion of animals of type T in the country (kg N/animal /yr); (see Table 4-20);

$AWMS_{(T)}$ = fraction of $N_{EX(T)}$ that is managed in one of the different distinguished animal waste management systems for animals of type T in the country; (see Table 4-21);

$EF_{3(AWMS)}$ = N_2O emission factor for an AWMS (kg N_2O-N/kg of Nex in AWMS); (see Table 4-22);

T = type of animal category;

Nitrogen excretion

General statistics about animal numbers are provided by FAO and detailed information is available for many countries. Default values are provided in Table 4-20, which was compiled on the basis of data provided by Ecetoc (1994), and references therein, Vetter et al. (1989), Steffens and Vetter (1990). There are still uncertainties in the values listed in Table 4-20. Estimates for cattle and swine may be too high. Hence, these estimates (default values) need further attention. The excretion data are in reasonable agreement with Bouwman (in press), although some of the excretion factors as given by Bouwman (in press) are lower than the factors in Table 4-20. Once available, countries may chose to use nitrogen excretion data from the Ammonia Expert Panel of the UN-ECE task force on emission inventories. For some countries it may be desirable to distinguish other animal types than indicated in Table 4-20. If such country values are available they should be used.

TABLE 4-20

TENTATIVE DEFAULT VALUES FOR NITROGEN EXCRETION PER HEAD OF ANIMAL PER REGION (KG/ANIMAL/YR)[a]

Region	Type of Animal					
	Non-dairy cattle	Dairy cattle	Poultry	Sheep	Swine	Other animals
North America	70	100	0.6	16	20	25
Western Europe	70	100	0.6	20	20	25
Eastern Europe	50	70	0.6	16	20	25
Oceania	60	80	0.6	20	16	25
Latin America	40	70	0.6	12	16	40
Africa	40	60	0.6	12	16	40
Near East & Mediterranean	50	70	0.6	12	16	40
Asia & Far East	40	60	0.6	12	16	40

[a] *Source:* Ecetoc (1994), Vetter et al. (1988), Steffens and Vetter (1990).

Animals as direct source of N_2O

Current available information suggests that some N_2O may be released directly from animals. However, the rate of release is probably low. A value less than 0.1 g N_2O–N/kg nitrogen excreted would result in a global N_2O emission of less than 0.1 Tg, which suggests that animals are a minor source. This source is therefore not included in the methodology.

Animal Waste Management Systems

The types of Animal Waste Management System (AWMS) distinguished by Safley et al. (1992) and their compilations for a large number of countries are proposed for this methodology. Descriptions of these management systems can be found Table 4-8. Tables 4-21 and 4-6 can be used to estimate nitrogen excretion per AWMS. The AWMS is an important regulating factor in N_2O emission from animal wastes during storage and treatment. The data provided per country in Safley et al. (1992) could be used for estimating N_2O emissions from animal wastes. Significant differences in emission factors are expected between some of the AWMS.

There are several AWMS considered here:

- Anaerobic lagoons;
- Liquid systems;
- Daily spread;
- Solid storage and drylot;
- Pasture range and paddock;

- Used for fuel;
- Other systems.

N_2O emissions from all AWMS are reported under Manure Management (Section 4.2), with three exceptions:

- stable manure that is applied to agricultural soils (e.g., daily spread);
- dung and urine deposited by grazing animals on fields (pasture range and paddock);
- manure used for fuel.

The first of these sources is captured in the methodology for estimating direct emissions from agricultural soils (Section 4.5.2). The second source is reported under direct soil emissions of N_2O from animal production (Section 4.5.3). The third source, manure used as fuel, is reported in the Energy Chapter.

The class "used for fuel" is not included here as a source of N_2O, because this possible source of N_2O is considered an energy-related emission. Nevertheless, countries should estimate the amount of manure nitrogen that is used as fuel, because that amount is not applied to soils. A problem exists for the class 'Used for fuel', as it includes 'anaerobic digesters'. Moreover, it is the dung that is used for fuel and not the urine. These two factors may lead to a possible overestimation of the amount of N_2O emitted from wastes in the class 'Used for fuel', if not properly corrected. While significant N_2O losses may occur during burning, no N_2O losses are expected from anaerobic digesters. Anaerobic digesters are used especially in Asia. Data of Erda Lin (personal communication, 1995) suggest, however, that only 0.5 per cent of the total amount of animal wastes in China are used in anaerobic digesters.

TABLE 4-21
DEFAULT VALUES FOR PERCENTAGE OF MANURE N PRODUCED IN DIFFERENT ANIMAL WASTE MANAGEMENT SYSTEMS IN DIFFERENT WORLD REGIONS (FROM SAFLEY ET AL., 1992)

Region	Type of Animal	% of Manure Production per Animal Waste Management Systems						
		Anaerobic Lagoon	Liquid System	Daily Spread	Solid Storage and Drylot	Pasture Range and Paddock	Used Fuel	Other System
North America	Non-dairy Cattle (D)	0	1	0	14	84	0	1
	Dairy Cattle	10	23	37	23	0	0	7
	Poultry (E)	5	4	0	0	1	0	90
	Sheep	0	0	0	2	88	0	10
	Swine	25	50	0	18	0	0	6
	Other animals (F)	0	0	0	0	92	0	8
Western Europe	Non-dairy Cattle (D)	0	55	0	2	33	0	9
	Dairy Cattle	0	46	24	21	8	0	1
	Poultry (E)	0	13	0	1	2	0	84
	Sheep	0	0	0	2	87	0	11
	Swine	0	77	0	23	0	0	0
	Other animals (F)	0	0	0	0	96	0	4
Eastern Europe	Non-dairy Cattle (D)	8	39	0	52	0	0	1
	Dairy Cattle	0	18	1	67	13	0	0
	Poultry (E)	0	28	0	0	1	0	71
	Sheep	0	0	0	0	73	0	27
	Swine	0	29	0	0	27	0	45
	Other animals (F)	0	0	0	0	92	0	8

TABLE 4-21 (CONTINUED)
DEFAULT VALUES FOR PERCENTAGE OF MANURE N PRODUCED IN DIFFERENT ANIMA WASTE MANAGEMENT SYSTEMS IN DIFFERENT WORLD REGIONS
(FROM SAFLEY ET AL., 1992)

Region	Type of Animal	% of Manure Production per Animal Waste Management Systems						
		Anaerobic Lagoon	Liquid System	Daily Spread	Solid Storage and Drylot	Pasture Range and Paddock	Used Fuel	Other System
Oceania	Non-dairy Cattle (D)	0	0	0	0	100	0	0
	Dairy Cattle	0	0	0	0	100	0	0
	Poultry (E)	0	0	0	0	3	0	98
	Sheep	0	0	0	0	100	0	0
	Swine	55	0	0	17	0	0	28
	Other animals (F)	0	0	0	0	100	0	0
Latin America	Non-dairy Cattle (D)	0	0	0	0	99	0	1
	Dairy Cattle	0	1	62	1	36	0	0
	Poultry (E)	0	9	0	0	42	0	49
	Sheep	0	0	0	0	100	0	0
	Swine	0	8	2	51	0	0	40
	Other animals (F)	0	0	0	0	99	0	1
Africa	Non-dairy Cattle (D)	0	0	1	3	96	0	0
	Dairy Cattle	0	0	12	0	83	0	5
	Poultry (E)	0	0	0	0	81	0	19
	Sheep	0	0	0	1	99	0	1
	Swine	0	7	0	93	0	0	0
	Other animals (F)	1	0	0	0	99	0	1

TABLE 4-21 (CONTINUED)
DEFAULT VALUES FOR PERCENTAGE OF MANURE N PRODUCED IN DIFFERENT ANIMA WASTE MANAGEMENT SYSTEMS IN DIFFERENT WORLD REGIONS (FROM SAFLEY ET AL., 1992)

Region	Type of Animal	% of Manure Production per Animal Waste Management Systems						
		Anaerobic Lagoon	Liquid System	Daily Spread	Solid Storage and Drylot	Pasture Range and Paddock	Used Fuel	Other System
Near East and Mediterranean	Non-dairy Cattle (D)	0	0	2	0	77	18	2
	Dairy Cattle	0	0	3	3	77	18	0
	Poultry (E)	0	1	0	0	71	0	28
	Sheep	0	0	0	0	100	0	0
	Swine	0	32	0	68	0	0	0
	Other animals (F)	0	0	0	0	100	0	0
Asia and Far East	Non-dairy Cattle (D)	0	0	16	14	29	40	0
	Dairy Cattle	6	4	21	0	24	46	0
	Poultry (E)	1	2	0	0	44	1	52
	Sheep	0	0	0	0	83	0	17
	Swine	1	38	1	53	0	7	0
	Other animals (F)	0	0	0	0	95	0	5

(D) Includes buffalo
(E) Includes chickens, turkeys and ducks
(F) Includes goats, horses, mules, donkeys and camels

Tentative (default) emission factors (EF_3) for the different AWMS are shown in Table 4-22. These factors were derived on the basis of a very limited amount of information. Uniform factors for all over the world are proposed. This may be incorrect, as temperature and moisture may have positive effects on the size of the processes and, hence, on losses. However, as animal production systems are found in warmer regions, and low-intensity systems have less easily hydrolysable nitrogen in the excretions discussed above, a uniform factor for all regions would seem appropriate.

TABLE 4-22
TENTATIVE DEFAULT VALUES FOR N_2O EMISSION FACTORS FROM ANIMAL WASTE PER ANIMAL WASTE MANAGEMENT SYSTEM, KG N_2O–N/KG NITROGEN EXCRETED

Animal Waste Management System[a]	Emission Factor EF_3
Anaerobic lagoons[b]	0.001 (<0.002)
Liquid systems[b]	0.001 (<0.001)
Daily spread[c]	0.0 (no range)
Solid storage and drylot[c]	0.02 (0.005-0.03)
Pasture range and paddock (grazing)[d]	0.02 (0.005-0.03)
Used as fuel[e]	Not included in this Chapter
Other systems[b]	0.005

[a] The fraction of manure nitrogen produced in different Animal Waste Management Systems for cattle, swine and buffalo can be estimated as proposed in Table 4-21, or as given by Safley et al. (1992).

[b] To be reported under "Manure Management".

[c] To be reported under "Agricultural Soils" (Workbook, Section 4-6) under direct soil emissions from agricultural fields after spreading. (Emissions are assumed not to occur before spreading).

[d] To be reported under "Agricultural Soils" (Workbook, Section 4-6) under direct soil emissions from animal production.

[e] To be reported in the Energy Chapter.

4.5.4 Indirect N_2O Emissions from Nitrogen Used in Agriculture.

OVERVIEW OF SOURCES

The pathways for synthetic fertiliser and manure input that give rise to indirect emissions are considered to be:

A. Volatilisation and subsequent atmospheric deposition of NH_3 and NO_x (originating from the application of fertilisers);
B. Nitrogen leaching and runoff;
C. Human consumption of crops followed by municipal sewage treatment;
D. Formation of N_2O in the atmosphere from NH_3;
E. Food processing.

Of these pathways, methodologies for estimating N_2O emissions from A-C are proposed. Nitrous oxide emissions from human waste are described below. However, these N_2O emissions are allocated to the Waste Chapter (see Section 6.4, Reference Manual and Workbook). At present, information is insufficient to estimate emissions from D and E.

In order to estimate the associated N_2O fluxes, the following data are needed:

- Synthetic nitrogen fertiliser consumption (N_{FERT}). This is available by country from FAO yearbooks and is probably the most reliable piece of input data.
- Livestock nitrogen excretion (N_{ex}) can be estimated reasonably well from FAO livestock populations and measured kg N/animal/yr excretion factors as given in Table 4-20.
- Crop production (Crop) is available from FAO production yearbooks in kg dry biomass/yr.
- Sewage nitrogen production can be estimated from FAO per capita protein consumption data (PROTEIN) and human population counts (Nr_{PEOPLE}). Protein consumption may vary by a factor of 2 between countries, e.g., Americans and Indians consume 110 and 55 g protein/person/day, respectively.

Emissions of NH_3 and NO_x (kg N/yr) are estimated from fertiliser use and livestock nitrogen excretion.

These N_2O–N emissions are to be calculated from a country's NO_x and NH_3 emissions and nitrogen transported in leaching and runoff, so that all N_2O formed as a result of NO_x and NH_3 emissions and leaching and runoff in country Z are assigned to country Z, even if the actual N_2O formation takes place in another country. This implies that NO_x and NH_3 and nitrogen from leaching and runoff imported into a country is not included in the calculations.

A. Atmospheric deposition of NO_x and NH_3

Atmospheric deposition of nitrogen compounds such as nitrogen oxides (NO_x) and ammonium (from NH_3) fertilise soils and surface waters and as such enhance biogenic N_2O formation. However, it is recognised that other sources of atmospheric inputs of N compounds to agricultural soils are important. These sources include fuel combustion, for example. Atmospheric deposition of these sources is not accounted here because only those N emissions originating from the application of fertilisers are presently considered. Indeed, Brumme and Beese (1992) showed that after two decades of atmospheric deposition of acidifying compounds (ammonium and sulphuric acids), N_2O emissions from German forest soils were enhanced by up to a factor of 5. Reported rates of N_2O emissions are between 0.002 and 0.016 kg N_2O–N/kg of the amount of nitrogen deposited onto soils (Bowden et al., 1991; Brumme and Beese, 1992; Kasimir, personal communication). This is within the range of emission factors suggested in Section 4.5.2 for synthetic fertilisers. We therefore propose to calculate N_2O-N emissions as 0.01 kg N_2O-N /kg of NO_x-N and NH_3-N emitted annually within a country (EF_4, Table 4-23).

TABLE 4-23
DEFAULT EMISSION FACTORS FOR INDIRECT EMISSIONS

$EF_{4\ (N\ deposition)}$	=	0.01 (0.002-0.02) kg N_2O-N/kg NH_3-N and NO_X–N emitted
$EF_{5\ (leaching/runoff)}$	=	0.025 (0.002-0.12) kg N_2O-N/kg N leaching/runoff
$EF_{6\ (sewage)}$	=	0.01 (0.002-0.02) kg N_2O-N/kg sewage-N produced

Agricultural ammonia emissions can be derived from NH_3 volatilisation studies. Animal manure (dung + urine) is one of the most important sources of NH_3. According to Van der Hoek (1994), up to 50 per cent of the mineral nitrogen in animal manure (i.e., about

25 per cent of total N) may be lost shortly as NH_3 after application to soil. He also shows that this percentage depends considerably on the application technique used. Schimel et al. (1986) assumed that, as a minimum estimate, 20 per cent of manure nitrogen applied to soils is volatilised as NH_3 soon after application. The amount of NH_3 volatilised may be lower in acid and near neutral pH soils. According to Bouwman (in preparation), about 25 per cent of livestock nitrogen excretion is emitted as NH_3 world-wide. For synthetic fertilisers Van der Hoek (1994) uses a much lower percentage of only 2 per cent of the nitrogen used in the Netherlands that is lost as NH_3. Bouwman (in preparation), however, estimated that almost 10 per cent of synthetic fertiliser-N is emitted as ammonia world-wide. Although climate and fertiliser type (e.g., urea or ammonium sulphate) may influence ammonia volatilisation, we use default values for NH_3 and NO_x volatilisation: 0.1 kg nitrogen/kg synthetic fertiliser nitrogen applied to soils and 0.2 kg nitrogen/kg of nitrogen excreted by livestock are proposed ($Frac_{GASF}$ and $Frac_{GASM}$, Table 4-19).

B. Leaching and Runoff

A considerable amount of fertiliser nitrogen is lost from agricultural soils through leaching and runoff. The leached/runoff nitrogen enters groundwater, riparian areas and wetlands, rivers and eventually the coastal ocean. In many world areas, it is one of the most important inputs of nitrogen to those systems. A WHO/UNEP report (1989) showed that over 10 per cent of European rivers had a nitrate content ranging from 9 to 25 mg nitrate-N/L. Other sources include sewage, industries and atmospheric deposition. Fertiliser nitrogen in ground water and surface waters enhances biogenic production of N_2O as the nitrogen undergoes nitrification and denitrification.

The fraction of the fertiliser and manure nitrogen lost to leaching and surface runoff ($Frac_{LEACH}$) may range from range 0.1-0.8 (Seitzinger and Kroeze, in preparation). A value of 0.3 is proposed as default here (Table 4-24). For this purpose total nitrogen excretion is used (Nex) in order to include manure produced during grazing:

EQUATION 3

$$N_{LEACH} = [N_{FERT} + N_{EX}] \times Frac_{LEACH}$$

The sum of the emission of N_2O due to N_{LEACH} in: 1) groundwater and surface drainage (EF_5-g), 2) rivers (EF_5-r), and 3) coastal marine areas (EF_5-e) is calculated to obtain the N_2O emission factor (EF_5) for N_{LEACH}. Although not specified, the total amount of nitrogen eventually denitrified remains the same but some is denitrified in riparian area and wetlands before the nitrogen reaches the ocean. In future assessment methodologies, a separate emission factor should be used in the workbook for each of these three environments.

TABLE 4-24
DEFAULT VALUES OF PARAMETERS FOR INDIRECT EMISSIONS

$Frac_{NPR}$	0.16 kg N/kg of protein
$Frac_{LEACH}$	0.3 (0.1-0.8) kg N/kg of fertiliser or manure N

Groundwater and surface drainage

Supersaturated concentrations of nitrous oxide in groundwater and in surface water draining agricultural lands may occur due to leaching of N_2O from the soil towards drainage and groundwater, or production during nitrification and/or denitrification of fertiliser nitrogen in the groundwater or drainage ditches. Many factors can affect the amount of N_2O in these waters including the amount of nitrogen leaching into the groundwater, different land use practices, soil types and climate. Fertiliser nitrogen in groundwater or drainage water is primarily in the form of NO_3-N. A review of the literature indicates that while the range of N_2O concentrations reported is large, there is some relationship between the concentration of N_2O-N and NO_3-N in groundwater and agricultural drainage water. The ratio of N_2O-N to NO_3-N concentration in groundwater and agricultural drainage water at over 25 locations in urban, agricultural and woodland areas in Japan, Israel and the United States ranged from 0.0001 to 0.06 (Dowdell et al., 1979; Minami and Fukuski, 1984; Ronen et al., 1988; Minami and Oshawa, 1990; Ueda et al., 1991; Ueda et al., 1993). The ratio of N_2O-N to NO_3-N in agricultural drainage ditches and groundwater under agricultural fields ranged from approximately 0.0003 to 0.06. The ratios of N_2O-N to NO_3-N in agricultural drainage ditches were generally lower (0.003 or less) than ratios in agricultural groundwater. Rapid loss of N_2O to the atmosphere may account for the generally lower ratios in drainage ditch water. The ratio of N_2O-N to NO_3-N in agricultural groundwater was generally between 0.003 and 0.06, with values between 0.007 and 0.02 common. Assuming that all N_{LEACH} is in the form of $NO_{3,}$ we recommend a default emission factor of 0.015 (EF_5-g) for N_2O from N_{LEACH} in groundwater and drainage ditches, with a range of 0.003 to 0.06. The amount of N_2O emitted from groundwater (by upward diffusion or following entry of groundwater into surface water through rivers, irrigation, and drinking water) and agricultural drainage water is then estimated as:

EQUATION 4

N_2O emissions from groundwater and agricultural drainage water

=

N_{LEACH} x EF_5-g

where EF_5–g = 0.015 kg N_2O-N/kg N_{LEACH}, assuming that all N_2O produced in a particular year is emitted during that year.

Rivers

Once N_{LEACH} from groundwater and surface water enters rivers, additional N_2O is produced associated with nitrification and denitrification of N_{LEACH} (Seitzinger and Kroeze, in preparation). It is assumed that minimal denitrification occurs in groundwater and therefore that all N_{LEACH} enters rivers.

Nitrification: N_2O can be produced during nitrification of N_{LEACH} in rivers. While much of the N_{LEACH} may enter rivers as nitrate, algae and aquatic plants can assimilate the nitrate into organic matter, which is released as ammonia, following decomposition of that organic matter. Ammonia in rivers is rapidly nitrified (Lipschultz et al., 1986). The N_{LEACH} entering rivers nitrifies on average 0.5-3 times during river transport. We assume for our default methodology that all N_{LEACH} entering rivers is nitrified once during river transport. The N_2O yield (moles N_2O-N/mol of NO_3-N) during nitrification is generally between 0.002 and 0.003 at atmospheric oxygen levels (0.2 atm partial pressure); Goreau et al., 1980), although enhanced yields of N_2O are found at reduced O_2 concentrations (Goreau et al., 1980). While reduced oxygen levels occur in some rivers, especially those with high nutrient inputs, we suggest an N_2O yield of 0.003 for nitrification.

Denitrification: During river transport a considerable amount of nitrogen is lost via denitrification in riverine sediments. A wide range of denitrification rates has been measured in rivers or streams; rates are generally lowest in unpolluted streams (Duff et al., 1984) with highest rates in polluted rivers/streams (Robinson et al., 1979; Cooper and

Cooke, 1984; Seitzinger, 1988, 1990; Christensen and Soerensen, 1988; Christensen et al., 1989). Estimates of the magnitude of N-removal via denitrification range from 1 to 75 per cent of the external nitrogen inputs based on mass balance models and/or from measurements of denitrification (Seitzinger, 1990). Factors likely to affect the fraction of nitrogen removed by denitrification include length and depth of the river, flow rate, water residence time, oxygen content, organic content of sediments, and season. In a number of rivers denitrification removed 50 per cent of the nitrogen inputs, even over short sections (Kaushik and Robinson, 1976; Hill, 1979, 1981, 1983; van Kessel, 1977; Swank and Caskey, 1982). For the assessment we assume that denitrification removes 50 per cent of N_{LEACH} inputs to rivers. N_2O associated with denitrification (Jorgensen et al., 1984) is released from river sediments. The ratio of $N_2O:N_2$ emitted from river sediments is generally within the range 0.001-0.005, although in heavily polluted sediments yields up to 6 per cent have been observed (Seitzinger, 1988). A constant ratio of 0.005 for N_2O-N emission to denitrification (N_2–N production) in rivers is suggested.

In summary, the emission factor for N_{LEACH} in rivers due to nitrification and denitrification [EF_5-r] is thus equal to 0.005 x N_{LEACH} [for nitrification] plus 0.005 x (N_{LEACH}/2) [for denitrification], or 0.0075 x N_{LEACH}. Therefore, N_2O-N produced from N_{LEACH} during river transport = N_{LEACH} x (EF_5-r), where EF_5-r = 0.0075.

Estuaries

Rivers are the major conduit for nitrogen transport to the coastal ocean (via estuaries). As discussed above, half of N_{LEACH} is assumed to be removed by denitrification in rivers in the form of N_2 and N_2O. The remaining 50 per cent of N_{LEACH} is discharged by rivers to estuaries. Nitrogen inputs to estuaries can undergo nitrification and denitrification, with associated N_2O production.

Nitrification: Pelagic nitrification rates in estuaries generally range from 0-22 umol/l/d (Berounsky and Nixon, 1993). Estuarine nitrification rates are affected by a number of factors such as ammonia concentrations, temperature (Berounsky and Nixon, 1985 and 1993), oxygen (Helder and DeVries, 1983), suspended particulate matter (Helder and DeVries, 1983; Owens, 1986), and light (Horrigan and Springer, 1990). However, no predictive factor has been developed to estimate pelagic nitrification rates across a range of estuaries. In Narragansett Bay (USA), approximately half of the river inputs of inorganic nitrogen to the Bay were nitrified in the bay (Berounsky and Nixon, 1993; Seitzinger and Kroeze, in preparation). For the assessment methodology, we assume that half of the rivers inputs of N_{LEACH} are nitrified again in estuaries, and that the ratio of N_2O-N to NO_3–N produced is 0.005, as discussed above for rivers.

Denitrification: Some of the most extensive studies of denitrification are in estuaries (Kemp et al., 1990; Jenkins and Kemp, 1984; Jensen et al., 1984 and 1988; Smith et al., 1985). A relatively good relationship has been found between denitrification and inorganic nitrogen inputs to estuaries from rivers. The amount of nitrogen removed by denitrification is equivalent to a relatively constant percentage (50 per cent) of inorganic nitrogen inputs to a variety of estuaries (Seitzinger, 1988). Those estuaries vary in a number of characteristics including nitrogen loading rates (25 to 516 x 10^{-6} mol N m^{-2}/h), extent of inter-tidal area (<1 per cent to 50 per cent), and latitude (subtropical to sub-arctic). For the assessment methodology, 50 per cent of the N_{LEACH} that is carried to estuaries by rivers is denitrified, and the ratio of N_2O-N to denitrification (N_2-N) emitted is 0.005, as discussed above for rivers. N_{LEACH} that enters estuaries but is not denitrified, is either buried in the sediments as organic nitrogen or exported to the continental shelf region where additional N_2O can be produced. Nitrous oxide production associated with this fraction of N_{LEACH} is not accounted for in this methodology.

In summary, the Phase II methodology assumes the following: 1) half of the N_{LEACH} is transported to estuaries by rivers, 2) half of the N_{LEACH} in estuaries is nitrified again in the estuary with a ratio of N_2O-N to NO_3-N of 0.005, and 3) half of the N_{LEACH} in estuaries is denitrified in the estuary with a N_2O-N to denitrification (N2-N) ratio of 0.005. Therefore, N_2O-N produced from N_{LEACH} in estuaries = N_{LEACH} x (EF_5-e) where EF_5-e = 0.0025.

The combined emission factor [EF_5] for N_2O due to N_{LEACH} in: 1) groundwater and surface drainage (EF_5-g = 0.015 kg N_2O-N/kg N_{LEACH}), 2) rivers (EF_5-r = 0.0075 kg N_2O-N/kg N_{LEACH}), and 3) coastal marine areas (EF_5-e = 0.0025 kg N_2O-N/kg N_{LEACH}) is 0.025 (EF_5). Therefore:

EQUATION 5

$$N_{LEACH} = [N_{FERT} + Nex] \text{ x } Frac_{LEACH} \text{ and } N_2O_{(L)} = N_{LEACH} \text{ x } EF_5$$

where the default values are $Frac_{LEACH}$ = 0.3 kg N/kg N input to soils and EF_5 = 0.025 kg N_2O-N/kg N_{LEACH} (see Tables 4-23 and 4-24).

C. Human consumption followed by municipal sewage treatment

Nitrous oxide emissions from human waste are described below. However, these N_2O emissions are allocated to the Waste Chapter (see Section 6.4, Reference Manual and Workbook).

Consumption of foodstuffs by humans results in the production of sewage. Sewage can be disposed of directly on land (night-soil or spray irrigation) or discharged into a water source (e.g., rivers and estuaries). Before disposal on land or into water, it also can be processed in septic systems or wastewater treatment facilities. During all of these stages, nitrous oxide can be produced during nitrification and denitrification of sewage nitrogen.

Sewage nitrogen (N_{SEWAGE}) production can be estimated from FAO per capita protein consumption data (Protein) and human population counts (Nr_{PEOPLE}), assuming that nitrogen constitutes about 16 per cent by weight of protein ($Frac_{NPR}$, Table 4-23).

EQUATION 6

$$N_{SEWAGE} = \text{Protein x } Frac_{NPR} \text{ x } Nr_{PEOPLE}$$

Nitrous oxide emissions resulting from sewage nitrogen are estimated following: land disposal or wastewater treatment of sewage, and input of sewage nitrogen to rivers and estuaries.

Disposal or wastewater treatment of sewage

No studies were found quantifying nitrous oxide emissions from land disposal of sewage, although supersaturated concentrations of N_2O in groundwater under cultivated land irrigated with sewage effluent have been reported (Ronen et al., 1988). A few studies have documented N_2O emission associated with wastewater treatment operations (e.g., Hemond and Duran, 1989; Hanaki et al., 1992; Hong et al., 1993; Debruyn et al., 1994; Czepiel et al., 1995).

Three studies have directly measured N_2O emissions from operating wastewater treatment facilities (Hemond and Duran, 1989; Czepiel et al., 1995; Velthof and Oenema, 1993). All studies reported low rates of N_2O emission. For example, nitrous oxide emissions from a secondary treatment wastewater facility in New Hampshire (USA) were approximately 0.0006 kg N_2O-N/kg sewage N, assuming 3.2 kg sewage nitrogen are produced/person/yr (Czepiel et al., 1995). Velthof and Oenema (1993) found N_2O losses of 0.022 kg/day per day in a vented closed waste water treatment facility that had a daily input of 900 kg N, suggesting that N_2O losses were 0.00005 kg/kg nitrogen entering the system. Additional N_2O released to the atmosphere following discharge of supersaturated effluent to the environment is also low (0.0007 kg N_2O-N/kg sewage N) (Hemond and Duran, 1989). Laboratory studies simulating wastewater treatments processes demonstrate that conditions in the treatment facility can affect the amount of N_2O produced, including the ratio of nitrate to oxidisable carbon and nitrogen loading rate (Nogita et al., 1981; Hanaki et al., 1992). It is difficult to relate these laboratory results to emissions from sewage treatment facilities. For example, in the laboratory study of Nogita et al. (1981), 100 times more N_2O-N was formed per unit of sewage nitrogen than in the field study of Czepiel et al. (1995).

For the Phase II methodology N_2O associated with sewage treatment and land disposal is assumed to be negligible. This is based on the low emission rates of N_2O reported for operating wastewater treatment facilities (Hemond and Duran, 1989; Czepiel et al., 1995; Velthof et al., submitted a;b), and the lack of information on N_2O production from land disposal of human sewage. This assumption should be reviewed in the future, as new data become available.

Rivers and estuaries

N_2O is produced in rivers and estuaries following nitrification and denitrification of sewage nitrogen inputs (Seitzinger and Kroeze, in preparation). The sewage nitrogen can be discharged directly to aquatic environments (e.g., rivers, estuaries) or enter aquatic environments following leaching from terrestrially disposed sewage. Here it is assumed that minimal removal of sewage nitrogen occurs during land disposal or sewage treatment, and that all sewage nitrogen enters rivers and/or estuaries. This latter assumption should be reviewed in the future, as more data become available.

Nitrous oxide emissions in rivers and estuaries due to nitrification and denitrification of sewage nitrogen are estimated using the same assumptions used for fertiliser nitrogen leached to rivers and estuaries (see B). These assumptions result in emission coefficients of EF_6–r = 0.0075 kg N_2O–N/kg N_{SEWAGE} (rivers) and EF_6–e = 0.0025 kg N_2O–N/kg N_{SEWAGE} (estuaries). The sum of N_2O emissions in rivers (0.0075 x N_{SEWAGE}) and estuaries (0.0025 x N_{SEWAGE}) associated with nitrification and denitrification of N_{SEWAGE} is calculated as:

EQUATION 7

$$N_2O_{(S)} = N_{SEWAGE} \times EF_6$$

where:

EF_6 = 0.1 kg N_2O–N/kg N_{SEWAGE} (Table 4-23)

D. Formation of N_2O in the atmosphere from NH_3

Dentener and Crutzen (1994) proposed that oxidation of NH_3 and subsequent reaction of the intermediate NH_2 radical with NO_2 could lead to a production of 0.9 (+0.9+–0.4) Tg N_2O (0.6 Tg N/yr).

The most important reactions for N_2O production are given by:

$NH_3 + OH \rightarrow NH_2 + H_2O$ R_1
$NH_2 + NO_2 \rightarrow N_2O + H_2O$ R_2
$NH_2 + O_3 \rightarrow NH_2O + O_2$ (and other products) R_3

The homogeneous reaction of NH_3 with OH radical (R_1) is rather slow, and is only of importance in regions with high OH and low sulphate aerosol concentrations. More than 95 per cent of the global amount of NH_3 oxidised by OH occurs between 30° N and 30° S. The chemistry of the amine radical NH_2 is not known, reactions R_2 and R_3 having uncertainties of a factor of 2 (DeMore et al.,1994). In addition, emissions and concentrations of NH_3 and NO_2 in tropical regions are poorly quantified.

Dentener and Crutzen (1994) parameterized natural NH_3 emissions from vegetation using a highly uncertain NH_3 canopy compensation point (the atmospheric concentration above which plants assimilate and below which they emit NH_3). Without considering this compensation point, N_2O production was reduced by 55 per cent. Other sources of NH_3 in the tropics include animal waste decomposition (both from wild and domestic animals), fertiliser application and biomass burning emissions. Considering the relative strengths of these sources, about half of the atmospheric N_2O production may be associated with agricultural nitrogen, amounting to about 0.4 Tg N_2O/yr. Due to the high uncertainty of this estimate (ca. 100 per cent), we have not included this potentially important source in our agricultural N_2O emissions inventory. More measurements on the co-occurrence of high NH_3, NO_2 and OH concentrations in the tropics are needed to provide more insight in the photochemical production of N_2O. Furthermore laboratory experiments on the reaction rates, especially of reactions R_2 and R_3, would be extremely valuable.

E. Food processing operations

Some food processing operations are sources of N_2O. A fraction of the edible crop harvest is not consumed by people and enters the waste stream, for instance when it is landfilled, composted, burned or fed to animals. At this point, there are no data to calculate the magnitude of this N_2O source and therefore, there is no methodology at the present time (For future purposes, this source would be reported under Industrial Processes).

METHODOLOGY FOR ESTIMATING INDIRECT N_2O

Based on the above, we propose the following methodology for calculating a country's indirect N_2O emissions (kg N/yr):

EQUATION 8

$$N_2O_{indirect} = N_2O_{(G)} + N_2O_{(L)} + N_2O_{(S)}$$

where:

$N_2O_{(G)}$ = N_2O produced from atmospheric deposition of NO_x and NH_3 (kg N/yr);

$N_2O_{(L)}$ = N_2O produced from nitrogen leaching and runoff (kg N/yr);

$N_2O_{(S)}$ = N_2O produced from human sewage (kg N/yr) to be reported in the Waste Chapter.

A. Atmospheric deposition of NO_x and NH_3

Methodology:

EQUATION 9

$$N_2O_{(G)} = (N_{FERT} \text{ x } Frac_{GASF} + N_{EX} \text{ x } Frac_{GASM}) \text{ x } EF_4$$

where :

EF_4 = emission factor for atmospheric deposition (kg N_2O-N/kg NH_3-N and NO_x–N emitted) (see Table 4-23);

$Frac_{GASF}$ = fraction of synthetic fertiliser nitrogen applied to soils that volatilises as NH_3 and NO_x (kg NH_3-N and NO_x–N/kg of N input) (see Table 4-19);

$Frac_{GASM}$ = fraction of livestock nitrogen excretion that volatilises as NH_3 and NO_x (kg NH_3-N and NO_x–N/kg of N excreted) (see Table 4-19).

Input:

N_{FERT} = fertiliser nitrogen use in country (kg N/yr). Recommended source: FAO data;

Nex = livestock nitrogen excretion in country (kg N/yr) (see Table 4-20).

B. Leaching and runoff

Methodology:

N_{LEACH} = $(N_{FERT} + N_{ex}) \times Frac_{LEACH}$

$N_2O_{(L)}$ = $N_{LEACH} \times EF_5$

where :

$Frac_{LEACH}$ = fraction of nitrogen input to soils that is lost through leaching and runoff (kg N/kg of nitrogen applied) (see Table 4-24);

EF_5 = emission factor for leaching/runoff (kg N_2O-N/kg N leaching/runoff) (see Table 4-23);

N_{LEACH} = N leaching in country (kg N/yr).

Input:

N_{FERT} = see A.

Nex = see A.

C. Sewage treatment (see Waste Chapter, Section 6.4)

Methodology:

EQUATION 10

$$N_{SEWAGE} = PROTEIN \times Nr_{PEOPLE} \times Frac_{NPR}$$

$$N_2O_{(S)} = N_{SEWAGE} \times EF_6$$

where:

EF_6 = emission factor for sewage treatment (kg N_2O-N/kg sewage-N produced) (see Table 4-23);

$Frac_{NPR}$ = fraction of nitrogen in protein (kg N/kg of protein) (see Table 4-24);

Input:

PROTEIN = annual per capita protein consumption in country (kg protein/person/yr). Recommended source: FAO;

Nr_{PEOPLE} = number of people in country. Recommended source: FAO.

Future Work

The revised methodology for N_2O described above is a generalised approach which treats all agricultural systems as being the same under all climates, in all soils, in all crops and in all management systems. This clearly provides uncertainties in inventory calculations. However, the ranges of conversion factors provided should cover the potential N_2O emissions from each country, whatever climate, soils and set of crops is involved. To make significant improvement in inventory methodologies for N_2O, the next step is to utilise process-based models to produce country inventories. These would include models of direct emissions from agricultural soils, appropriate animal management models for N_2O from animal production, simulation models which represent nitrogen transformations in aquatic systems, including riparian areas, wetlands, rivers estuaries, continental shelves and the deep ocean.

Since soil carbon and nitrogen cycles are tightly integrated, both carbon and nitrogen should be considered together so that various aspects of the carbon and nitrogen cycle and CO_2 and N_2O production can be more accurately defined. For example, the amount of nitrogen leached from agricultural fields represents a very large component of the global N_2O production according to this revised methodology. The accuracy of the nitrogen leaching fraction prediction is closely tied to carbon turnover in the soil as it controls nitrogen mineralisation and immobilisation. The turnover and retention of nitrogen in all soils is intimately linked with the carbon cycle. Conversely, carbon retention in soils is directly tied to mineral nitrogen availability.

There are additional issues that include: (1) development of methodologies that represent the effect of cropping system, soil, and climate on CO_2 and N_2O budgets; (2) including soil methane oxidation in national budgets (without the soil sink component, atmospheric methane concentrations would be increasing about two times faster than the increase rate observed in the 1980's); (3) including the impact of NO_x emissions from agricultural soils on local and regional atmospheric oxidants and ozone concentration; (4) determining the impact of carbon and nitrogen losses and retention on system sustainability; (5) considering mitigation methodologies to decrease CO_2 and N_2O emissions from agriculture and to improve the soil sink capacity for CH_4; and (6) investigating errors that may arise as a result of aggregating field scale data to the national level.

Appendix A

Estimates of Nitrous Oxide Emissions from Dung and Urine Deposits of Grazing Animal.

This appendix presents a brief summary of estimates of nitrous oxide emissions derived from dung and urine deposits of grazing animals.

TABLE A-1
EMISSION OF N_2O FROM ANIMAL DUNG AND URINE DEPOSITED IN GRASSLAND :
THE AMOUNT OF N_2O EMITTED IS EXPRESSED IN % OF THE AMOUNT OF N EXCRETED BY THE GRAZING ANIMAL.

Country	Soil Type	Treatment	Period	N_2O Emission	Reference
United Kingdom	clay loam	urine	4wks	1-5	Monaghan and Barraclough (1993)
New Zealand	silt loam	urine	6 wks	<0.5	Sherlock and Goh (1983)
Germany	loess	urine	11 wks	3.8	Flessa et al. (submitted)
Germany	loess	dung	11 wks	0.5	Flessa et al. (submitted)
The Netherlands	clay	urine	4 wks	0.5	Velthof and Oenema (1994)
The Netherlands	peat	urine	3 wks	38	Koops et al. (unpublished)
The Netherlands	sand	urine	2 wks	8-16	De Klein and Logtestijn (1994)
United Kingdom	clay loam	grazing	1 wk	1 8	Velthof et al. (submitted a;b)
The Netherlands	sand	grazing	32 wks	1.0	Velthof and Oenema (1995)
The Netherlands	clay	grazing	32 wks	2.1	Velthof and Oenema (1995)
The Netherlands	peat	grazing	32 wks	1.5	Velthof and Oenema 1995)
The Netherlands	peat	grazing	32 wks	7.7	Velthof and Oenema (1995)
Germany	-	urine/dung	1 yr	0.4-1.3	Poggemann et al. (1995)
The Netherlands	sand	grazing	2 yrs	1.5	Velthof et al. (submitted a;b)
The Netherlands	clay	grazing	2 yrs	3.3	Velthof et al. (submitted a;b)
The Netherlands	peat	grazing	2 yr	2.3	Velthof et al. (submitted a;b)
The Netherlands	peat	grazing	2 yrs	9.8	Velthof et al. (submitted a;b)
New Zealand	silt loam	grazing	1 yr	0.2-1.0	Carran et al. (1995)