CHAPTER 4

AGRICULTURE

4. AGRICULTURE

4.1 Overview

Agricultural activities contribute directly to emissions of greenhouse gases through a variety of different processes. This chapter discusses four greenhouse gas-emitting activities:

- **CH_4 and N_2O emissions from domestic livestock (enteric fermentation and manure management)**
  - ***CH_4 emissions from enteric fermentation in domestic livestock***
    Methane is produced in herbivores as a by-product of enteric fermentation, a digestive process by which carbohydrates are broken down by micro-organisms into simple molecules for absorption into the bloodstream. Both ruminant animals (e.g., cattle, sheep) and some non-ruminant animals (e.g., pigs, horses) produce CH_4, although ruminants are the largest source since they are able to digest cellulose, a type of carbohydrate, due to the presence of specific micro-organisms in their digestive tracts. The amount of CH_4 that is released depends on the type, age, and weight of the animal, the quality and quantity of the feed, and the energy expenditure of the animal.

  - ***CH_4 emissions from manure management***
    CH_4 is produced from the decomposition of manure under anaerobic conditions. These conditions often occur when large numbers of animals are managed in a confined area (e.g., dairy farms, beef feedlots, and swine and poultry farms), where manure is typically stored in large piles or disposed of in lagoons.

  - ***N_2O emissions from manure management***
    During storage of manure, some manure nitrogen is converted to N_2O. Emissions of N_2O related to manure handling before the manure is added to soils are included in this source category. (Manure-related N_2O emissions from soils are considered agricultural soil emissions).

- **CH_4 emissions from rice cultivation**

  Anaerobic decomposition of organic material in flooded rice fields produces methane, which escapes to the atmosphere primarily by transport through the rice plants. The amount emitted is believed to be a function of rice species, number and duration of harvests, soil type and temperature, irrigation practices, and fertiliser use. The seasonally integrated CH_4 flux depends upon the input of organic carbon, water regimes, time and duration of drainage, soil type etc.

- **CH_4, CO, N_2O, and NO_X emissions from agricultural burning (savanna and agricultural burning)**
  - ***CH_4, CO, N_2O, and NO_X emissions from the prescribed burning of savannas***
    The burning of savannas – areas in tropical and subtropical formations with continuous grass coverage – results in the instantaneous emissions of carbon

dioxide, but because the vegetation regrows between burning cycles, the carbon dioxide released into the atmosphere is reabsorbed during the next vegetation growth period. Net CO_2 emissions are therefore assumed to be zero. However, savanna burning also releases gases other than CO_2, including methane, carbon monoxide, nitrous oxide and oxides of nitrogen. Unlike CO_2 emissions, these are net emissions.

– ***CH_4, CO, N_2O, and NO_X emissions from the burning of agricultural residues***

The burning of crop residues is not thought to be a net source of carbon dioxide because the carbon released to the atmosphere is reabsorbed during the next growing season. However, this burning is a significant source of emissions of methane, carbon monoxide, nitrous oxide, and nitrogen oxides. It is important to note that some crop residues are removed from the fields and burned as a source of energy, especially in developing countries. Non-CO_2 emissions from this type of burning are dealt with in the *Energy* module of this manual. Crop residue burning must be properly allocated to these two components in order to avoid double counting.

- **CH_4, CO_2, and N_2O emissions from agricultural soils**

Emissions of N_2O from agricultural soils are primarily due to the microbial processes of nitrification and denitrification in the soil. Three types of emission can be distinguished: direct soils emissions, direct soil emissions of N_2O from animal production (including stable emissions to be reported under Manure Management) and indirect emissions. Increases in the amount of nitrogen added to the soil generally result in higher N_2O emissions (Bouwman, 1990). Direct soil emissions may result from the following nitrogen input to soils: (1) synthetic fertilisers, (2) nitrogen from animal waste, (3) biological nitrogen fixation, (4) reutilised nitrogen from crop residues, and (5) sewage sludge application. In addition, cultivation of organic soils may increase soil organic matter mineralisation and, in effect, N_2O emissions. Direct soil emissions of N_2O from animal production include those induced by grazing animals. Emissions from other animal waste management systems are reported under "Manure Management". Indirect N_2O emissions take place after nitrogen is lost from the field as NO_X, NH_3 or after leaching or runoff. Agricultural soils may also emit or remove CO_2 and/or CH_4. For example, peat compost used as a soil amendment in agriculture and gardening may result in CO_2 emissions or removals. Carbon emissions from organic, mineral and limed soils are discussed in Chapter 5.

4.2 Methane And Nitrous Oxide Emissions From Domestic Livestock Enteric Fermentation And Manure Management

4.2.1 Overview of Methane and Nitrous Oxide Emissions from Livestock

This section covers methane and nitrous oxide emissions from enteric fermentation and the management of manure from domestic livestock. Cattle are the most important source of methane from enteric fermentation in most countries because of their high numbers, large size, and ruminant digestive system. Methane emissions from manure management are usually smaller than enteric fermentation emissions, and are principally associated with confined animal management facilities where manure is handled as a liquid. This section presents a brief overview of the key factors affecting methane and nitrous oxide emissions from these sources. The methods for estimating methane emissions are then presented.[1] The method for estimating nitrous oxide emissions from manure management is presented in Section 4.5.3.

Enteric Fermentation

Methane is produced during the normal digestive processes of animals. The amount of methane produced and excreted by an individual animal is dependent primarily on the following:

- **Digestive System**
  The type of digestive system has a significant influence on the rate of methane emission. *Ruminant animals* have the highest emissions because a significant amount of methane-producing fermentation occurs within the rumen. The main ruminant animals are cattle, buffalo, goats, sheep and camels. *Pseudo-ruminant animals* (horses, mules, asses) and *monogastric animals* (swine) have relatively lower methane emissions because much less methane-producing fermentation takes place in their digestive systems.
- **Feed Intake**
  Methane is produced by the fermentation of feed within the animal's digestive system. Generally, the higher the feed intake, the higher the methane emission. Feed intake is positively related to animal size, growth rate, and production (e.g., milk production, wool growth, or pregnancy).

The amount of methane emitted by a population of animals is calculated by multiplying the emission rate per animal by the number of animals. To reflect the variation in emission rates among animal types, the population of animals is divided into subgroups, and an emission rate per animal is estimated for each subgroup. Types of population subgroup are recommended in the method[2].

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[1] All GHG emissions from the burning of animal waste are estimated in Section 1.4; Greenhouse Gas Emissions from Burning Traditional Biomass Fuels. CO_2 from the burning of animal waste is part of a closed cycle and is not counted as net CO_2.

[2] Countries are encouraged to carry out emissions inventory calculations at a finer level of detail if possible. Many countries have available more detailed information than

Human management of wildlife can affect the total number of animals and therefore their emissions, though the associated emissions are believed to be small. The key issue is distinguishing those emissions resulting from human interventions from those emissions that would have occurred naturally. No methodology for estimating these emissions is presented here, though they may be estimated if national experts can fully document their approach, including all assumptions and methods. If these emissions are estimated, they should be reported in the "Other" subcategories of the Enteric Fermentation and Animal Wastes Tables (4 A & B) of *Volume 1: Reporting Instructions*.

Manure Management

Livestock manure is principally composed of organic material. When this organic material decomposes in an anaerobic environment (i.e., in the absence of oxygen), methanogenic bacteria, as part of an interrelated population of micro-organisms, produce methane.

The principal factors affecting methane emission from animal manure are the amount of manure produced and the portion of the manure that decomposes anaerobically. The amount of manure that is produced is dependent on the amount produced per animal and the number of animals. The portion of the manure that decomposes anaerobically depends on how the manure is managed. When manure is stored or treated as a liquid (e.g., in lagoons, ponds, tanks, or pits), it tends to decompose anaerobically and produce a significant quantity of methane. When manure is handled as a solid (e.g., in stacks or pits) or when it is deposited on pastures and rangelands, it tends to decompose aerobically and little or no methane is produced.

To estimate methane emission, the animal population must be divided into subgroups to reflect the varying amounts of manure produced per animal, and the manner in which the manure is handled. Population subgroups are recommended in the method.

Nitrous oxide is formed when manure nitrogen is nitrified or denitrified. The amount of N_2O released depends on the system and duration of waste management. Emissions of N_2O taking place during storage or handling of manure (i.e., before the manure is added to soils) are reported under " Manure Management". Manure-induced N_2O emissions from soils are considered soil emissions (See Section 4.5 of this Reference Manual).

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was used in constructing default values here. Countries may wish to calculate emissions estimates at a finer level of detail by subcategory – further disaggregating recommended activity categories and subcategories – or they may choose to subdivide the categories on some other basis which they feel is appropriate to their particular national circumstances. Working at finer levels of disaggregation does not change the basic nature of the calculations. Once emissions have been calculated at whatever is determined by the national experts to be the most appropriate level of detail, results should also be aggregated up to the minimum standard level of information requested in the IPCC proposed methodology. This will allow for comparability of results among all participating countries. The data and assumptions used for finer levels of detail should also be reported to the IPCC to ensure transparency and replicability of methods. *Volume 1: Reporting Instructions* discusses these issues in more detail.

Box 1

HUMAN WASTE USED AS FERTILISER

Human waste is sometimes used as fertiliser, and can result in emissions or removals of CH_4, N_2O or CO_2. At present, no methodology can be recommended for estimating CH_4 and CO_2 emissions. N_2O emissions from Human waste are described in Chapters 4 & 6. The development of a methodology specifically for this source has been identified as an area for future work. Countries are nevertheless encouraged to estimate emissions from this source if they are able to do so. CH_4 emissions from this source can be estimated in one of at least two ways:

- Emissions from human waste used as fertiliser may be estimated in the present section by adapting the methodology for estimating emissions from livestock manure to use the data provided below. In this case, the estimate should be reported in the "Other" line in Tables 4 A & B in *Volume 1: Reporting Instructions*.

  Bhattacharya et al. (1993) has reported these characteristics of human waste:

  Dry waste per day = 0.107 kg/head/day
  Fractional carbon content = 0.375

  And Thomas (1994, in press) reports the following values:

  Volatile solid production = 0.06 kg/head/day
  Dry matter production = 0.09 kg/head/day
  Fractional carbon content = 4.46% of dry matter

- These emissions can also be estimated using the methodology for sewage treatment in the Waste section. In this case, the emissions should then be treated as wastewater disposed of in aerobic (shallow) ponds, and should be reported in Table 6 B.

In any case, care should be taken to avoid double counting emissions from this source.

4.2.2 Inventory Method for Methane – Overview

The method for estimating methane emission from enteric fermentation and manure management requires three basic steps:

Step 1: Divide the livestock population into subgroups and characterise each subgroup. It is recommended that national experts use three year averages of activity data if available. This is to help prevent bias in the event that the base year of the inventory was an exceptional year not representative of the country's normal activity level.

Step 2: Estimate emission factors for each subgroup in terms of kilograms of methane per animal per year – separate emission factors are required for enteric fermentation and manure.

Step 3: Multiply the subgroup emission factors by the subgroup populations to estimate subgroup emission, and sum across the subgroups to estimate total emission.

These three steps can be performed at varying levels of detail and complexity. This chapter presents the following two approaches:

- **Tier 1**
  A simplified approach that relies on default emission factors drawn from previous studies. The Tier 1 approach is likely to be sufficient for most animal types in most countries.
- **Tier 2**
  A more complex approach that requires country-specific information on livestock characteristics and manure management practices. The Tier 2 approach is recommended when the data used to develop the default values do not correspond well with the country's livestock and manure management conditions. Because cattle characteristics vary significantly by country, it is recommended that countries with large cattle populations consider using the Tier 2 approach for estimating methane emissions from cattle and cattle manure. Similarly, because buffalo and swine manure management practices vary significantly by country, it is recommended that countries with large buffalo and swine populations consider using the Tier 2 approach for estimating methane emissions for manure from these animals.

Some countries for which livestock emissions are particularly important may wish to go beyond the Tier 2 method and incorporate additional country-specific information in their estimates. Although countries are encouraged to go beyond the Tier 2 approach presented below when data are available, these more complex analyses are only briefly discussed here. Table 4-1 summarises the recommended approaches for the livestock emissions included in this inventory.

TABLE 4-1
DOMESTIC LIVESTOCK INCLUDED IN THE METHODS

Livestock	Recommended Emissions Inventory Methods	
	Enteric Fermentation	Manure Management
Dairy Cattle	Tier 2[a]	Tier 2[a]
Non-dairy Cattle	Tier 2[a]	Tier 2[a]
Buffalo	Tier 1	Tier 2[a]
Sheep	Tier 1	Tier 1
Goats	Tier 1	Tier 1
Camels	Tier 1	Tier 1
Horses	Tier 1	Tier 1
Mules and Asses	Tier 1	Tier 1
Swine	Tier 1	Tier 2[a]
Poultry	(Not Estimated)	Tier 1

[a] The Tier 2 approach is recommended for countries with large livestock populations. Implementing the Tier 2 approach for additional livestock subgroups may be desirable when the subgroup emissions are a large portion of total methane emissions for the country.

4.2.3 Inventory Method for Methane – Tier I Approach

This Tier 1 method is simplified so that only readily-available animal population data are needed to estimate emissions. Default emission factors are presented for each of the recommended population subgroups. Each step is discussed in turn.

TIER I: STEP I – LIVESTOCK POPULATIONS

The *average annual population* of livestock is required for each of the livestock categories listed in Table 4-1. In some cases the population fluctuates during the year. For example, a census done before calving will give a much smaller number than a census done after calving. A representative average of the population is therefore needed. In the case of poultry and swine, the number of animals *produced* each year exceeds the annual average population because the animals live for less than 12 months. The population data can be obtained from the FAO Production Yearbook (FAO, 1990) or similar country-specific livestock census reports.

The dairy cattle population is estimated separately from other cattle (see Table 4-2). Dairy cattle are defined in this method as mature cows that are producing milk in commercial quantities for human consumption. This definition corresponds to the dairy cow population reported in the FAO Production Yearbook.

In some countries the dairy cattle population is comprised of two well-defined segments: high-producing "improved" breeds in commercial operations; and low-producing cows managed with traditional methods. These two segments can be combined, or can be evaluated separately by defining two dairy cattle categories. However, the dairy cattle category does not include cows kept principally to produce calves or to provide draft power. Low productivity multi-purpose cows should be considered as non-dairy cattle.

Data on the average milk production of dairy cattle is also required. These data are expressed in terms of kilograms of whole fresh milk produced per year per dairy cow, and can be obtained from the FAO Production Yearbook or similar country-specific reports. If two or more dairy cattle categories are defined, the average milk production per cow is required for each category.

Finally, the livestock populations must be described in terms of warm, temperate, or cool climates for purposes of estimating emissions from livestock manure. Data on the annual average temperature of the regions where livestock are managed should be used as follows:

- Areas with annual average temperatures less than 15°C are defined as cool.
- Areas with annual average temperatures from 15°C to 25°C inclusive are defined as temperate.
- Areas with annual average temperatures greater than 25°C are defined as warm.

For each livestock population, the fraction in each climate should be estimated. These data can be developed from country-specific climate maps and livestock census reports. To the extent possible, the temperature data should reflect the locations where the livestock are managed. If necessary, data from nearby cities can be used. Table 4-2 summarises the animal population data that must be collected in Step 1.

TABLE 4-2

ANIMAL POPULATION DATA COLLECTED IN TIER I STEP I

Livestock	Data Collected				
	Population (# head)	Milk Production (kg/head/yr)	Population By Climate (%)		
			Cool	Temperate	Warm
Dairy Cattle	Average Annual Population	Milk Production per Head	% Cool	% Temp.	% Warm
Non-dairy Cattle	Average Annual Population	Not Applicable (NA)	% Cool	% Temp.	% Warm
Buffalo	Average Annual Population	(NA)	% Cool	% Temp.	% Warm
Sheep	Average Annual Population	(NA)	% Cool	% Temp.	% Warm
Goats	Average Annual Population	(NA)	% Cool	% Temp.	% Warm
Camels	Average Annual Population	(NA)	% Cool	% Temp.	% Warm
Horses	Average Annual Population	(NA)	% Cool	% Temp.	% Warm
Mules and Asses	Average Annual Population	(NA)	% Cool	% Temp.	% Warm
Swine	Average Annual Population	(NA)	% Cool	% Temp.	% Warm
Poultry	Average Annual Population	(NA)	% Cool	% Temp.	% Warm

Data can be obtained from the FAO Production Yearbook and country-specific livestock census reports. Climates are defined in terms of average annual temperature as follows: Cool = less than 15°C; Temperate = from 15°C to 25°C inclusive; Warm = greater than 25°C.

TIER I: STEP 2 – EMISSION FACTORS

The purpose of this step is to select emission factors that are most appropriate for the country's livestock characteristics. Default emission factors for enteric fermentation and manure management have been drawn from previous studies, and are organised by region for ease of use. The basis for the emission factors, described more fully under Tier 2, includes the following:

- *Enteric Fermentation:*
  - *Feed Intake*: Feed intake is estimated based on the energy intake required by the animal for maintenance (the basic metabolic functions needed to stay alive) and production (growth, lactation, work, and gestation). The livestock characteristics required to estimate feed intake are taken from regional and country-specific studies and include: population structure (portion of adults and young), weight, rate of weight gain, amount of work performed, portion of cows giving birth each year, and milk production per cow.

  - *Conversion of Feed Energy to Methane*: The rate at which feed energy is converted to methane is estimated based on the quality of the feed consumed – low quality feed has a slightly higher methane conversion rate. Feed quality is assessed in terms of digestibility on a regional basis.

- *Manure Management:*

  - *Manure Production*: Manure production is estimated based on feed intake and digestibility, both of which are used to develop the enteric fermentation emission factors.

  - *Methane Producing Potential*: Methane producing potential (referred to as B_o) is the maximum amount of methane that can be produced from a given quantity of manure. The methane producing potential varies by animal type and the quality of the feed consumed. Reported measurements for selected animals are used.

  - *Methane Conversion Factor (MCF)*: The MCF defines the portion of the methane producing potential (B_o) that is achieved. The MCF varies with the manner in which the manure is managed and the climate, and can theoretically range from 0 to 100 per cent. Manure managed as a liquid under hot conditions promotes methane formation and emissions. These manure management conditions have high MCFs, of 65 to 90 per cent. Manure managed as dry material in cold climates does not readily produce methane, and consequently has an MCF of about 1 per cent. Laboratory measurements were used to estimate MCFs for the major manure management techniques.

  - *Manure Management Practices*: Regional assessments of manure management practices are used to estimate the portion of the manure that is handled with each manure management technique.

The data used to estimate the default emission factors for enteric fermentation and manure management are presented in Appendix A and Appendix B respectively, at the end of this section.

Table 4-3 shows the enteric fermentation emission factors for each of the animal types except cattle. As shown in the table, emission factors for sheep and swine vary for developed and developing countries. The differences in the emission factors are driven by differences in feed intake and feed characteristic assumptions (see Appendix A). Although point estimates are given for the emission factors, an uncertainty of about ±20 per cent exists due to variations in animal management and feeding. Deviations from the emission factors can be larger than 20 per cent under specialised feeding or management conditions.

Table 4-4 presents the enteric fermentation emission factors for cattle. A range of emission factors is shown for typical regional conditions. As shown in the table, the emission factors vary by over a factor of four on a per head basis.

While the default emission factors shown in Table 4-4 are broadly representative of the emission rates within each of the regions described, emission factors vary among countries within regions. Also, as with the emission factors shown in Table 4-3, an uncertainty of about ±20 per cent exists due to variations in animal management and feeding. Animal size and milk production are important determinants of emission rates for dairy cows. Relatively smaller dairy cows with low levels of production are found in Asia, Africa, and the Indian subcontinent. Relatively larger dairy cows with high levels of production are found in North America and Western Europe.

TABLE 4-3
ENTERIC FERMENTATION EMISSION FACTORS
(KG PER HEAD PER YR)

Livestock	Developed Countries	Developing Countries
Buffalo	55	55
Sheep	8	5
Goats	5	5
Camels	46	46
Horses	18	18
Mules and Asses	10	10
Swine	1.5	1.0
Poultry	Not Estimated	Not Estimated

All estimates are ± 20 %

Sources: Emission factors for buffalo and camels from Gibbs and Johnson (1993). Emission factors for other livestock from Crutzen et al. (1986).

Animal size and population structure are important determinants of emission rates for non-dairy cattle. Relatively smaller non-dairy cattle are found in Asia, Africa, and the Indian subcontinent. Also, many of the non-dairy cattle in these regions are young. Non-dairy cattle in North America, Western Europe and Oceania are larger, and young cattle constitute a smaller portion of the population[3].

Select emission factors from Tables 4-3 and 4-4 by identifying the region most applicable to the country being evaluated. The data collected on the average annual milk production by dairy cows should be used to help select a dairy cow emission factor. If necessary, interpolate between dairy cow emission factors shown in the table using the data collected on average annual milk production per head.

Table 4-5 shows the default manure management emission factors for each animal type except cattle, buffalo, and swine. Separate emission factors are shown for developed and developing countries, reflecting the general differences in feed intake and feed characteristics of the animals in the two regions. These emission factors reflect the fact that virtually all the manure from these animals is managed in dry manure management systems, including pastures and ranges, drylots, and daily spreading on fields (Woodbury and Hashimoto, 1993).

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[3] For each animal category, it is important to use the average weight of the animal during the year for estimating emissions. Because the weights of mature animals may fluctuate seasonally, a representative weight should be selected that considers conditions throughout the year. For growing animals, the average weight is generally less than the final (or end) weight of the animal at the end of the year. Growth rate statistics should be used to estimate the average weight during the year for purposes of estimating emissions.

TABLE 4-4
ENTERIC FERMENTATION EMISSION FACTORS FOR CATTLE

Regional Characteristics	Cattle Type	Emission Factor (kg/head/yr)	Comments
***North America*:** Highly productive commercialised dairy sector feeding high quality forage and grain. Separate beef cow herd, primarily grazing with feed supplements seasonally. Fast-growing beef steers/heifers finished in feedlots on grain. Dairy cows are a small part of the population.	Dairy	118	Average milk production of 6,700 kg/head/yr
	Non-dairy	47	Includes beef cows, bulls, calves, growing steers/heifers, and feedlot cattle.
***Western Europe*:** Highly productive commercialised dairy sector feeding high quality forage and grain. Dairy cows also used for beef calf production. Very small dedicated beef cow herd. Minor amount of feedlot feeding with grains.	Dairy	100	Average milk production of 4,200 kg/head/yr.
	Non-dairy	48	Includes bulls, calves, and growing steers/heifers.
***Eastern Europe*:** Commercialised dairy sector feeding mostly forages. Separate beef cow herd, primarily grazing. Minor amount of feedlot feeding with grains.	Dairy	81	Average milk production of 2,550 kg/head/yr.
	Non-dairy	56	Includes beef cows, bulls, and young.
***Oceania*:** Commercialised dairy sector based on grazing. Separate beef cow herd, primarily grazing rangelands of widely varying quality. Growing amount of feedlot feeding with grains. Dairy cows are a small part of the population.	Dairy	68	Average milk production of 1,700 kg/head/yr.
	Non-dairy	53	Includes beef cows, bulls, and young.
***Latin America*:** Commercialised dairy sector based on grazing. Separate beef cow herd grazing pastures and rangelands. Minor amount of feedlot feeding with grains. Growing non-dairy cattle comprise a large portion of the population.	Dairy	57	Average milk production of 800 kg/head/yr.
	Non-dairy	49	Includes beef cows, bulls, and young.
***Asia*:** Small commercialised dairy sector. Most cattle are multi-purpose, providing draft power and some milk within farming regions. Small grazing population. Cattle of all types are smaller than those found in most other regions.	Dairy	56	Average milk production of 1,650 kg/head/yr.
	Non-dairy	44	Includes multi-purpose cows, bulls, and young
***Africa and Middle East*:** Commercialised dairy sector based on grazing with low production per cow. Most cattle are multi-purpose, providing draft power and some milk within farming regions. Some cattle graze over very large areas. Cattle of all types are smaller than those found in most other regions.	Dairy	36	Average milk production of 475 kg/head/yr.
	Non-dairy	32	Includes multi-purpose cows, bulls, and young
***Indian Subcontinent*:** Commercialised dairy sector based on crop by-product feeding with low production per cow. Most bullocks provide draft power and cows provide some milk in farming regions. Small grazing population. Cattle in this region are the smallest compared to cattle found in all other regions.	Dairy	46	Average milk production of 900 kg/head/yr.
	Non-dairy	25	Includes cows, bulls, and young. Young comprise a large portion of the population

TABLE 4-5
MANURE MANAGEMENT EMISSION FACTORS
(KG PER HEAD PER YR)

Livestock	Developed Countries			Developing Countries		
	Cool	Temp.[a]	Warm	Cool	Temp.[a]	Warm
Sheep	0.19	0.28	0.37	0.10	0.16	0.21
Goats	0.12	0.18	0.23	0.11	0.17	0.22
Camels	1.6	2.4	3.2	1.3	1.9	2.6
Horses	1.4	2.1	2.8	1.1	1.6	2.2
Mules and Asses	0.76	1.14	1.51	0.60	0.90	1.2
Poultry[b]	0.078	0.117	0.157	0.012	0.018	0.023

The range of estimates reflects cool to warm climates. Climate regions are defined in terms of annual average temperature as follows: Cool = less than 15°C; Temperate = 15°C to 25°C inclusive; and Warm = greater than 25°C. The Cool, Temperate and Warm regions are estimated using MCFs of 1 %, 1.5 % and 2 %, respectively.

a Temp. = Temperate climate region.

b Chickens, ducks, and turkeys.

All estimates are ±20 %.

Sources: Emission factors developed from: feed intake values and feed digestibilities used to develop the enteric fermentation emission factors (see Appendix A); MCF, and B_o values reported in Woodbury and Hashimoto (1993). All manure is assumed to be managed in dry systems, which is consistent with the manure management system usage reported in Woodbury and Hashimoto (1993).

The ranges of values shown in Table 4-5 reflect the range of Methane Conversion Factor values of 1 to 2 per cent. The higher value is appropriate for manure managed in warm climates, while the lower value is appropriate for manure managed in cooler and dryer climates. A middle value is assigned to temperate conditions. The uncertainty in the emission factors remains substantial, however, because field measurements are required to validate the laboratory measurements that form the basis for the MCFs used in the analysis. Appendix B, at the end of this section, summarises the data used to estimate the emission factors shown in Table 4-6.

The climate data collected in Step 1 is used to select the emission factors from Table 4-6. A weighted average emission factor for each animal type is computed by multiplying the percentages of the animal populations in each climate region by the emission factor for each climate region. For example, if sheep in a developing country were 25 per cent in a temperate region and 75 per cent in a warm region, the emission factor for sheep would be estimated at about 0.2 kg/head/yr as follows:

$$\text{Emission Factor} = (25\% \times 0.16) + (75\% \times 0.21) = 0.1975 \text{ kg/head/yr.}$$

An alternative way of handling these calculations is to sub-divide the category of sheep into two populations: one in warm and one in temperate region. Calculations could then be done separately and summed.

Because the manure from cattle, buffalo, and swine is managed in a variety of ways, including both dry and liquid systems, the variations in manure management practices among regions and countries must be considered to develop emission factors for these animals. Table 4-6 presents emission factors based on regional manure management practices described in Safley et al. (1992).

TABLE 4-6
MANURE MANAGEMENT EMISSION FACTORS FOR CATTLE, SWINE AND BUFFALO

Regional Characteristics	Livestock Type	Emission Factor by Climate Region[a] (kg/head/yr)		
		Cool	Temperate	Warm
North America: Liquid-based systems are commonly used for dairy and swine manure. Non-dairy manure is usually managed as a solid and deposited on pastures or ranges.	Dairy Cattle	36	54	76
	Non-dairy Cattle	1	2	3
	Swine	10	14	18
Western Europe: Liquid/slurry and pit storage systems are commonly used for cattle and swine manure. Limited cropland is available for spreading manure.	Dairy Cattle	14	44	81
	Non-dairy Cattle	6	20	38
	Swine	3	10	19
	Buffalo	3	8	17
Eastern Europe: Solid based systems are used for the majority of manure. About one-third of livestock manure is managed in liquid-based systems.	Dairy Cattle	6	19	33
	Non-dairy Cattle	4	13	23
	Swine	4	7	11
	Buffalo	3	9	16
Oceania: Virtually all livestock manure is managed as a solid on pastures and ranges. About half of the swine manure is managed in anaerobic lagoons.	Dairy Cattle	31	32	33
	Non-dairy Cattle	5	6	7
	Swine	20	20	20
Latin America: Almost all livestock manure is managed as a solid on pastures and ranges. Buffalo manure is deposited on pastures and ranges.	Dairy Cattle	0	1	2
	Non-dairy Cattle	1	2	1
	Swine	0	1	2
	Buffalo	1	1	2
Africa: Almost all livestock manure is managed as a solid on pastures and ranges.	Dairy Cattle	1	1	1
	Non-dairy Cattle	0	1	1
	Swine	0	1	2
Middle East: Over two-thirds of cattle manure is deposited on pastures and ranges. About one-third of swine manure is managed in liquid-based systems. Buffalo manure is burned for fuel or managed as a solid.	Dairy Cattle	1	2	2
	Non-Dairy Cattle	1	1	1
	Swine	1	3	6
	Buffalo	4	5	5
Asia: About half of cattle manure is used for fuel with the remainder managed in dry systems. Almost 40% of swine manure is managed as a liquid. Buffalo manure is managed in drylots and deposited in pastures and ranges.	Dairy Cattle	7	16	27
	Non-dairy Cattle	1	1	2
	Swine	1	4	7
	Buffalo	1	2	3
Indian Subcontinent: About half of cattle and buffalo manure is used for fuel with the remainder managed in dry systems. About one-third of swine manure is managed as a liquid.	Dairy Cattle	5	5	6
	Non-dairy Cattle	2	2	2
	Swine	3	4	6
	Buffalo	4	5	5

[a] Cool climates have an average temperature below 15°C; temperate climates have an average temperature from 15°C to 25°C inclusive; warm climates have an average temperature above 25°C. All climate categories are not necessarily represented within every region. For example, there are no significant warm areas in Eastern or Western Europe. Similarly, there are no significant cool areas in Africa and the Middle East. See Appendix B for the derivation of these emission factors.

Note: Significant buffalo populations do not exist in North America, Oceania, or Africa.

As shown in the table, the emission factors for dairy cattle range between 81 kg/head/yr in warm parts of Western Europe to 0 kg/head/yr in cool parts of Latin America. The emission factors for non-dairy cattle range between 38 kg/head/yr in warm parts of Western Europe to 1 kg/head/yr in cool parts of North America and Latin America. In addition to climate, the range of emission factors is due to the manure management practices used in each region. For example, the emission factors for North American dairy cattle manure and European dairy and non-dairy cattle manure are relatively high because the manure is often managed using liquid systems that promote methane production. The emission factors for North American non-dairy cattle and for all animals in Africa and the Middle East are relatively low because their manure is generally managed using dry systems that do not promote methane production.

To select emission factors from Table 4-6, first identify the appropriate region, such as Latin America. Within that region, identify the animal type of interest. For that animal type three values are given for the three climate regions. Compute a weighted average emission factor for the animal type by multiplying the percentages of the animal population in each climate region by the emission factor for each climate region. Appendix B summarises the estimates of manure management system usage and MCFs that underlie the emission factors in Table 4-6.

As with the other manure management emission factors, there is substantial uncertainty in the estimates shown in Table 4-6 because field measurements are required to validate the laboratory measurements that form the basis for the MCFs used in the analysis, and because there is uncertainty and variability in the manner in which manure is managed in each region.

TIER 1: STEP 3 – TOTAL EMISSION

To estimate total emission the selected emission factors are multiplied by the associated animal population and summed. The emission estimates should be reported in gigagrams (Gg). Because the emission factors are reported in kilograms per head per year, the total emissions in Gg is estimated as follows for each animal category:

$$\text{emission factor (kg/head/yr) x population (head) / (}10^6\text{ kg/Gg)} \\ = \text{emissions Gg/yr.}$$

As a point of reference, in 1990 total annual global methane emissions from domestic livestock enteric fermentation were of the order of 0.060 to 0.100 Gg (Gibbs and Johnson, 1993). Enteric fermentation emissions from countries with large populations of livestock may be on the order of 0.001 to 0.005 Gg per year. Countries with smaller populations of livestock would likely have emissions of less than 0.001 Gg per year.

In 1990 total annual global methane emissions from manure management was on the order of 0.010 to 0.018 Gg (Woodbury and Hashimoto, 1993). Manure management emissions from countries where manure is managed in liquid-based systems may be on the order of 0.001 to 0.002 Gg per year. Countries where manure is not managed in liquid-based systems would likely have emissions of much less than 0.001 Gg per year.

4.2.4 Tier 2 Approach for Methane Emissions From Enteric Fermentation

The Tier 2 approach is recommended for estimating methane emissions from enteric fermentation from cattle for those countries with large cattle populations. As contrasted with the Tier 1 method, this approach requires much more detailed information on the cattle population. Using this detailed information, more precise estimates of the cattle emission factors are developed. When the Tier 2 method is used the default emission factors listed in Tier 1 for cattle are not used.

This Tier 2 approach is similar to the August 1991 OECD method (OECD, 1991), with some modifications:

- The Blaxter and Clapperton (1965) equation is replaced with a recommended set of methane conversion rate "rules of thumb."
- Feed energy intake requirements for pregnancy have been added.
- The energy requirements required for grazing have been reduced based on newly available data from AAC (1990).
- The equations used to relate gross energy intake to net energy used by the animal have been made more general to fit a wider variety of feed conditions.

The three steps outlined for Tier 1 are also used here.

ENTERIC FERMENTATION
TIER 2: STEP I – LIVESTOCK POPULATION

To develop precise estimates of emissions, cattle should be divided into categories of relatively homogeneous groups. For each category a representative animal is chosen and characterised for the purpose of estimating an emission factor. Table 4-7 presents a set of recommended representative cattle types. Three main categories, Mature Dairy Cattle, Mature Non-dairy Cattle, and Young Cattle, are recommended as the minimum set of representative types. The subcategories listed should be used when data are available. In particular, the sub-population of cows providing milk to calves should be identified among non-dairy cattle because the feed intake necessary to support milk production can be substantial. In some countries the feedlot category is needed so that the implications of the high-grain diets can be incorporated.

TABLE 4-7
RECOMMENDED REPRESENTATIVE CATTLE TYPES

Main Categories	Subcategories
Mature Dairy Cattle	Dairy Cows used principally for commercial milk production
Mature Non-dairy Cattle	Mature Females: •Beef Cows: used principally for producing beef steers and heifers •Multiple-Use Cows: used for milk production, draft power, and other uses Mature Males: •Breeding Bulls: used principally for breeding purposes •Draft Bullocks: used principally for draft power
Young Cattle	Pre-Weaned Calves Growing Heifers, Steers/Bullocks and Bulls Feedlot-Fed Steers and Heifers on High-Grain Diets

For each of the representative animal types defined, the following information is required:

- annual average population (number of head);
- average daily feed intake (megajoules (MJ) per day and kg per day of dry matter); and
- methane conversion rate (percentage of feed energy converted to methane).

Generally, data on average daily feed intake are not available, particularly for grazing animals. Consequently, the following data should be collected for estimating the feed intake for each representative animal type[4]:

- weight (kg);
- average weight gain per day (kg);[5]
- feeding situation: confined animals; animals grazing good quality pasture; and animals grazing over very large areas;
- milk production per day (kg/day);[6]
- average amount of work performed per day (hours/day);
- percentage of cows that give birth in a year;[7] and
- feed digestibility (%).[8]

These data should be obtained from country-specific cattle evaluations. Some data, such as weight, weight gain, and milk production, may be available from production statistics. Care should be taken to use the live cattle weights, as contrasted with slaughter weights. Appendix A, at the end of this section, lists the data used to develop the default emission factors presented in Tier 1. Individual country data can be compared to the data presented in Appendix A to ensure that the data collected are reasonable.

Data on methane conversion rates are also not generally available. The following rules of thumb are recommended for the methane conversion rates:

- Developed Countries. A 6 per cent conversion rate (±0.5 per cent) is recommended for all cattle in developed countries except feedlot cattle consuming diets with a large quantity of grain. For feedlot cattle on high grain diets a rate of 4 per cent (±0.5 per cent) is recommended. In circumstances where good feed is available (i.e., high digestibility and high energy value) the lower bounds of these ranges can be used. When poorer feed is available, the higher bounds are more appropriate.
- Developing Countries. Several recommendations are made for different animal management situations in developing countries:

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[4] In many, if not most, cattle management circumstances, the principal driving factors that affect feed intake are: weight, milk production and feed digestibility.

[5] This may be assumed to be zero for mature animals.

[6] Milk production is required for dairy cows and non-dairy cows providing milk to calves.

[7] This is only relevant for mature cows.

[8] Feed digestibility is defined as the proportion of energy in the feed that is not excreted in the faeces. Digestibility is commonly expressed as a percentage (%). Common ranges for feed digestibility for cattle are 50% to 60% for crop by-products and rangelands; 60% to 70% for good pastures, good preserved forages, and grain-supplemented forage-based diets; and 75% to 85% for grain-based diets fed in feedlots.

  - All dairy cows and young cattle are recommended to have a conversion rate of 6.0 per cent (±0.5 per cent). These cattle are generally the best-fed cattle in these regions.
  - All non-dairy cattle, other than young stall-fed animals, consuming low-quality crop by-products, are recommended to have a conversion rate of 7.0 per cent (±0.5 per cent) because feed resources are particularly poor in many cases in these regions.
  - Grazing cattle are recommended to have a conversion rate of 6.0 per cent (±0.5 per cent), except for grazing cattle in Africa, which are recommended to have a rate of 7.0 per cent (±0.5 per cent) because of the forage characteristics found in many portions of tropical Africa.

These rules of thumb are a rough guide based on the general feed characteristics and production practices found in many developed and developing countries. Country-specific exceptions to these general rules of thumb should be taken into consideration as necessary based on detailed data from cattle experts.

ENTERIC FERMENTATION
TIER 2: STEP 2 – EMISSION FACTORS

The emission factors for each category of cattle are estimated based on the feed intake and methane conversion rate for the category. Feed intake is estimated based on the feed energy requirements of the representative animals, subject to feed-intake limitations. The net energy system described in NRC (1984 and 1989) is recommended as the starting point for the estimates. Because the NRC system was developed for feeding conditions in temperate regions, several adjustments were made to avoid potential biases when applied to evaluate feed-energy intakes for tropical cattle (see Appendix C). Comparisons with alternative feeding systems (e.g., ARC, 1980) indicate that the emissions estimates are not sensitive to the feeding system used as the basis for making the estimates.

The net energy system specifies the amount of feed energy required for the physiological functions of cattle, including maintenance, growth and lactation. Feed energy requirements for work have also been estimated, and are included in this analysis for the draft animals in developing countries. Energy requirements for pregnancy have also been added for the portion of cows that give birth in each year. The following information is required to estimate feed energy intakes:

- *Maintenance*
  Maintenance refers to the apparent feed energy required to keep the animal in energy equilibrium, i.e., there is no gain or loss of energy in the body tissues (Jurgens, 1988). For cattle, net energy for maintenance (NE_m) has been estimated to be a function of the weight of the animal raised to the 0.75 power (NRC, 1984):

EQUATION 1

$$NE_m \text{ (MJ/day)} = 0.322 \times (\text{weight in kg})^{0.75}$$

NRC (1989) recommends that lactating dairy cows be allowed a slightly higher maintenance allowance:

$$NE_m \text{ (MJ/day)} = 0.335 \times (\text{weight in kg})^{0.75} \quad \text{dairy cows\}}$$

- *Feeding*

  Additional energy is required for animals to obtain their food. Grazing animals require more energy for this activity than do stall-fed animals. The following energy requirements are added for this activity based on their feeding situation:[9]

  EQUATION 2

  NE_{feed} =

  Confined animals (pens and stalls): no additional NE_m;

  Animals grazing good quality pasture: 17 % of NE_m; and

  Animals grazing over very large areas: 37 % of NE_m.

- *Growth*

  The energy requirements for growth can be estimated as a function of the weight of the animal and the rate of weight gain. NRC (1989) presents formulae for large- and small-frame males and females, the estimates of which vary by about ± 25 per cent. The equation for large-frame females is recommended, which is about the average for the four types:

  EQUATION 3

  $$NE_g \text{ (MJ/day)} = 4.18 \times \{(0.035\ W^{0.75} \times WG^{1.119}) + WG\}$$

  where:

  W = animal weight in kilograms (kg); and

  WG = weight gain in kg per day.

  The relationships for NE_g were developed for temperate agriculture conditions, and may over-estimate energy requirements for tropical conditions, particularly for draft animals that may have a lower fat content in their weight gain (Graham, 1985). However, no data are available for improving the estimates at this time.

- *Lactation*

  Net energy for lactation has been expressed as a function of the amount of milk produced and its fat content (NRC, 1989):

  EQUATION 4

  $$NE_l \text{ (MJ/day)} = \text{kg of milk/day} \times (1.47 + 0.40 \times \text{Fat \%})$$

  At 4.0 per cent fat, the NE_l in MJ/day is about 3.1 x kg of milk per day.

- *Draft Power*

  Various authors have summarised the energy intake requirements for providing draft power (e.g., Lawrence, 1985; Bamualim and Kartiarso, 1985; and Ibrahim, 1985). The strenuousness of the work performed by the animal influences the energy requirements, and consequently a wide range of energy requirements have been estimated. The values by Bamualim and Kartiarso show that about 10 per cent of

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[9] The original OECD method recommended slightly higher energy additions. These revised figures are based on newly-published information in AAC (1990).

NE_m requirements are required per hour of typical work for draft animals. This value is used as follows:

EQUATION 5

$$NE_{draft}\ (MJ/day) = 0.10 \times NE_m \times \text{hours of work per day}$$

- *Pregnancy*
  Daily energy requirements for pregnancy are presented in NRC (1984). Integrating these requirements over a 281-day gestation period yields the following equation:

EQUATION 6

$$NE_{pregnancy}\ (MJ/281\text{-day period}) = 28 \times \text{calf birth weight in kg}$$

The following equation can be used to estimate the approximate calf birth weight as a function of the cow's weight:[10]

EQUATION 7

$$\text{Calf birth weight (kg)} = 0.266 \times (\text{cow weight in kg})^{0.79}$$

Manipulating Equations 6 and 7, in conjunction with Equation 1, shows that the NE required for pregnancy is about 7.5 per cent of NE_m for the range of cow sizes considered in this analysis. Therefore, a factor of 7.5 per cent of NE_m is added to account for the energy required for pregnancy for the portion of cows giving birth each year.

Based on these equations, each of the net energy components for each of the cattle categories can be estimated from the data collected in Step 1: weight in kilograms; feeding situation; weight gain per day in kilograms; milk production in kilograms of 4 per cent fat-corrected milk; number of hours of work performed per day; and portion that give birth.

These net energy requirements must be translated into gross energy intakes. Also, by estimating the gross energy intake, the net energy estimates can be checked for reasonableness against expected ranges of feed intake as a percentage of animal weight. To estimate gross energy intake, the relationship between the net energy values and gross energy values of different feeds must be considered. This relationship can be summarised briefly as follows:

Digestible Energy = Gross Energy - Faecal Losses

Metabolisable Energy = Digestible Energy - Urinary and Combustible Gas Losses

Net Energy = Metabolisable Energy - Heat Increment

Net Energy = Gross Energy - Faecal Losses - Urinary and Combustible Gas Losses - Heat Increment

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[10] This species-specific equation from Robbins and Robbins (1979) was adjusted to the mean cow and calf weight of a typical beef breed of cattle. This adjustment increases the coefficient in the equation from 0.214 to 0.266.

The quantitative relationship among these energy values varies among feed types. Additionally, the values depend on how the feeds are prepared and fed, and the level at which they are fed. For the purposes of this method, simplifying assumptions are used to derive a relationship between net energy and digestible energy that is reasonably representative for the range of diets typically fed to cattle. Gross energy intake is then estimated using this relationship and the digestibility data collected in Step 1.

Given the digestibility of the feed (defined in Step 1), a general relationship between digestible energy and metabolisable energy can be used as follows (NRC, 1984):

EQUATION 8

$$\text{Metabolisable Energy (ME)} = 0.82 \times \text{Digestible Energy (DE)}$$

Equation 8 is a simplified relationship; larger (smaller) methane conversion rates would tend to reduce (increase) the coefficient to values below (above) 0.82.

NRC (1984) presents separate quantitative relationships between metabolisable energy and net energy used for growth versus net energy used for other functions. Using Equation 8, the NRC relationships can be re-arranged to quantify the ratio of NE to DE, as follows:

EQUATION 9

$$NE/DE = 1.123 - (4.092 \times 10^{-3} \times DE\%) + (1.126 \times 10^{-5} \times (DE\%)^2) - 25.4/DE\%$$

EQUATION 10

$$NE_g/DE = 1.164 - (5.160 \times 10^{-3} \times DE\%) + (1.308 \times 10^{-5} \times (DE\%)^2) - 37.4/DE\%$$

where:

NE/DE = the ratio of net energy consumed for maintenance, lactation, work and pregnancy to digestible energy consumed;

NE_g/DE = the ratio of net energy consumed for growth to digestible energy consumed; and

DE% = digestible energy as percentage of gross energy, expressed in per cent (e.g., 65%).

Because the NRC (1984) relationships were developed based on diets with relatively high digestibilities (generally above 65 per cent), they may not be appropriate for the relatively low digestibility diets that are commonly found in tropical livestock systems. In particular, the non-linear nature of the relationships could appear to increase the estimates of feed intake for low-digestibility feeds. An apparent increase in feed intake would lead to an apparent increase in emissions estimates.

Based on a review of other energy systems (e.g., ARC, 1980), a linear relationship between digestible energy and net energy was derived for digestibilities below 65 per cent as follows (see Appendix C):

EQUATION 11

$$NE/DE = 0.298 + (0.00335 \times DE\%)$$

EQUATION 12

$$NE_g/DE = -0.036 + (0.00535 \times DE\%)$$

Given the estimates for feed digestibility (from Step 1) and equations 9 through 12, the gross energy intake (GE in MJ/day) can be estimated as follows:

EQUATION 13

$$GE = \frac{\left(NE_m + NE_{feed} + NE_l + NE_{draft} + NE_{pregnancy}\right) \times \left(100 / DE\%\right)}{\left(NE / DE\right) + \left(NE_a / \{NE_g / DE\}\right)}$$

where:

$\{NE/DE\}$ is computed from equation 9 for digestibility greater than 65 per cent and from equation 11 for digestibility less than or equal to 65 per cent;

$\{NE_g/DE\}$ is computed from equation 10 for digestibility greater than 65 per cent and from equation 12 for digestibility less than or equal to 65 per cent; and

DE% is digestibility in per cent (e.g., 60%).

To check the estimate of daily gross energy intake from Equation 13, the estimate can be converted in daily intake in kilograms by dividing by 18.45 MJ/kg. This estimate of intake in kilograms should generally be between 1.5 per cent and 3.0 per cent of the animal's weight.

Using Equation 13 and the cattle data summarised in Appendix A, Gibbs and Johnson (1993) found that the intake estimates are consistent with expected intakes as a percentage of body weight and previously published values. For example, the intake estimate for Indian cattle is the equivalent of about 10,000 MJ per year of metabolisable energy (ME). Winrock (1978) estimates the average ME requirements for Indian cattle at 10,600 MJ per year. Similarly, the ME values implied for U.S.A dairy and non-dairy cows are 58,000 MJ and 31,000 MJ per year, respectively, which are similar to estimates of 62,000 MJ and 31,700 MJ derived in US EPA (1993). Consequently, for a diverse set of conditions, the intake estimates correspond to reasonably expected ranges from previously published estimates.

To estimate the emission factor for each cattle type, the feed intake is multiplied by the methane conversion rate (from Step 1) as follows:

EQUATION 14

$$\text{Emissions (kg/yr)} = [\text{Intake (MJ/day)} \times Y_m \times (365\ \text{days/yr})] / [55.65\ \text{MJ/kg of methane}]$$

where Y_m is the methane conversion rate expressed in decimal form (such as 0.06 for 6 per cent). The result of this step of the method is an emission factor for each cattle type defined in Step 1.

ENTERIC FERMENTATION
TIER 2: STEP 3 – TOTAL EMISSIONS

To estimate total emissions the selected emission factors are multiplied by the associated animal population and summed. As described above under Tier 1, the emissions estimates should be reported in gigagrams (Gg).

4.2.5 Tier 2 Approach for Methane Emissions from Manure Management

The Tier 2 approach provides a more detailed method for estimating methane emissions from manure management systems. The Tier 2 approach is recommended for countries with large cattle, buffalo and swine populations managed under confined conditions. Compared to the Tier 1 approach, this method requires additional detailed information on animal characteristics and the manner in which manure is managed. Using this additional information, emission factors are estimated that are specific to the conditions of the country, and the default emission factors from Tier 1 are not used.

The Tier 2 approach is similar to the original OECD method described in OECD (1991). Improvements to the method have been made to incorporate more recent figures on methane conversion factors and to link the method more closely to the animal characteristic data collected for estimating enteric fermentation.

MANURE MANAGEMENT
TIER 2: STEP 1 – LIVESTOCK POPULATIONS

To develop precise estimates of emissions, the animals should be divided into relatively homogeneous groups. For each category a representative animal is chosen and characterised for purposes of estimating an emission factor. Suggested categories for cattle are discussed above under the enteric fermentation Tier 2 method and are summarised in Table 4-7. Similar categories can be used for buffalo. Categories for swine could include sows, boars, and growing animals (farrows to finishers). For each of the representative animal types defined, the following information is required:

- annual average population (number of head) by climate region (cool, temperate, and warm);
- average daily volatile solids (VS) excretion (kg of dry matter per day);[11]

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[11] Volatile solids (VS) are the degradable organic material in livestock manure.

- methane-producing potential (B_o) of the manure (cubic metres (m^3) of methane per kg of VS);
- manure management system usage (percentage of manure managed with each management system).

Population data are generally available from country-specific livestock census reports. As described above under Tier 1, the portion of each animal population in cool, temperate, and warm climate regions is required.

Often, data on average daily VS excretion are not available. Consequently, the VS values may need to be estimated from feed intake levels. The enteric fermentation Tier 2 method should be used to estimate feed intake levels for cattle and buffalo.[12] For swine, country-specific swine production data may be required to estimate feed intake. To develop the default emission factors for swine presented in Tier 1, average feed intake estimates for swine in developed and developing countries were used from Crutzen et al. (1986) (see Appendix B, at the end of this section).

Once feed intake is estimated, the VS excretion rate is estimated as:[13]

EQUATION 15

$$\text{VS (kg dm/day)} = \text{Intake (MJ/day)} \times (1\ \text{kg}/18.45\ \text{MJ}) \times (1 - \text{DE\%}/100) \times (1 - \text{ASH\%}/100)$$

where:

VS = VS excretion per day on a dry weight basis;

dm = dry matter;

Intake = the estimated daily average feed intake in MJ/day;

DE% = the digestibility of the feed in per cent (e.g., 60%);

ASH% = the ash content of the manure in per cent (e.g., 8%).

For cattle, the DE% value used should be the same value used to implement Tier 2 for enteric fermentation. The ash content of cattle and buffalo manure is generally around 8 per cent. For swine, the default emission factors were estimated using 75 per cent and 50 per cent digestibility for developed and developing countries, respectively, and an ash content of 2 per cent and 4 per cent for developed and developing countries, respectively. Appendix B summarises the data used to estimate the VS excretion rates for cattle, buffalo, and swine.

The maximum methane-producing capacity for the manure (B_o) varies by species and diet. Country-specific data should be used where feasible. A range of representative B_o values

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[12] By using the enteric fermentation Tier 2 method to estimate feed intake, consistency is assured in the data underlying the emissions estimates for both enteric fermentation and manure management.

[13] The energy density of feed is about 18.45 MJ per kg of dry matter. This value is relatively constant across a wide range of forage and grain-based feeds commonly consumed by livestock.

for cattle, buffalo, and swine populations were used to develop the default emission factors as follows (see Appendix B):

- Dairy Cattle
  - Developed Countries: 0.24 m^3/kg VS
  - Developing Countries: 0.13 m^3/kg VS
- Non-dairy Cattle
  - Developed Countries: 0.17 m^3/kg VS
  - Developing Countries: 0.10 m^3/kg VS
- Buffalo in all regions: 0.10 m^3/kg VS
- Swine
  - Developed Countries: 0.45 m^3/kg VS
  - Developing Countries: 0.29 m^3/kg VS

The portion of manure managed in each manure management system must also be collected for each representative animal type. Table 4-8 summarises the main types of manure management systems. The first four types in the table, pasture, daily spread, solid storage, and drylot, are all dry manure management systems. These systems produce little or no methane. The wet manure management systems, liquid/slurry, anaerobic lagoon, and pit storage, are the primary sources of manure methane emissions. To implement this Tier 2 method, at a minimum the proportion of manure managed in wet and dry systems must be estimated.

TABLE 4-8
MANURE MANAGEMENT SYSTEMS AND METHANE CONVERSION FACTORS (MCFS)

System		MCF by Climate[a]			Source
		Cool	Temperate	Warm	
Pasture/Range/Paddock: the manure from pasture and range grazing animals is allowed to lie as is, and is not managed.		1%	1.5%	2%	b
Daily Spread: manure is collected in solid form by some means such as scraping. The collected manure is applied to fields regularly (usually daily).		0.1%	0.5%	1%	b
Solid Storage: manure is collected as in the daily spread system, but is stored in bulk for a long period of time (months) before any disposal.		1%	1.5%	2%	b
Drylot: in dry climates animals may be kept on unpaved feedlots where the manure is allowed to dry until it is periodically removed. Upon removal the manure may be spread on fields.		1%	1.5%	5%	b
Liquid/Slurry: these systems are characterised by large concrete lined tanks built into the ground. Manure is stored in the tank for six or more months until it can be applied to fields. To facilitate handling as a liquid, water may be added to the manure.		10%	35%	65%	b
Anaerobic Lagoon: anaerobic lagoon systems are characterised by flush systems that use water to transport manure to lagoons. The manure resides in the lagoon for periods from 30 days to over 200 days. The water from the lagoon may be recycled as flush water or used to irrigate and fertilise fields.		90%	90%	90%	c
Pit Storage: liquid swine manure may be stored in a pit while awaiting final disposal. The length of storage time varies, and for this analysis is divided into two categories: less than one month or greater than one month.	< 30 Days	5%	18%	33%	b
	> 30 Days	10%	35%	65%	b
Anaerobic Digester: the manure, in liquid or slurry form, is anaerobically digested to produce methane gas for energy. Emissions are from leakage and vary with the type of digester.		5-15%	5-15%	5-15%	d
Burned for Fuel: manure is collected and dried in cakes and burned for heating or cooking. Emissions occur while the manure is stored before it is burned. Methane emission associated with the combustion of the manure are not considered here. Combustion-related emissions are estimated in the *Traditional Biomass Fuels* Section of the *Energy* chapter.		5-10%	5-10%	5-10%	e

a Cool climates have an average temperature below 15°C; temperate climates have an average temperature from 15°C to 25°C inclusive; warm climates have an average temperature above 25°C.
b Hashimoto and Steed (1993).
c Safley et al., (1992) and Safley and Westerman (1992).
d Yancun et al. (1985), Stuckey (1984) and Lichtman (1983).
e Safley et al. (1992).

The default emission factors presented in Tier 1 are based on manure management system usage data collected by Safley et al. (1992). Appendix B presents these data by region for cattle, buffalo and swine. Although the data in Appendix B can be used as defaults, country-specific data, e.g., obtained through a survey, would improve the basis for implementing the Tier 2 method. The resulting estimates must show the portion of manure from each animal type managed within each management system, by climate region.

MANURE MANAGEMENT
TIER 2: STEP 2 – EMISSION FACTORS

Emission factors are estimated for each animal type based using the data collected in Step 1 and the methane conversion factors (MCFs) for each manure management system. The MCF defines the portion of the methane producing potential (B_o) that is achieved. The MCF varies by manure management system and climate and can range between 0 and 100 per cent. Table 4-8 presents the latest available MCF estimates for the major manure management systems that have been developed.

To calculate the emission factor for each animal type, a weighted average methane conversion factor (MCF) is calculated using the estimates of the manure managed by waste system within each climate region. The average MCF is then multiplied by the VS excretion rate and the B_o for the animal type. In equation form, the estimate is as follows:

EQUATION 16

$$EF_i = VS_i \text{ x } 365 \text{ days/yr x } B_{oi} \text{ x } 0.67 \text{ kg/m}^3 \text{ x } \sum_{jK} MCF_{jK} \text{ x } MS\%_{ijK}$$

where:

EF_i = annual emission factor (kg) for animal type *i* (e.g., dairy cows);

VS_i = daily VS excreted (kg) for animal type *i*;

B_{oi} = maximum methane producing capacity (m^3/kg of VS) for manure produced by animal type *i*;

MCF_{jk} = methane conversion factors for each manure management system *j* by climate region *k*; and

$MS\%_{ijk}$ = fraction of animal type *i*'s manure handled using manure system *j* in climate region *k*.

MANURE MANAGEMENT
TIER 2: STEP 3 – TOTAL EMISSIONS

To estimate total emissions the selected emission factors are multiplied by the associated animal population and summed. As described above under Tier 1, the emissions estimates should be reported in gigagrams (Gg).

4.2.6 Beyond Tier 2 for Methane

The default values used in the Tier 1 and 2 methods were derived from available livestock and manure management data and are generally representative of regional conditions. Because livestock and manure management conditions can vary significantly across and within countries, the default values may not reflect adequately the conditions in a given country. Additionally, the variability of conditions has not been well characterised to date.

The emissions estimates can be improved by going beyond the Tier 2 default data and collecting key country- or region-specific data. Data elements that would benefit from data collection initiatives (such as targeted surveys of major livestock types) include the following:

- *Cattle weight*
  In many regions the weights of cattle are not well quantified.

- *Feed intake*
  Field data on feed intake would be valuable for validating the feed intake estimates made under Tier 2 for cattle.
- *Manure production*
  Field data on manure production by livestock would be valuable for validating the manure production estimates made under Tier 2.
- *Manure management*
  Field data on manure management system usage would improve the basis for making the estimates. Considerations of seasonal management practices could be incorporated into the data.

In addition to these data collection initiatives, measurement programmes can be used to improve the basis for making the estimates. In particular, measurements of emissions from manure management systems under field conditions is needed. Techniques for making these measurements are described in IAEA (1992). Additionally, measurements of the maximum methane producing ability of manure (B_o) from livestock in tropical regions is needed.

Additionally, new techniques are being deployed to measure emissions from cattle under field conditions (Johnson et al., 1993). Using these techniques, coefficients used in Tier 2 can be verified (such as the methane conversion rate) and the emissions estimates can be validated. Targeted assessments of tropical cattle populations would be most valuable.

4.2.7 Inventory Method for Nitrous Oxide - Overview

The method for estimating N_2O emissions from manure management is described in detail in Section 4.5.3 of this Reference Manual, where emissions from several animal waste management systems are considered. All emissions of N_2O taking place before the manure is added to soils are to be reported under ''Manure Management''. These include emissions from anaerobic lagoons, liquid systems, solid storage and drylot, and ''other systems''. Emissions resulting from manure used for fuel are included in the Energy Chapter. All manure-induced soil emissions are considered soil emissions here.

Appendix A

Data Underlying Methane Default Emission Factors For Enteric Fermentation

This appendix presents the data used to develop the default emission factors for methane emissions from enteric fermentation. The detailed information presented for cattle and buffalo was developed in Gibbs and Johnson (1993). The Tier 2 method was implemented with these data to estimate the default emission factors for cattle and buffalo. Also presented are the summary data from Crutzen et al. (1986) that were used to estimate the emission factors for the other species.

TABLE A-1
DATA FOR ESTIMATING ENTERIC FERMENTATION EMISSION FACTORS FOR DAIRY CATTLE

Regions	Weight kg	Weight Gain kg/day	Feeding Situation	Milk kg/day	Work hrs/day	% Pregnant	Digestibility of Feed %	CH_4 Conversion %
North America[a]	600	0	Stall Fed	18.4	0	90%	65%	6%
Western Europe	550	0	Stall Fed	11.5	0	90%	60%	6%
Eastern Europe[b]	550	0	Stall Fed	7.0	0	80%	60%	6%
Oceania[c]	500	0	Stall Fed	4.7	0	80%	60%	6%
Latin America[d]	400	0	Pasture/Range	2.2	0	80%	60%	6%
Asia[e]	350	0	Stall Fed	4.5	0	80%	60%	6%
Africa & Middle East	275	0	Stall Fed	1.3	0	67%	60%	6%
Indian Subcontinent[f]	275	0	Stall Fed	2.5	0	50%	55%	6%

a Based on estimates for the United States

b Based on estimates for the former USSR

c Based on estimates for Australia.

d Based on estimates for Brazil.

e Based on estimates for China.

f Based on estimates for India.

Source: Gibbs and Johnson (1993).

TABLE A-2
DATA FOR ESTIMATING ENTERIC FERMENTATION EMISSION FACTORS FOR NON-DAIRY CATTLE

Type	Weight kg	Weight Gain kg/day	Feeding Situation	Milk kg/day	Work hrs/day	Pregnant %	Digestibility of Feed %	CH_4 Conversion %	Day Weighted Population Mix %	Emission Factors kg/head/yr
North America[a]										
Mature Females	500	0.0	Pasture/Range	3.3	0.0	80%	60%	6.0%	36%	69
Mature Males	800	0.0	Pasture/Range	0.0	0.0	0%	60%	6.0%	2%	75
Calves on milk	100	0.9	Pasture/Range	0.0	0.0	0%	NA	0.0%	16%	0
Calves on forage	185	0.9	Pasture/Range	0.0	0.0	0%	65%	6.0%	8%	42
Growing heifers/steers	265	0.7	Pasture/Range	0.0	0.0	0%	65%	6.0%	17%	47
Replacement/growing	375	0.4	Pasture/Range	0.0	0.0	0%	60%	6.0%	11%	56
Feedlot cattle	415	1.3	Stall Fed	0.0	0.0	0%	75%	3.5%	11%	37
Western Europe										
Mature Males	600	0.0	Pasture/Range	0.0	0.0	0%	60%	6.0%	22%	60
Replacement/growing	400	0.4	Pasture/Range	0.0	0.0	0%	60%	6.0%	54%	84
Calves on milk	230	0.3	Pasture/Range	0.0	0.0	0%	65%	0.0%	15%	0
Calves on forage	230	0.3	Pasture/Range	0.0	0.0	0%	65%	6.0%	8%	33
Eastern Europe[b]										
Mature Females	500	0.0	Pasture/Range	3.3	0.0	67%	60%	6.5%	30%	74
Mature Males	600	0.0	Pasture/Range	0.0	0.0	0%	60%	6.5%	22%	65
Young	230	0.4	Pasture/Range	0.0	0.0	0%	60%	6.0%	48%	40
Oceania[c]										
Mature Females	400	0.0	Pasture/Range	2.4	0.0	67%	55%	6.0%	51%	63
Mature Males	450	0.0	Pasture/Range	0.0	0.0	0%	55%	6.0%	11%	55
Young	200	0.3	Pasture/Range	0.0	0.0	0%	55%	6.0%	38%	39

a Based on estimates for the United States.
b Based on estimates for the former USSR.
c Based on estimates for Australia.

TABLE A-2 (CONTINUED)
DATA FOR ESTIMATING ENTERIC FERMENTATION EMISSION FACTORS FOR NON-DAIRY CATTLE

Type	Weight kg	Weight Gain kg/day	Feeding Situation	Milk kg/day	Work hrs/day	Pregnant %	Digestibility of Feed %	CH_4 Conversion %	Day Weighted Population Mix %	Emission Factors kg/head/yr
Latin America[d]										
Mature Females	400	0.0	Large Areas	1.1	0.0	67%	60%	6.0%	37%	58
Mature Males	450	0.0	Large Areas	0.0	0.0	0%	60%	6.0%	6%	57
Young	230	0.3	Large Areas	0.0	0.0	0%	60%	6.0%	58%	42
Asia[e]										
Mature Females-Farming	325	0.0	Stall Fed	1.1	0.55	33%	55%	6.5%	27%	48
Mature Females-Grazing	300	0.0	Pasture/Range	1.1	0.00	50%	60%	6.0%	9%	41
Mature Males-Farming	450	0.0	Stall Fed	0.0	1.37	0%	55%	6.5%	24%	58
Mature Males-Grazing	400	0.0	Pasture/Range	0.0	0.00	0%	60%	6.0%	8%	44
Young	200	0.2	Pasture/Range	0.0	0.00	0%	60%	6.0%	32%	31
Africa										
Mature Females	200	0.0	Stall Fed	0.3	0.55	33%	55%	6.5%	13%	31
Draft Bullocks	275	0.0	Stall Fed	0.0	1.37	0%	55%	6.5%	13%	40
Mature Females-Grazing	200	0.0	Large Areas	0.3	0.00	33%	55%	7.5%	6%	46
Bulls - Grazing	275	0.0	Large Areas	0.0	0.00	0%	55%	7.5%	25%	55
Young	75	0.1	Pasture/Range	0.0	0.00	0%	60%	6.0%	44%	14
Indian Subcontinent[f]										
Mature Females	125	0.0	Stall Fed	0.6	0.00	33%	50%	7.5%	40%	31
Mature Males	200	0.0	Stall Fed	0.0	2.74	0%	50%	7.5%	10%	41
Young	80	0.1	Stall Fed	0.0	0.00	0%	50%	6.0%	50%	17

d Based on estimates for the Brazil.
e Based on estimates for the China.
f Based on estimates for India.
Source: Gibbs and Johnson (1993)

TABLE A-3
DATA FOR ESTIMATING ENTERIC FERMENTATION EMISSION FACTORS FOR BUFFALO

Type	Weight kg	Weight Gain kg/day	Feeding Situation	Milk kg/day	Work hrs/day	Pregnant %	Digestibility of Feed %	CH_4 Conversion %	Day Weighed Population Mix %	Emissions Factors kg/head/yr
Indian Subcontinent[a]										
Adult Males	350 - 550	0.00	Stall Fed	0.00	1.37	0%	55%	7.5	14%	55 - 77
Adult Females	250 - 450	0.00	Stall Fed	2.70	0.55	33%	55%	7.5	40%	57 - 80
Young	100 - 300	0.15	Stall Fed	0.00	0.00	0%	55%	7.5	46%	23 - 50
Other Countries[b]										
Adult Males	350 - 550	0.00	Stall Fed	0.00	1.37	0%	55%	7.5	45%	55 - 77
Adult Females	250 - 450	0.00	Stall Fed	0.00	0.55	25%	55%	7.5	45%	45 - 67
Young	100 - 300	0.15	Stall Fed	0.15	0.00	0%	55%	7.5	10%	23 - 50

a Based on estimates for India.

b Based on estimates for China.

Source: Gibbs and Johnson (1993).

TABLE A-4
DATA FOR ESTIMATING ENTERIC FERMENTATION EMISSION FACTORS FOR OTHER ANIMALS

Animal Type		Feed Intake (MJ/head/day)	Methane Conversion Factor (%)
Sheep	Developed Countries	20	6%
	Developing Countries	13	6%
Goats	Developed Countries	14	5%
	Developing Countries	14	5%
Camels	Developed Countries	100	7%
	Developing Countries	100	7%
Horses	Developed Countries	110	2.5%
	Developing Countries	110	2.5%
Mules/Asses	Developed Countries	60	2.5%
	Developing Countries	60	2.5%
Swine	Developed Countries	38	0.6%
	Developing Countries	13	1.3%
Poultry	Developed Countries	Not Estimated	
	Developing Countries		

Sources: Feed intake and methane conversion for all animals from Crutzen et al (1986). Methane conversion for camels modified as in Gibbs and Johnson (1993).

Appendix B

Data Underlying Methane Default Emission Factors for Manure Management

This appendix presents the data used to develop the default emission factors for methane emissions from manure management. The detailed information presented for cattle and buffalo was developed in Gibbs and Johnson (1993). The swine feed intake data are from Crutzen et al. (1986). The manure management system usage data and B_o estimates are from Safley et al. (1992). The methane conversion factor (MCF) data are from Woodbury and Hashimoto (1993). The Tier 2 method was implemented with these data to estimate the default emission factors for cattle, buffalo, and swine. Also presented are the summary feed intake data from Crutzen et al. (1986) and the manure-related data from Safley et al. (1992) and Woodbury and Hashimoto (1993) that were used to estimate the emission factors for the other species.

TABLE B-1
FEED INTAKE AND MANURE PRODUCTION FOR CATTLE

Region	Livestock Category	Sub-Population	Mass (kg)	Feed Digest (%)	Energy Intake (MJ/day)	Feed Intake (kg/day)	Category Population %	Manure (kg/h/d dm)	VS (kg/h/d)	B_o (m^3 CH_4/kg VS)
North America	Dairy Cattle	Average	600	65%	299.5	16.2	100%	5.68	5.23	0.24
	Non-dairy Cattle	Mature Females	500	60%	174.0	9.4	36%	3.77	3.47	0.17
		Mature Males	800	60%	189.3	10.3	2%	4.10	3.78	0.17
		Young on milk	100	NA	NA	NA	15%	negligible	negligible	0.17
		Young	185	65%	107.2	5.8	8%	2.03	1.87	0.17
		Young	265	65%	120.1	6.5	17%	2.28	2.10	0.17
		Young	375	60%	143.2	7.8	11%	3.10	2.86	0.17
		Feedlot	415	75%	161.8	8.8	11%	2.19	2.02	0.17
		Avg. Non-dairy Cattle	357	53%	128.0	6.9	100%	2.55	2.35	0.17
Western Europe	Dairy Cattle	Average	550	60%	254.7	13.8	100%	5.52	5.08	0.24
	Non-dairy Cattle	Mature Males	600	60%	152.5	8.3	22%	3.31	3.04	0.17
		Young Replacements	400	60%	149.8	8.1	55%	3.25	2.99	0.17
		Young Calves	230	65%	83.7	4.5	23%	1.59	1.46	0.17
		Avg. Non-dairy Cattle	405	61%	135.1	7.3	100%	2.88	2.65	0.17

TABLE B-1 (CONTINUED)
FEED INTAKE AND MANURE PRODUCTION FOR CATTLE

Region	Livestock Category	Sub-Population	Mass (kg)	Feed Digestibility (%)	Energy Intake (MJ/day)	Feed Intake (kg/day)	Category Population %	Manure (kg/h/d dm)	VS (kg/h/d)	B_o (m^3 CH4/kg VS)
Eastern Europe	Dairy Cattle	Average	550	60%	207.2	11.2	100%	4.49	4.13	0.24
	Non-dairy Cattle	Mature Females	500	60%	172.9	9.4	30%	3.75	3.45	0.17
		Mature Males	600	60%	152.5	8.3	21%	3.31	3.04	0.17
		Young	230	60%	102.2	5.5	49%	2.22	2.04	0.17
		Avg. Non-dairy Cattle	391	60%	134.4	7.3	100%	2.91	2.68	0.17
Oceania	Dairy Cattle	Average	500	60%	174.1	9.4	100%	3.77	3.47	0.24
	Non-dairy Cattle	Mature Females	400	55%	160.5	8.7	52%	3.91	3.60	0.17
		Mature Males	450	55%	138.8	7.5	10%	3.38	3.11	0.17
		Young	200	55%	98.6	5.3	38%	2.41	2.21	0.17
		Avg. Non-dairy Cattle	330	55%	134.9	7.3	100%	3.29	3.03	0.17
Latin America	Dairy Cattle	Average	400	60%	145.9	7.9	100%	3.16	2.91	0.13
	Non-dairy Cattle	Mature Females	400	60%	148.0	8.0	37%	3.21	2.95	0.10
		Mature Males	450	60%	144.0	7.8	5%	3.12	2.87	0.10
		Young	230	60%	107.5	5.8	58%	2.33	2.14	0.10
		Avg. Non-dairy Cattle	305	60%	124.4	6.7	100%	2.70	2.48	0.10

TABLE B-1 (CONTINUED)
FEED INTAKE AND MANURE PRODUCTION FOR CATTLE

Region	Livestock Category	Sub-Population	Mass (kg)	Feed Digestibility (%)	Energy Intake (MJ/day)	Feed Intake (kg/day)	Category Population %	Manure (kg/h/d dm)	VS (kg/h/d)	B_o (m^3 CH4/kg VS)
Africa & Middle East	Dairy Cattle	Average	275	60%	92.8	5.0	100%	2.01	1.85	0.13
	Non-dairy Cattle	Mature Females	200	55%	73.2	4.0	13%	1.79	1.64	0.10
		Mature Males Draft	275	55%	93.2	5.1	13%	2.27	2.09	0.10
		Mature Females Grazing	200	55%	93.6	5.1	5%	2.28	2.10	0.10
		Mature Males Grazing	275	55%	112.3	6.1	25%	2.74	2.52	0.10
		Young	75	60%	36.2	2.0	44%	0.78	0.72	0.10
		Avg. Non-dairy Cattle	173	57%	70.6	3.8	100%	1.68	1.54	0.10
Asia	Dairy Cattle	Average	350	60%	141.6	7.7	100%	3.07	2.82	0.13
	Non-dairy Cattle	Mature Females Farming	325	55%	113.2	6.1	27%	2.76	2.54	0.10
		Mature Females Grazing	300	60%	105.0	5.7	9%	2.28	2.09	0.10
		Mature Males Farming	450	55%	134.9	7.3	24%	3.29	3.03	0.10
		Mature Males Grazing	400	60%	112.5	6.1	8%	2.44	2.24	0.10
		Young	200	60%	79.3	4.3	32%	1.72	1.58	0.10
		Avg. Non-dairy Cattle	319	57%	106.8	5.8	100%	2.49	2.29	0.10
Indian Subcontinent	Dairy Cattle	Average Dairy Cow	275	55%	117.7	6.4	100%	2.87	2.64	0.13
	Non-dairy Cattle	Mature Females	125	50%	63.8	3.5	40%	1.73	1.59	0.10
		Mature Males	200	50%	83.7	4.5	10%	2.27	2.09	0.10
		Young	80	50%	43.2	2.3	50%	1.17	1.08	0.10
		Avg. Non-dairy Cattle	110	50%	55.5	3.0	100%	1.50	1.38	0.10

Ash content estimated at 8%. Cattle characteristics from Gibbs and Johnson (1993).

TABLE B-2
FEED INTAKE AND MANURE PRODUCTION FOR SWINE AND BUFFALO

Region	Livestock Category	Sub-Population	Mass (kg)	Feed Digestibility (%)	Energy Intake (MJ/day)	Feed Intake (kg/day)	Category Population %	Manure (kg/h/d dm)	VS (kg/h/d)	B_o (m^3CH_4/kg VS)
Developing Countries	Swine	Average	28	50%	13.0	0.7	100%	0.35	0.34	0.29
Developed Countries	Swine	Average	82	75%	38.0	2.1	100%	0.51	0.50	0.45
Indian Subcontinent[a]	Buffalo	Adult Males	450	55%	134.5	7.3	14%	3.28	3.02	0.10
		Adult Females	350	55%	139.0	7.5	40%	3.39	3.12	0.10
		Young	200	55%	74.0	4.0	46%	1.80	1.66	0.10
		Average	295	55%	108.0	5.9	100%	2.65	2.43	0.10
Rest of World[b]	Buffalo	Adult Males	450	55%	134.5	7.3	45%	3.28	3.02	0.10
		Adult Females	350	55%	114.0	6.2	45%	2.78	2.56	0.10
		Young	200	55%	74.0	4.1	10%	1.80	1.66	0.10
		Average	380	55%	119.0	6.5	100%	2.91	2.68	0.10

[a] Estimates based on data for India.

[b] Estimates based on data for China.

Ash content taken as 8% for buffalo and 2% and 4% for swine in developed and developing countries, respectively.

Sources: Buffalo characteristics from Gibbs and Johnson (1993). Swine feed intake estimates from Crutzen et al. (1986).

TABLE B-3
MANURE MANAGEMENT EMISSION FACTOR DERIVATION FOR DAIRY CATTLE

Climate	Manure Management System MCFs								
	Lagoon	Liquid/ Slurry	Solid Storage	Drylot	Pasture/ Range	Daily Spread	Digester	Burned for Fuel	Other
Cool	90%	10%	1%	1%	1%	0.1%	10%	10%	1%
Temperate	90%	35%	1.5%	1.5%	1.5%	0.5%	10%	10%	1%
Warm	90%	65%	2%	5%	2%	1%	10%	10%	1%

Region	Dairy Cattle Characteristics			Manure Management System Usage (MS%)									EMISSION FACTORS kg CH_4 / hd / yr		
	Mass[a] (kg)	B_o (m^3CH_4/kg Vs)	VS[b] (kg/hd/day)	Lagoon	Liquid/ Slurry	Solid Storage	Drylot	Pasture/ Range	Daily Spread	Digester	Burned for Fuel	Other	Cool	Temperate	Warm
North America	600	0.24	5.2	10%	23%	18%	5%	0%	37%	0%	0%	7%	36	54	76
Western Europe	550	0.24	5.1	0%	40%	18%	0%	19%	20%	0%	2%	1%	14	44	81
Eastern Europe	550	0.24	4.1	0%	18%	68%	0%	13%	1%	0%	0%	0%	6	19	33
Oceania	500	0.24	3.5	16%	1%	0%	0%	76%	8%	0%	0%	0%	31	32	33
Latin America	400	0.13	2.9	0%	1%	1%	0%	36%	62%	0%	0%	0%	0	1	2
Africa	275	0.13	1.9	0%	0%	1%	0%	83%	5%	0%	6%	4%	1	1	1
Middle East	275	0.13	1.9	0%	1%	2%	0%	80%	2%	0%	17%	0%	1	2	2
Asia	350	0.13	2.8	4%	38%	0%	0%	20%	29%	2%	7%	0%	7	16	27
Indian Subcontinent	275	0.13	2.6	0%	1%	0%	0%	27%	19%	1%	51%	0%	5	5	6

[a] Average dairy cow mass for each region.

[b] Average VS production for per head per day for the average dairy cow.

Emission Factors (EF) for a climate region, k, are calculated as follows:

$$EF_k = B_o \times VS \times 365 \times \sum_{j=1}^{\text{all manure systems}} MS\%_j \times MCF_{jk}$$

TABLE B-4
MANURE MANAGEMENT EMISSION FACTOR DERIVATION FOR NON-DAIRY CATTLE

Climate	Manure Management System MCFs								
	Lagoon	Liquid / Slurry	Solid Storage	Drylot	Pasture / Range	Daily Spread	Digester	Burned for Fuel	Other
Cool	90%	10%	1%	1%	1%	0.1%	10%	10%	1%
Temperate	90%	35%	1.5%	1.5%	1.5%	0.5%	10%	10%	1%
Warm	90%	65%	2%	5%	2%	1%	10%	10%	1%

Region	Non-dairy Cattle Characteristics			Manure Management System Usage (MS%)									EMISSION FACTORS kg CH_4 / hd / yr		
	Mass[a] (kg)	B_o (m^3CH_4 /kg VS)	VS[b] (kg/hd/day)										Cool	Temperate	Warm
North America	357	0.17	2.4	0%	1%	0%	14%	84%	0%	0%	0%	1%	1	2	3
Western Europe	405	0.17	2.7	0%	50%	0%	2%	38%	0%	0%	2%	8%	6	20	38
Eastern Europe	391	0.17	2.7	0%	28%	0%	0%	26%	0%	0%	0%	46%	4	13	23
Oceania	330	0.17	3.0	3%	0%	0%	6%	91%	0%	0%	0%	0%	5	6	7
Latin America	305	0.10	2.5	0%	0%	0%	0%	99%	0%	0%	0%	1%	1	1	1
Africa	173	0.10	1.5	0%	0%	0%	1%	95%	1%	0%	3%	0%	0	1	1
Middle East	173	0.10	1.5	0%	0%	0%	1%	79%	2%	0%	17%	2%	1	1	1
Asia	319	0.10	2.3	0%	0%	0%	46%	50%	2%	0%	2%	0%	1	1	2
Indian Subcontinent	110	0.10	1.4	0%	1%	0%	4%	22%	20%	1%	53%	0%	2	2	2

a Average non-dairy cow mass for each region.

b Average VS production for per head per day for the average non-dairy animal.

Emission Factors (EF) for a climate region, k, are calculated as follows:

$$EF_k = B_o \times VS \times 365 \times \sum_{j=1}^{\text{all manure systems}} MS\%_j \times MCF_{jk}$$

TABLE B-5
MANURE MANAGEMENT EMISSION FACTOR DERIVATION FOR BUFFALO

Climate	Manure Management System MCFs						
	Liquid/ Slurry	Drylot	Pasture/ Range	Daily Spread	Digester	Burned for Fuel	Other
Cool	10.0%	1.0%	1.0%	0.1%	10.0%	10.0%	1.0%
Temperate	35.0%	1.5%	1.5%	0.5%	10.0%	10.0%	1.0%
Warm	65.0%	5.0%	2.0%	1.0%	10.0%	10.0%	1.0%

Region	Buffalo Characteristics			Manure Management System Usage (MS%)							EMISSION FACTORS kg CH_4 / hd / yr		
	Mass[a] kg	B_o m^3CH_4 /kg VS	VS[b] kg/hd/day								Cool	Temperate	Warm
North America	(not applicable)												
Western Europe	380	0.10	3.9	20%	79%	0%	0%	0%	0%	0%	3	8	17
Eastern Europe	380	0.10	3.9	24%	0%	29%	0%	0%	0%	47%	3	9	16
Oceania	(not applicable)												
Latin America	380	0.10	3.9	0%	0%	99%	0%	0%	0%	1%	1	1	2
Africa	(not applicable)												
Middle East	380	0.10	3.9	0%	0%	20%	19%	0%	42%	19%	4	5	5
Asia	380	0.10	3.9	0%	41%	50%	4%	0%	5%	0%	1	2	3
Indian Subcontinent	295	0.10	3.1	0%	4%	19%	21%	1%	55%	0%	4	5	5

[a] Average buffalo mass for each region.

[b] Average VS production for per head per day for the average buffalo.

Emission Factors (EF) for a climate region, k, are calculated as follows:

$$EF_k = B_o \times VS \times 365 \times \sum_{j=1}^{\text{all manure systems}} MS\%_j \times MCF_{jk}$$

TABLE B-6
MANURE MANAGEMENT EMISSION FACTOR DERIVATION FOR SWINE

Climate	**Manure Management Systems MCFs**								
	Lagoon	Liquid/ Slurry	Solid Storage	Drylot	Pit <1 month	Pit >1 month	Daily Spread	Digester	Other
Cool	90%	10%	1%	1%	5%	10%	0.1%	10%	1%
Temperate	90%	35%	1.5%	1.5%	18%	35%	0.5%	10%	1%
Warm	90%	65%	2%	5%	33%	65%	1%	10%	1%

Region	Swine Characteristics			Manure Management System Usage (MS%)									**EMISSION FACTORS** kg CH_4 / hd / yr		
	Mass[a] kg	B_o m^3CH_4 /kg VS	VS[b] kg/hd/day										Cool	Temperate	Warm
North America	82	0.45	0.5	24%	1%	2%	16%	10%	26%	0%	0%	19%	10	14	18
Western Europe	82	0.45	0.5	0%	0%	21%	2%	3%	73%	0%	0%	1%	3	10	19
Eastern Europe	82	0.45	0.5	8%	0%	39%	14%	19%	19%	0%	0%	1%	4	7	11
Oceania	82	0.45	0.5	54%	0%	3%	15%	0%	0%	0%	0%	28%	20	20	20
Latin America	28	0.29	0.3	0%	8%	10%	41%	0%	0%	2%	0%	40%	0	1	2
Africa	28	0.29	0.3	0%	6%	6%	87%	1%	0%	0%	0%	0%	0	1	2
Middle East	28	0.29	0.3	0%	14%	0%	69%	0%	17%	0%	0%	0%	1	3	6
Asia	28	0.29	0.3	0%	40%	0%	54%	0%	0%	0%	7%	0%	1	4	7
Indian Subcontinent	28	0.29	0.3	9%	22%	16%	30%	3%	0%	9%	8%	3%	3	4	6

a Average swine mass for each region.

b Average VS production for per head per day for the average swine.

Emission Factors (EF) for a climate region, k, are calculated as follows:

$$EF_k = B_o \times VS \times 365 \times \sum_{j=1}^{\text{all manure systems}} MS\%_j \times MCF_{jk}$$

TABLE B-7
MANURE MANAGEMENT EMISSION FACTOR DERIVATION FOR OTHER LIVESTOCK

Animal		Animal Characteristics						Manure Management System MCFs[a]			Emission Factors[a] kg CH_4 / hd / yr		
		Mass (kg)	Digest (%)	Intake/d (kg Feed)	% Ash (Dry Basis)	VS/day (kg VS)	B_o (m^3/kg VS)	MCF Cool	MCF Temperate	MCF Warm	Cool	Temperate	Warm
Sheep	Developed Countries	43	60%	1.08	8.0	0.40	0.19	1%	1.5%	2%	0.19	0.28	0.37
	Developing Countries	28	50%	0.70	8.0	0.32	0.13	1%	1.5%	2%	0.10	0.16	0.21
Goats	Developed Countries	30	60%	0.76	8.0	0.28	0.17	1%	1.5%	2%	0.12	0.18	0.23
	Developing Countries	30	50%	0.76	8.0	0.35	0.13	1%	1.5%	2%	0.11	0.17	0.22
Camels	Developed Countries	217	50%	5.42	8.0	2.49	0.26	1%	1.5%	2%	1.59	2.38	3.17
	Developing Countries	217	50%	5.42	8.0	2.49	0.21	1%	1.5%	2%	1.28	1.92	2.56
Horses	Developed Countries	238	70%	5.96	4.0	1.72	0.33	1%	1.5%	2%	1.39	2.08	2.77
	Developing Countries	238	70%	5.96	4.0	1.72	0.26	1%	1.5%	2%	1.09	1.64	2.18
Mule/Asses	Developed Countries	130	70%	3.25	4.0	0.94	0.33	1%	1.5%	2%	0.76	1.14	1.51
	Developing Countries	130	70%	3.25	4.0	0.94	0.26	1%	1.5%	2%	0.60	0.90	1.19
Poultry[b]	Developed Countries	1.1	NR[c]	NR	NR	0.10	0.32	1%	1.5%	2%	0.078	0.117	0.157
	Developing Countries	NR	NR	NR	NR	0.02	0.24	1%	1.5%	2%	0.012	0.018	0.023

a The range of estimates reflects cool to warm climates. Cool climates have an average annual temperature below 15°C; temperate climates have an average annual temperature between 15°C and 25°C; and warm climates have an average annual temperature above 25°C.

b Poultry include chickens, ducks, and turkeys.

c Not reported.

Sources: Except for poultry, emission factors were developed from: feed intake values and feed digestibilities used to develop the enteric fermentation emission factors (see Appendix A); MCF, and B_o values reported in Woodbury and Hashimoto (1993). All manure is assumed to be managed in dry systems, which is consistent with the manure management system usage reported in Woodbury and Hashimoto (1993). Emission factors for poultry are based on Safley et al. (1992) and Woodbury and Hashimoto (1993).

Appendix C

Derivation of Tier 2 Enteric Fermentation Equations For Methane

This appendix summarises the derivation of the relationship between net energy (NE) and digestible energy (DE) that is used to estimate total feed-intake requirements for cattle. This derivation is drawn from Gibbs and Johnson (1993).

As described in the main text, the relationship among the energy values of feed consumed by cattle can be summarised as follows:

Digestible Energy = Gross Energy - Faecal Losses

Metabolisable Energy = Digestible Energy - Urinary and Combustible Gas Losses

Net Energy = Metabolisable Energy - Heat Increment

Net Energy = Gross Energy - Faecal Losses - Urinary and Combustible Gas Losses - Heat Increment

NRC (1984) presents the following quantitative relationships among these energy values:

$$ME = 0.82 \times DE \qquad (C.1)$$

$$NE_m = (1.37 \times ME) - (0.138 \times ME^2) + (0.0105 \times ME^3) - 1.12 \qquad (C.2)$$

$$NE_g = (1.42 \times ME) - (0.174 \times ME^2) + (0.0122 \times ME^3) - 1.65 \qquad (C.3)$$

where:

DE = digestible energy in Mcal/kg (dry matter basis);

ME = metabolisable energy in Mcal/kg (dry matter basis);

NE_m = net energy for maintenance in Mcal/kg (dry matter basis); and

NE_g = net energy for growth in Mcal/kg (dry matter basis).

Using these relationships, the ratio of NE_m and NE_g to ME or DE can be derived as follows:

$$NE/DE = 1.123 - (4.092 \times 10^{-3} \times DE\%) + (1.126 \times 10^{-5} \times (DE\%)^2) - 25.4/DE\% \qquad (C.4)$$

$$NE_g/DE = 1.164 - (5.160 \times 10^{-3} \times DE\%) + (1.308 \times 10^{-5} \times (DE\%)^2) - 37.4/DE\% \qquad (C.5)$$

where:

NE/DE = the ratio of net energy consumed for maintenance, lactation, work and pregnancy to digestible energy consumed;

NE_g/DE = the ratio of net energy consumed for growth to digestible energy consumed; and

$DE\%$ = digestible energy as percentage of gross energy, expressed in per cent (e.g., 65%).

Graph C-1 shows the relationships in graphical form. As shown in the graph, the ratio of NE to DE is non-linear, with an increasing slope with decreasing DE. These relationships imply that at lower values of DE, cattle are able to recover a decreasing portion of the energy to use for maintenance or growth.

For the purpose of estimating methane emissions from cattle, applying these relationships to cattle consuming relatively low-quality feeds (such as cattle in many tropical countries) may be inappropriate because the relationships were developed based on analyses of the higher-quality feeds typically found in the United States temperate agriculture system. Consequently, the experimental basis for extrapolating the non-linear relationships to low levels of DE is not very strong.

In examining other energy systems, it is seen that they also indicate that the rate of net energy retention declines at lower values of digestible energy. Unlike the NRC system, however, many imply a *linear* relationship between NE and DE. The UK energy system (ARC, 1980), which is typical of the energy systems used in Europe, has a slope for the linear NE_m:DE relationship that is similar to the slope of the non-linear NRC relationship in the range of 65-70 per cent digestibility. In the same way, the slope of the UK NE_g:DE relationship is similar to the slope of the non-linear NRC relationship in the range of 60-65 per cent digestibility.

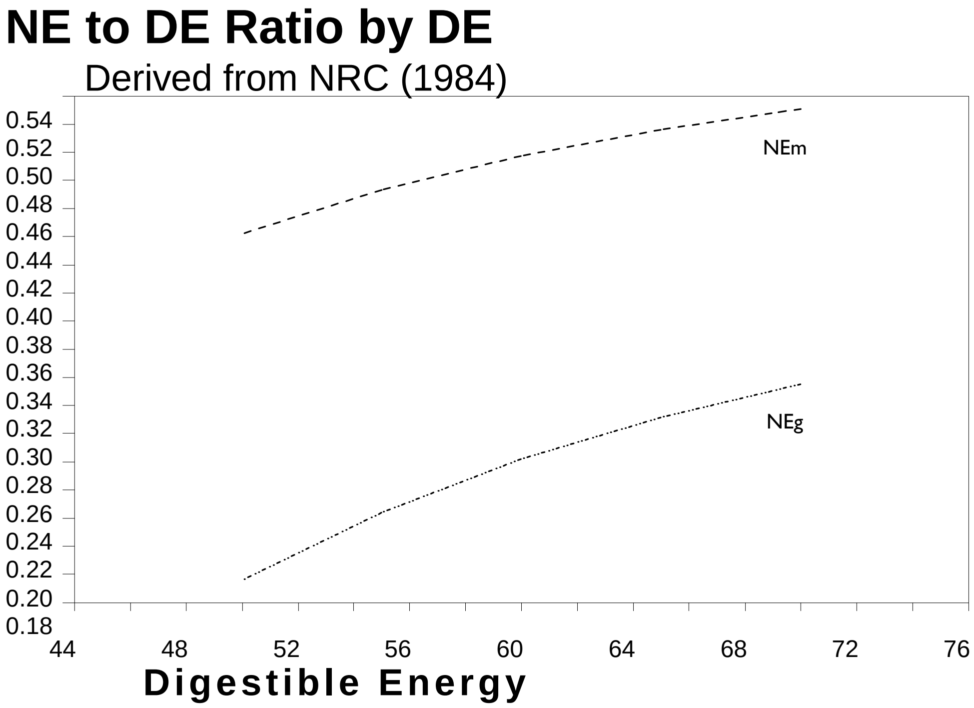
To avoid possible biases in estimating feed-intake requirements in this study, the relationships were extrapolated linearly for DE values below 65 per cent using the average slopes of the NRC relationships between 60 and 70 per cent DE. The derived equations are as follows:

$$NE/DE = 0.298 + 0.00335 \times DE\% \quad (C.6)$$

$$NE_g/DE = -0.036 + 0.00535 \times DE\% \quad (C.7)$$

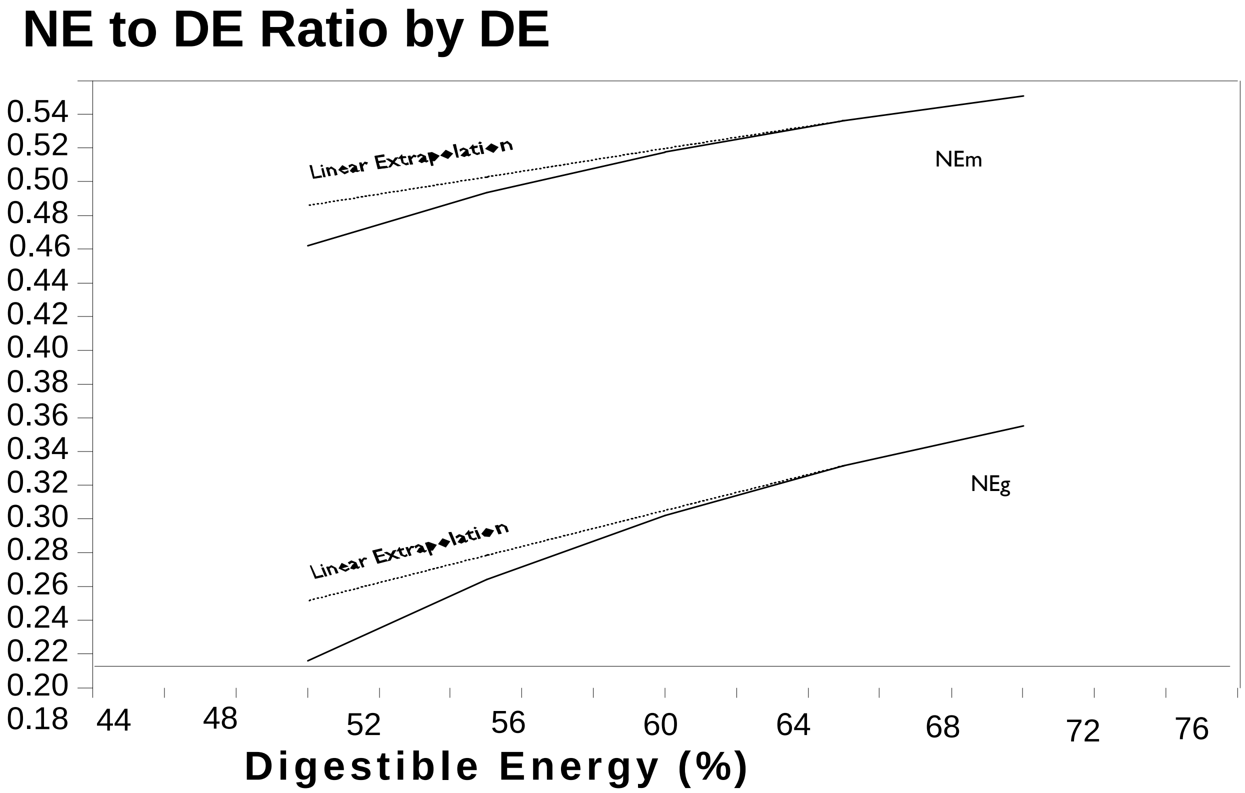
Graph C-2 shows the extrapolated linear relationships along with the non-linear estimates. As expected, the linear extrapolations fall *above* the original non-linear estimates.

The implication of making this adjustment to the NRC (1984) relationship for the global emissions estimate is relatively minor. Gibbs and Johnson (1993) report that using the non-linear relationship to estimate global emissions from cattle increases the 1990 emissions estimate by 1000 Gg, from 58,100 Gg to 59,100 Gg. Considering the wide range of factors that contribute to uncertainty in the estimates, including characterisation of animal populations, this adjustment has a minor influence on the estimates.



Graph C-1: NRC NE:DE Relationship

Graph C-2: Linear Extrapolation Of The NRC NE:DE Relationship



4.3 Methane Emissions from Rice Cultivation: Flooded Rice Fields

4.3.1 Overview

Anaerobic decomposition of organic material in flooded rice fields produces methane (CH_4), which escapes to the atmosphere primarily by diffusive transport through the rice plants during the growing season. Upland rice fields, which are not flooded and therefore do not produce significant quantities of CH_4, account for approximately 10 per cent of the global rice production and about 15 per cent of the global rice area under cultivation. The remaining area is grown for wetland rice, consisting of irrigated, rainfed, and deepwater rice. The global wetland rice area harvested annually in the early 1980s was about 123.2 million hectares (total harvested area including upland rice is 144 Mha), over 90 per cent of which was in Asia (Neue et al., 1990).[14]

Of the wide variety of sources of atmospheric CH_4, rice paddy fields are considered one of the most important. The Intergovernmental Panel on Climate Change (IPCC, 1996) estimated the global emission rate from paddy fields at 60 Tg/yr, with a range of 20 to 100 Tg/yr. This is about 5-20 per cent of the total emission from all anthropogenic sources. This figure is mainly based on field measurements of CH_4 fluxes from paddy fields in the United States, Spain, Italy, China, India, Australia, Japan and Thailand.

The measurements at various locations of the world show that there are large temporal variations of CH_4 fluxes and that the flux differs markedly with soil type and texture, application of organic matter and mineral fertiliser (Neue and Sass, 1994). The wide variations in CH_4 fluxes also indicate that the flux is critically dependent upon several factors including climate, characteristics of soils and paddy, and agricultural practices, particularly water regime. The parameters that affect methane emissions vary widely both spatially and temporally. Multiple year data sets near the same location and under similar conditions can lead to substantial differences in seasonal methane emission levels, making it difficult to establish a single number as the methane emission level from a field, let alone at a regional or country level. Thus, at the current level of understanding, a reported range in methane emission levels for a country is more realistic than a single number.

Methane production processes

The major pathways of CH_4 production in flooded soils are the reduction of CO_2 with H_2, with fatty acids or alcohols as hydrogen donor, and the transmethylation of acetic acid or methanol by methane-producing bacteria (Takai, 1970; Conrad 1989). In paddy fields, the kinetics of the reduction processes are strongly affected by the composition and texture of soil and its content of inorganic electron acceptors. The period between flooding of the soil and the onset of methanogenesis can apparently be different for the various soils. However, it is unclear if soil type also affects the rates of methanogenesis and CH_4 emission when steady state conditions have been reached (Conrad, 1989).

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[14] The term "harvested area" has a different meaning from "cultivated area" in that the former accounts for double and triple cropping. For example, if a country has 10 million hectares of land under rice cultivation, all of which are double-cropped (i.e., two crops of rice are grown on each hectare each year), then this country has 20 million hectares of rice area harvested annually.

The redox[15] potential is one important factor for production of CH_4 in soils. The Eh, or electron activity, of the soil gradually decreases after flooding. Patrick (1981) demonstrated that the redox potential of a soil must be below approximately -150 mV in order to have CH_4 production. Yamane and Sato (1964) also showed that the evolution of CH_4 from flooded paddy soils did not commence until the Eh fell below -150 mV.

Carbon substrate and nutrient availability are also important factors. Application of rice straw to paddy fields significantly increases the CH_4 emission rate compared with application of compost prepared with rice straw or chemical fertiliser.

Soil temperature is known to be an important factor in affecting the activity of soil micro-organisms. This is to a certain extent related to the soil moisture content because both the heat capacity and the heat conductivity are lower for a dry soil than for a wet soil. Yamane and Sato (1961) have already found that CH_4 formation reached a maximum at 35°C in waterlogged alluvial soils. The rate of methane formation was very small below 20°C.

Because the conversion rate of substrate to CH_4 depends on the temperature, it is generally observed that the momentary local emission of CH_4 from the soil to the atmosphere depends on the temperature. However, the dependence of the seasonally integrated emissions of CH_4 on temperature is much weaker. That emission depends primarily on the total input of organic substrate: although the temperature determines the time it takes to convert the substrate to CH_4, that time is generally short compared to a season. Thus the methodology proposed here will be based on the seasonally integrated CH_4 emission, whose temperature dependence can be neglected in first approximation.

It is generally recognised that CH_4 formation is only efficient in a narrow pH range around neutrality (pH from 6.4 to 7.8). The effect of flooding is to increase the pH in acid soil, while it decreases the pH in alkaline soil. The increase of pH in acid soils is mainly due to the reduction of acidic Fe^{3+} to Fe^{2+} which simultaneously reduces the Eh. The addition of nitrate as chemical fertiliser to flooded soils may suppress the production of CH_4, because nitrate acts, as well as Fe^{3+}, Mn^{4+}, as a terminal electron acceptor in the absence of molecular oxygen during anaerobic respiration, and poises the redox potential of soils at values such that the activity of strict anaerobes is prevented. The addition of sulphate may also inhibit methane production for similar reasons as nitrate.

There are three processes of CH_4 release into the atmosphere from rice fields. Diffusion loss of CH_4 across the water surface is the least important process. Methane loss as bubbles (ebullition) from paddy soils is a common and significant mechanism, especially if the soil texture is not clayey. During land preparation and initial growth of rice, ebullition is the major release mechanism. The third process is CH_4 transport through rice plants, which has been reported as the most important phenomenon (Seiler et al., 1984; Schütz et al., 1989b).

Many researchers reported that more than 90 per cent of total CH_4 emitted during the cropping season is released by diffusive transport through the aerenchyma system of the rice plants and not by diffusion or ebullition. Emission through rice plant, may be expected to show great seasonal variations as a function of changes in soil conditions and variations in plant growth.

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[15] Redox (Eh) refers to oxidation-reduction, two processes that take place simultaneously. Oxidation is the loss of an electron by an atom, and reduction is the gain of an electron by an atom.

Methane emission rates are also a function of the partial pressure of CH_4 in the soil. Part of the CH_4 produced in the soil is consumed in the oxidised rhizosphere of rice roots or in the oxidised soil-floodwater interface. It is known that soil methanotrophic bacteria can grow with CH_4 as their sole energy source, and other soil bacteria, such as *Nitrosomonas* species are also able to consume CH_4 (Conrad, 1993). Methane is also leached to ground water, as a small part dissolves in water. Therefore a reduction in soil methane does not necessarily mean that all this CH_4 has been emitted into the atmosphere.

Global emissions from rice fields

The total harvested area of rice has increased from 86 Mha in 1935 to 144 Mha in 1985, which means an annual average increase of 1.05 per cent. The average annual increase was 1.23 per cent between 1959 and 1985. However, in the last few years, the rate of expansion of the total rice acreage has decreased (Minami, 1994).

Table 4-9 provides a summary of measured emissions at a number of specific research sites around the world. It should be noted that methane fluxes from paddy rice fields vary substantially from day to day, and during a day (e.g., day and night). The data presented here are based on frequent measurements which capture the diurnal variations, and variations over the growing season. Based on area and production statistics, with average emission values, a number of researchers have estimated global emissions from rice.

TABLE 4-9
REPRESENTATIVE METHANE EMISSIONS FROM RICE PADDY FIELDS IN VARIOUS LOCATIONS OF THE WORLD

Country	Location	Range of CH_4 flux mg/m^2/hr	Season total g/m^2	Experimental Treatment	Reference
Australia	Griffith	2.8	10	-	NGGIC, 1996
China	Beijing	14.6 - 48.9	27 - 91	OM, WM	Chen et al., 1993
	Beijing	9.4 - 26.8	12 - 39	MF, OM, WM	Yao and Chen, 1994a, 1994b
	Beijing	1.9 - 48.9	5.3 - 100.9	MF, OM, SO, WM	Shao, 1993
	Hangzhou, Zhejiang	6.9 - 50.6	14 - 82	MF, OM, SE	Wassmann et al., 1993a
	Nanjing, Jiangsu	2.6 - 14.3	6 - 34	MF, OM, WM	Chen et al., 1993
	Taoyuan, Hunan	6.5 - 56.2	12 - 115	MF, OM, SE	Wassmann et al., 1993b
	Tuzhu, Sichuan	58.0	167		Khalil et al., 1991
	Wuxian, Jiangsu	3.2 - 6.2	10 - 19	CU, MF, OM, SE, WM	Cai et al., 1994
India	Allahabad, Uttar Pradesh	0.2	0.5	NAV	Mitra, 1992
	Barrackpore, West Benegal	0.7, 20.2	1.8, 6.3	NAV	Mitra, 1992
	Cuttack, Orissa	2.7-7.2	7-19	CU	Mitra, 1992
	Faizabad, Uttar Pradesh	0.8	2	NAV	Mitra, 1992
	Garagacha, West Benegal	11	29	NAV	Mitra, 1992
	Jorhat, Assam	18.1	46	NAV	Mitra, 1992
	Kalyani, West Bengal	4.1	10.8	NAV	Mitra, 1992
	Koirapur, West Bengal	6.1	19	NAV	Mitra, 1992
	Madras, Tamil Nadu	5.8	11	NAV	Mitra, 1992
	New Delhi	0.02-0.21	0.06-0.58	MF	Mitra, 1992
	Purulia, West Bengal	4.2	11	NAV	Mitra, 1992
	Trivandrum, Kerala	5.1	9	NAV	Mitra, 1992
Indonesia	Taman Bogo, Lampung	18.0 - 27.1	31 - 47	MF, OM	Nugroho et al., 1994a
	Taman Bogo, Lampung	17.9 - 31.7	30 - 50	MF, OM	Nugroho et al., 1994b
	Sukamandi, West Java	8.7 - 20.2	19 - 44	WM, CU	Husin et al., 1995
Italy	Vercelli	5 - 28	18 - 75	MF, OM	Schütz et al., 1989a

Table 4-9 (cont.)
Representative Methane Emissions From Rice Paddy Fields In Various Locations of the World

Country	Location	Range of CH_4 flux mg/m^2/hr	Season total g/m^2	Experimental Treatment	Reference
Japan	Kawachi	16.3	45		Yagi and Minami, 1990a; Minami, 1994
	Mito	1.2 - 4.1	4 - 13	MF, OM	Yagi and Minami, 1990a; Minami, 1994
	Ryugasaki	2.8 - 15.4	11 - 28	MF, OM	Yagi and Minami, 1990a; Minami, 1994,
	Ryugasaki	1.9 - 7.9	7 - 12	WM	Yagi and Minami, 1990a; Minami, 1994
	Taya	7.0	26		Yagi and Minami, 1990a; Minami, 1994
	Tsukuba	0.2 - 0.4	<1.1	OM	Yagi and Minami, 1990a; Minami, 1994
Korea	Suwon	0.66 - 4.55	9 - 60	OM, WM	Shin et al., 1995
Philippines	Los Banos	0.8 - 18.5	2 - 42	MF, OM	Neue et al., 1994
	Los Banos	3.3 - 7.9	7 - 19	SE	Wassmann et al., 1994
Spain	Savilla	4	12		Seiler et al., 1984
Thailand	Ayutthaya	3.3 - 7.9	13 - 20	CU, OM, WM	Siriratpiraya, 1990
	Bang Khen	4.3 - 21.7	16 - 55	SE	Minami, 1994; Yagi et al., 1994b
	Chai Nat	1.6	4		Minami, 1994 Yagi et al., 1994b
	Chiang Mai	3.7 - 5.5	9 - 13	MF, OM	Jermsawatdipong et al., 1994
	Chiang Mai	9.0 - 9.5	20 - 21	CU	Siriratpiriya et al 1995
	Khlong Luang	3.8	8		Minami, 1994 Yagi et al., 1994b
	Khon Kaen	23.0	76		Minami, 1994; Yagi et al., 1994b

TABLE 4-9 (CONT.)
REPRESENTATIVE METHANE EMISSIONS FROM RICE PADDY FIELDS IN VARIOUS LOCATIONS OF THE WORLD

Country	Location	Range of CH_4 flux mg/m^2/hr	Season total g/m^2	Experimental Treatment	Reference
Thailand (cont.)	Nakompathom	9.4-12.0	25-32	SE	Tomprayoon et al., 1991
	Pathumthani	1.9 - 4.6	5 - 11	MF, OM	Jermsawatdipong et al., 1994
	Phitsanulok	6.6 - 7.2	17 - 18	SE	Katoh et al, 1995
	Phrae	16.6 - 22.2	51 - 69	SE	Minami, 1994; Yagi et al., 1994b
	Ratchaburi	3.2 - 42.5	9 - 117	MF, OM	Jermsawatdipong et al., 1994
	San Pa Tong	10.4 - 16.1	25 - 40	SE	Minami, 1994; Yagi et al., 1994b
	Surin	15.0 - 24.5	41 - 66	MF, OM	Jermsawatdipong et al., 1994
	Surin	13.3	41		Jermsawatdipong et al., 1994
	Suphan Buri	19.5 - 32.2	51 - 75	SE	Minami, 1994; Yagi et al., 1994b
USA	Beaumont, Texas	2.5 - 23.5	5 - 36	OM, SO	Sass et al., 1990, 1991a, 1991b
	Beaumont, Texas	0.6 - 6.3	1 -15	WM	Sass et al., 1992
	Crowley, Louisiana	10.2 - 17.9	21 - 37	MF	Lindau et al., 1991
	Crowley, Louisiana	12.6 - 85.0	22 - 149	MF, OM, SE	Lindau and Bollich, 1993
	Crowley, Louisiana	27 - 99	60 - 220	MF	Lindau, 1994
	Davis, California	3.4 - 10.4	18		Cicerone et al., 1983, 1992
	Knights Landing, California	0.5 - 18.8	1 - 58	MF, OM	Cicerone et al., 1992

Experimental treatment: CU - cultivars, MF - fertilisers, OM - organic matters, SE - seasons (early and late rices, or dry and rainy seasons), SO - soil types, WM - water management.

NAV = not available

Source: Modified from K Minami (1995)

Global emissions of CH_4 from rice paddies reported by several researchers are summarised in Table 4-10. Extrapolation of emission rates to a global scale is very difficult, because the effects of variations in agricultural practices, number of crops per year, soil types and other factors discussed above are uncertain.

The IPCC (IPCC, 1996) presented a candidate list of CH_4 sources to the atmosphere as annual release rates. The total annual source is constrained by the observed rate of atmospheric increase of concentrations and by the estimated atmospheric lifetime to be 535 Tg CH_4/yr. Rice paddies are listed as a source of 60 ± 40 Tg CH_4/yr.

TABLE 4-10
REPRESENTATIVE GLOBAL ANNUAL METHANE EMISSIONS FROM RICE FIELDS AS ESTIMATED BY DIFFERENT AUTHORS*

Reference	Estimate (Tg CH_4/yr)
Koyama (1964)	190
Ehhalt and Schmidt (1978)	280
Cicerone and Shetter (1981)	59
Khalil and Rasmussen (1983)	95
Seiler et al (1984)	35-59
Blake and Rowland (1988)	142-190
Crutzen (1985)	120-200
Holzapfel-Pschorn and Seiler (1986)	70-170
Cicerone and Oremland (1988)	60-170
Schütz et al. (1989a)	50-150
Aselman and Crutzen (1989)	60-140
Schütz et al. (1990)	50-150
Wang et al. (1990)	60-120
Neue et al. (1990)	25-60
Bouwman (1990)	53-114
Yagi and Minami (1990b)	22-73
IPCC (1990)	25-170
Minami(1994)	12-113
Sass (1994)	25-54
Parashar et al (1994)	20
IPCC (1996)	20-100

Source: Modified from K. Minami (1994)

4.3.2 Methods For Estimating Emissions

Emissions of methane from rice fields can be represented as follows:

EQUATION 1

$$F_C = EF \times A \times 10^{-12}$$

where:

F_C = estimated annual emission of methane from a particular rice water regime and for a given organic amendment, in Tg per year;

EF = methane emission factor integrated over integrated cropping season, in g/m^2;

A = annual harvested area cultivated under conditions defined above. It is given by the cultivated area times the number of cropping seasons per year, i.e., in m^2/yr.

The seasonally integrated emission factor is evaluated from direct field measurements of methane fluxes for a single crop.

In practice, it will be necessary to calculate the total annual emissions from a country as a sum of the emissions over a number of conditions. Total rice production can be divided into subcategories based on different biological, chemical and physical factors that control methane emissions from rice fields. In large countries, this may include different geographic regions. To account for the different conditions, F is defined as the sum of Fc (see Equation 1). This approach to emissions estimation can be represented as follows:

EQUATION 2

$$F = \Sigma_i \Sigma_j \Sigma_k EF_{ijk} \times 10^{-12}$$

where:

- ijk: are categories under which methane emissions from rice fields may vary.

For instance, i may represent water levels in the rice fields such as fields inundated for the duration of the growing season (flooded regime) or fields under water only intermittently. This occurs either under managed irrigation when water is not readily available or when rains do not maintain flooded conditions throughout the growing season (intermittent regime) as given in Table 4-12. j, k, may represent water regimes modified by other factors like organic inputs, soil textures, fertilisation regimes under each of the conditions represented by the index i, and so on. As more factors are identified, more categories need to be included. Inclusion of additional parameters should lead to an improvement of the estimate of the total emissions. The summation should include all cropping seasons.

The factors clearly identified by field experiments as being most important are (1) water regime with inorganic fertilisers (except sulphate-containing inorganic fertilisers which inhibit CH_4 production); (2) organic fertiliser applications; (3) soil type, and soil texture; (4) cultivar; and (5) agricultural practices such as direct seeding or transplanting. Data show that in continuously flooded fields, some types of organic fertilisers and certain cultivars lead to higher emissions compared to rice grown without organic amendments or intermittent or managed irrigation in which the fields are not continuously inundated and only where chemical fertilisers are used.

At present there are insufficient data to incorporate most of these factors. Nonetheless, the estimates can be improved substantially by incorporating the current knowledge on water regimes, organic amendments and soil types etc. For some countries the effects of organic fertiliser can be included.

4.3.3 Summary of Recommended Method

Basic Method

Data on rice cultivation under different water management techniques should be available from most of the important rice-producing countries. Therefore, the basic method for estimating emissions from each country includes estimates based on rice ecosystems (Kush,1984; Neue, 1989) relating to water regime (Table 4-12), namely:

- **Upland**: Fields are never flooded for a significant period of time.
- **Lowland**: Fields are flooded for a significant period of time.
  - **Irrigated**: Water regime is fully controlled.
    - **Continuously flooded:** Fields have standing water throughout the rice growing season and may only dry for harvest.
    - **Intermittently flooded :** Fields have at least one aeration period of more than 3 days during the cropping season.
      - **Single aeration:** Fields have a single aeration during the cropping season at any growth stage.
      - **Multiple aeration:** Fields have more than one aeration period during the cropping season.
  - **Rainfed**: Water regime depends solely on precipitation.
    - **Flood prone:** The water level may rise up to 50 cm during the cropping season.
    - **Drought prone**: Drought periods occur during every cropping season.
  - **Deep water rice**: Floodwater rises to more than 50 cm for a significant period of time during the cropping season.
    - Fields inundated with water depth from 50-100 cm.
    - Fields inundated with water depth > 100 cm.

The discussion refers to a single inventory (or base) year, (e.g., 1990) but an average over three years around the base year (e.g., 1989-1991) is recommended for the activity data, if possible.

For the inventory year, a number of input data are required.

- Area of rice cultivation by water management regime in square metres (m^2). As discussed above, that area is multiplied by the number of crops per year. This includes areas counted for each crop.
- Seasonally integrated flux (EF) emission values for areas of different rice ecosystems (water regimes) without organic amendments.
- of enhancement factors for organic amendments.

The result is methane emissions for each category. The total emissions for the country is the sum of the individual results for each category.

Default data

In many cases, especially at the beginning of the process, there will be important rice-growing areas for which specific fluxes will not be available. In such cases the regional and country-specific default data provided in Tables 4-12 and 4-13 can be used to carry out first order estimates. These data may also be used by national experts for comparison. Several ongoing activities to improve comparable measurement data have been identified. See Appendix for further information.

Area Statistics

Table 4-11 contains information on harvested area of rice according to statistics from the FAO Yearbook (UN, 1992), China Agricultural Yearbook (1990), IRRI RICE Almanac (IRRI, 1994) and World Rice Statistics (IRRI, 1993). Allocation of areas to categories, e.g., irrigated, rainfed (flood prone and lowland rainfed) and upland rice for main rice-producing countries were based on the IRRI Rice Almanac (IRRI, 1994) and for other rice-producing countries these categories were based on IRRI (1990), Huke (1982) and Grist (1986). Actual percentage of the irrigated, rainfed, and flood prone areas which are continuously flooded or have an aeration period greater than 3 days or multiple aerations, are to be obtained from the country specific data.

Seasonally Integrated flux values

Tables 4-12 and 4-13 provides default EF values for various categories of water regimes and multiplication factors for organic amendments. Emissions from upland rice are assumed to be 0 and ignored in the emission calculations.

For continuously flooded rice, a "model" average seasonally integrated emissions for rice-growing countries of the world was estimated from existing data (Table 4-13) to be 20 g/m^2. These flux values are representative of flooded rice fields where organic fertiliser is not used.

For intermittently flooded rice, a simple correction is applied. Fluxes are taken to be 50 per cent of the flooded (non-organic) value of 20 g/m^2 for single aeration and 20 per cent for multiple aeration. For other water regimes new default values are given in Table 4-2. For irrigated and continuously flooded, lowland rice ecosystems, the default seasonally integrated methane emission is 20 g/m^2 (see Table 4-13) for soils 'without organic amendments'. For conversion to methane emissions from soils 'with organic amendments', a default correction factor of 2 (Range 2-5) is applied to the corresponding rice ecosystems for the 'without organic amendment' category. This is because organic amendments of flooded rice paddies increase methane emission to the atmosphere (Yagi and Minami, 1990a; Sass et al., 1991a, 1991b; Neue et al., 1994). A comprehensive review of methane flux measurements over the past decade from a variety of countries and with different organic amendments and inorganic fertiliser treatments, is presented by Minami (1995). The amount of methane that is emitted as a result of organic soil amendments depends greatly on the amount and condition of readily available decomposable carbon contained in the treatment. Schütz et al (1989a) observed increases from a control value of 28.6 g CH_4 m^2/yr to 68.4 g CH_4 m^2/yr with added rice straw, (a factor of 2.4). Cicerone et al. (1992) observed increases from a control value of 1.4 g CH_4 m^2/yr to up to 58.2 g CH_4 m^2/yr with added straw, a factor of over 40 times higher. In field studies in the Philippines, Denier van der Gon and Neue (1995) found that fields treated with green manure applied at a rate of 22 tonnes/ha emitted over twice as much methane as fields in which the application rate was 11 tonnes/ha.

Methane emission rates are highly sensitive to water management. Periodic drainage of irrigated rice paddies results in a significant decrease in methane emissions. Yagi and Minami (1990a) reported a decrease in methane emission rates as a result of a mid-season drainage in Japanese rice fields. Sass et al. (1992) found that a single midseason drain reduced seasonal emission rates by 50 per cent (from 9.27 g/m^2 to 4.86 g/m^2). In addition, multiple short periods of drainage (2-3 days) approximately every three weeks during the growing season reduced methane emissions to an insignificant amount (1.15 g/m^2) without decreasing rice grain yield. Yagi et al. (1996) compared a continuously flooded plot with constant irrigation with an intermittently drained plot with short-term draining periods several times during the rice growing season. Total seasonal methane emission rates during the cultivation period were 14.8 g/m^2 and 8.6 g/m^2 for 1991 and 9.5 g/m^2 and 5.2 g/m^2 for 1993 in the continuously flooded and intermittently drained plots, respectively. Scaling factors in Table 4-12 have been developed using the data from the literature.

Default values in Tables 4-12 and 4-13 can be used for initial calculation where local measurements are not adequate. However, national experts are encouraged to use locally available data if available. If this is done it is important to ensure that these coefficients are based on a sufficient number of measurements to capture the variability and produce a representative seasonal average value, which is needed for inventory calculations (see Appendix).

Possible Refinements

National experts are encouraged to go beyond the basic method, and add as much detail as can be scientifically justified, based on laboratory and field experiments on various amendments and theoretical calculations, to arrive at the estimate of emissions from rice cultivation in their country. These details should be incorporated into subcategories (indices j,k in Equation 2) under each of the main water management categories in Equation 2 so that they can be compared at that level with data from other countries.

> For example:
>
> Where emission data are available for different fertiliser types, this may be incorporated into the calculations. Each category, (e.g., continuously flooded) would be further divided as follows:

F (continuously flooded) = F (flooded chemical) + F (flooded/organic amendment)

This procedure would then be repeated for as many separate subcategories as have been defined. Each amendment may be incorporated in the same manner.

In all cases, the emission inventory must be fully documented. The documentation has two aspects. First, method of calculation must be specified as in Equation 2. Matrices of amendments must be delineated. Second, all data and original sources must be referenced, if not included explicitly as part of the inventory report. It is desirable in all cases to rely on published information, whether from the county's government/scientific institutions an international organisation such as the UN-FAO, or the scientific literature.

TABLE 4-11

DEFAULT ACTIVITY DATA - HARVESTED RICE

Country or Region	1990 Area (1000s ha)	Irrigated[a] (%)	Upland Rice (%)	Rainfed[b] (%)
Americas				
USA	1114	100	0	0
Belize	2	10	90	0
Costa Rica	53	10	90	0
Cuba	150	100	0	0
Dominican Rep	93	98	2	0
El Salvador	15	10	90	0
Guatemala	15	10	90	0
Haiti	52	40	60	0
Honduras	19	10	90	0
Jamaica	0	40	60	0
Mexico	123	41	59	0
Nicaragua	48	10	90	0
Panama	92	5	95	0
Puerto Rico	0	75	25	0
Trinidad & Tobago	5	45	55	0
Argentina	103	100	0	0
Bolivia	110	25	75	0
Brazil	3945	19	75	6 (0 + 6)
Chile	35	79	21	0
Columbia	435	67	23	10 (0 + 10)
Ecuador	266	40	10	50
Guyana	68	95	5	0
Paraguay	34	50	50	0
Peru	185	84	16	0
Surinam	58	100	0	0
Uruguay	108	100	0	0
Venezuela	119	90	10	0

TABLE 4-11 (CONT.)
DEFAULT ACTIVITY DATA - HARVESTED RICE

Country or Region	1990 Area (1000s ha)	Irrigated[a] (%)	Upland Rice (%)	Rainfed[b] (%)
Asia				
Brunei	1	79	21	0
Hong Kong	0	100	0	0
Syria	0	100	0	0
Turkey	52	100	0	0
India	42321	53 (16 + 37)	15	32 (16 + 16)
Pakistan	2113	100	0	0
Bangladesh	10435	22	8	70 (23 + 47)
Myanmar	4760	18	6	76 (24 + 52)
Nepal	1445	23	3	74 (8 + 66)
Afghanistan	173	100	0	0
Bhutan	25	50	4	46 (42 + 4)
China [3]	33265	93	2	5 (0 + 5)
Indonesia	10502	72 (22 + 50)	11	17 (10 + 7)
Iran	570	100	0	0
Iraq	78	100	0	0
Japan	2074	99 (2 + 97)	1	0
Malaysia	639	66	12	22 (1 + 21)
Philippines	3319	61	2	37 (2 + 35)
Sri Lanka	828	37	7	56 (3 + 53)
Taiwan	700	97	3	0
Thailand	9650	7	1	92 (7 + 85)
Kampuchea	1800	8	2	90 (42 + 48)
Laos	638	2	37	61 (0 + 61)
Vietnam	6028	53	8	39 (11 + 28)
Democratic Republic of Korea	670	67	13	20
Republic of Korea	1242	100 (9 + 91)	0	0

TABLE 4-11 (CONT.)
DEFAULT ACTIVITY DATA - HARVESTED RICE

Country or Region	1990 Area (1000s ha)	Irrigated[a] (%)	Upland Rice (%)	Rainfed[b] (%)
Europe				
Albania	2	100	0	0
Bulgaria	11	100	0	0
France	20	100	0	0
Greece	15	100	0	0
Hungary	11	100	0	0
Italy	208	100	0	0
Portugal	33	100	0	0
Romania	37	100	0	0
Spain	81	100	0	0
Former USSR	624	100	0	0
Former Yugoslavia	8	100	0	0
Pacific				
Australia	102	100	0	0
Fiji	13	50	50	0
Africa				
Algeria	1	100	0	0
Angola	18	100	0	0
Benin	7	10	90	0
Burkina Faso	19	89	11	0
Burundi	12	25	75	0
Cameroon	15	25	75	0
C African Rep	10	25	75	0
Chad	39	25	75	0
Comoros	13	100	0	0
Congo	4	25	75	0
Egypt	436	100	0	0
Gabon	0	25	75	0
Gambia	14	90	10	0
Ghana	85	24	76	0
Guinea Bissau	57	25	75	0
Guinea	608	8	47	45

TABLE 4-11 (CONT.)
DEFAULT ACTIVITY DATA - HARVESTED RICE

Country or Region	1990 Area (1000s ha)	Irrigated[a] (%)	Upland Rice (%)	Rainfed[b] (%)
Ivory Coast	583	6	87	7
Kenya	15	25	75	0
Liberia	168	0	94	6
Madagascar	1160	10	14	76 (2 + 74)
Malawi	29	25	75	0
Mali	222	25	75	0
Mauritania	14	100	0	0
Morocco	6	100	0	0
Mozambique	109	25	75	0
Niger	29	35	65	0
Nigeria	1567	16	51	33 (33 + 0)
Rwanda	3	25	75	0
Senegal	73	25	75	0
Sierra Leone	339	1	67	32
Somalia	5	50	50	0
South Africa	1	100	0	0
Sudan	1	50	50	0
Swaziland	0	25	75	0
Tanzania	375	3	22	75 (0 + 75)
Togo	21	4	96	0
Uganda	37	25	75	0
Zaire	393	5	90	5
Zambia	11	25	75	0
Zimbabwe	0	25	75	0

a Numbers in brackets indicate continuously flooded and intermittently flooded respectively.

b Numbers in brackets indicate continuously flood-prone and drought-prone respectively.

c Values are currently being updated.

Sources: DeDatta (1975), Huke, (1982), Grist (1986), IRRI (1990), NGGIC (1996).

Notes: Areas were taken from FAO Yearbook (UN, 1992), China Agricultural Yearbook (1990), World Rice Statistics (IRRI, 1990) and IRRI Rice Almanac 1993-1995 (IRRI, 1993).

TABLE 4-12
SCALING FACTORS FOR METHANE EMISSIONS FOR RICE ECOSYSTEMS RELATIVE TO CONTINUOUSLY FLOODED FIELDS (WITHOUT ORGANIC AMENDMENTS)

Category	Sub-Category[a]			Scaling Factors (relative to emission factors for continuously flooded fields)
Upland	None			0
Lowland	Irrigated	Continuously flooded		1.0
		Intermittently flooded[b]	Single aeration	0.5 (0.2-0.7)
			Multiple aeration	0.2 (0.1-0.3)
	Rainfed	Flood prone		0.8 (0.5-1.0)
		Drought prone		0.4 (0-0.5)
	Deep water	Water depth 50-100 cm		0.8 (0.6-1.0)
		Water depth > 100 cm		0.6 (0.5-0.8)

[a] Other rice ecosystem categories, like swamps and inland, saline or tidal wetlands may be discriminated within each sub-category according to local emisson measurements.

[b] Defined as > 3 days aeration during the vegetative period.

Note: For irrigated and continuously flooded, lowland rice ecosystems, the default seasonally integrated methane emission is 20 g/m^2 (see Table 4-13) for soils ‘without organic amendments’. For conversion to methane emissions from soils ‘with organic amendments’, apply a default correction factor of 2 (Range 2-5) to the corresponding rice ecosystem for the ‘without organic amendment’ category.

TABLE 4-13
SEASONALLY INTEGRATED METHANE EMISSION FACTORS FOR CONTINUOUSLY FLOODED RICE WITHOUT ORGANIC FERTILISER IN VARIOUS LOCATIONS OF THE WORLD

Country	Seasonally Integrated Emission Factor, EF[a] (g/m^2)	Literature/Remarks
Australia	22.5	NGGIC, 1996
China	13 (10-22)	Wassman et al., 1993a
India	10 (5 - 15)	Mitra, 1996 Parashar et al., 1996
Indonesia	18 (5 - 44)	Nugroho et al., 1994a,b
Italy	36 (17-54)	Schütz et al., 1989a
Japan	15	Minami, 1995
Republic of Korea	15	Shin et al., 1995
Philippines	(25 - 30)	Neue et al., 1994; Wassman et al., 1994
Thailand	16 (4 - 40)	Towpryaoon et al., 1993
USA (Texas)	25 (15 - 35)	Sass and Fisher, 1995
Arithmetic Mean [b]	20 (12-28)	-

[a] It is recognised that the emission factors presented in Table 4-13 will need to be periodically updated as better data become available. However, this dataset represents the best available information at the time of compilation.

[b] The arithmetic mean of the seasonally integrated emission factor, EF, is derived from the values shown in Table 4-13. The range of emission factors is defined as the standard deviation about the mean.

Appendix

Intercomparability of Methane Emission Data from Rice Cultivation

Chamber measurements

Each laboratory should provide a standardised emission program of control flux measurements to ensure the intercomparibilty and intercalibration of extended data sets.

An emission standardisation programme will consist of a specified experimental plan for seasonal and annual flux measurements along with specific accompanying data on location and climate, soil, water management, plant characteristics, fertiliser treatment and a detailed cropping calendar.

- Methane flux measurements should be recorded at least twice per week over an entire season. Experiments should be continued during fallow and/or alternate cropping times as well as during the entire normal local rice growing season including land preparation. In areas where double or triple cropping is practised, data should be collected for all growing seasons.
- Since emissions are strongly influenced by daily temperature changes, the diel pattern of emission (6-12 flux rates within a 24 hour period) should be determined at least three times during the season.
- A data log of all agricultural events should be kept., e.g., transplant date, panicle initiation, heading, anthesis, harvest, etc. as well as fertilisation, water management schedule, weeding schedule, herbicide and pesticide treatments.
- At the time of each flux measurement, one should also collect the air temperature, flood water temperature, and the soil temperature.
- Fertilisation treatment in the standardisation (continuous irrigation) plots should be according to local practices, but limited to inorganic fertiliser. The application rate as kg N ha^{-1} and number and timing of applications should be reported.
- Fertilisation treatment in other experimental areas, including organic fertilisers, should reflect local practices. Amounts, type, and timing of applications should be reported for each phase of the cropping sequence at all scales.
- Standardisation chamber plots are to be kept flooded from shortly before transplanting until maturity. During flood, the water should be kept at a 5 cm minimum constant depth. A daily log should be kept of the amount of water added and when.

  Other water management regimes should be investigated when they are practised locally. A daily log should be kept of times and durations of draining, the amounts of water added and other applicable data.

For further information, *see 'Global Measurements Standards of methane emissions for irrigated rice cultivation'*, IGAC (1994).

4.4 Greenhouse Gas Emissions from Agricultural Burning

4.4.1 Introduction

Where there is open burning associated with agricultural practices, a number of greenhouse gases (GHGs) are emitted from combustion. All burning of biomass produces substantial CO_2 emissions. However, in agricultural burning, the CO_2 released is not considered to be **net** emission. The biomass burned is generally replaced by regrowth over the subsequent year. An equivalent amount of carbon is removed from the atmosphere during this regrowth, to offset the total carbon released from combustion. Therefore the long term net emissions of CO_2 are considered to be zero. Agricultural burning releases other gases in addition to CO_2 which are by-products of incomplete combustion: methane, carbon monoxide, nitrous oxide, and oxides of nitrogen, among others. These non-CO_2 trace gas emissions from biomass burning are net transfers from the biosphere to the atmosphere. It is important to estimate these emissions in national inventories.[16]

There are two major types of agricultural burning addressed in this section – savanna burning and field burning of crop residues. The approach is essentially the same as that used for non-CO_2 trace gases for all burning of unprocessed biomass, including traditional biomass fuels and open burning of cleared forests. For all these activities, there is a common approach in the proposed methodology, in that crude estimates of non-CO_2 trace gas emissions can be based on ratios to the total carbon released. The carbon trace gas releases (CH_4 and CO) are treated as direct ratios to total carbon released. Non-methane volatile organic compounds (NMVOCs) can be treated in a similar way. However, no default values for NMVOC are provided in this version of the *Guidelines*. To handle nitrogen trace gases, nitrogen to carbon ratios are used to derive total nitrogen released and then emissions of N_2O and NO_x are estimated based on ratios of these gases to total nitrogen released. Tables 4-15 and 4-16 provide suggested default values for non-CO_2 trace gas emission ratios. These are presented with ranges, which

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[16] For biomass combustion, CO_2 emissions are frequently not considered **net** emissions, and this is the case for agricultural burning. One could argue, in such cases, that this burning could be considered a short term sink of CO_2. That is, a portion of carbon in biomass is being released as **net** emissions of CH_4 and CO, while regrowth is removing the full amount of the original carbon from the atmosphere in the next cycle. Each year plants take up a certain amount of carbon from the atmosphere. When they are burned some of that carbon is converted to CO, and CH_4, so that an amount less than the total CO_2 which was taken up by the plants is re-emitted as CO_2. See Howden et al. (1996), for a more detailed discussion of this proposal. Treating emissions of CO and CH_4 to the atmosphere, as a sink for atmospheric CO_2, however, is inconsistent with the proposed IPCC emissions methodology. In particular, the other carbon compounds emitted are converted back into CO_2 in the atmosphere over periods of days up to a decade or so. Thus, over the time horizons of interest for CO_2, (i.e. more than 100 years) there is no sink of CO_2.

emphasise their uncertainty. However, the basic calculation methodology requires that users select a best estimate value.[17]

The calculation of immediate trace gas emissions, based on the default emission ratios provided in Tables 4-15 and 4-16, produces relatively crude estimates with substantial uncertainties.[18] Use of specific emission ratios which vary by type of burning, region, etc. may allow for more precise calculations. The calculations described here ignore the contemporary fluxes associated with past burning activities. These delayed releases are known to exist, but are poorly understood at present. This and other possible refinements are discussed at the end of this section.

4.4.2 Prescribed Burning of Savannas

Background

The term savanna refers to tropical and subtropical vegetation formations with a predominantly continuous grass cover, occasionally interrupted by trees and shrubs (Bouliere and Hadley 1970). These formations exist in Africa, Latin America, Asia, and Australia. The growth of vegetation in savannas is controlled by alternating wet and dry seasons: most of the growth occurs during the wet season; man-made and/or natural fires are frequent and generally occur during the dry season. The global area of savannas is uncertain, in part due to lack of data and in part due to differing ecosystem classifications. Estimates of the areal extent of savannas range from 1300-1900 million hectares world-wide, about 60 per cent of which are humid savannas (annual rainfall of 700 mm or more) and 40 per cent are arid savannas (annual rainfall of less than 700 mm) (Bolin et al., 1979; Whittaker and Likens, 1975; Lanly, 1982; Lacey et al., 1982; and Hao et al., 1990). Large-scale burning takes place primarily in the humid savannas because the arid savannas lack sufficient grass cover to sustain fire. Humid savannas are burned every one to four years on average with the highest frequency in the humid savannas of Africa (as cited in Hao et al., 1990).

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[17] Emissions inventory developers are encouraged to provide estimates of uncertainty along with these best estimate values where possible or to provide some expression of the level of confidence associated with various point estimates provided in the inventory. Procedures for reporting this uncertainty or confidence information are discussed in *Volume 1: Reporting Instructions*.

[18] Emission ratios used in this section and presented in the tables are derived from Crutzen and Andreae (1990), Delmas (1993), Delmas and Ahuja (1993) and Lacaux, et al. (1993). They are based on measurements in a wide variety of fires, including forest and savanna fires in the tropics and laboratory fires using grasses and agricultural wastes as fuel. In many cases these ratios are general averages for all biomass burning. Research will need to be conducted in the future to determine if more specific emission ratios, e.g., specific to the type of biomass and burning conditions, can be obtained. Also, emission ratios vary significantly between the flaming and smouldering phases of a fire. CO_2, N_2O, and NO_x are mainly emitted in the flaming stage, while CH_4 and CO are mainly emitted during the smouldering stage (Lobert et al., 1990). The relative importance of these two stages will vary between fires in different ecosystems and under different climatic conditions, and so the emission ratios will vary. As inventory methodologies are refined, emission ratios should be chosen to represent as closely as possible the ecosystem type being burned, as well as the characteristics of the fire.

Savannas are intentionally burned during the dry season primarily for agricultural purposes such as ridding the grassland of weeds and pests, promoting nutrient cycling, and encouraging the growth of new grasses for animal grazing. Savanna burning may be distinguished from other biomass burning activities like open forest clearing because there is little net change in the ecosystem biomass in the savanna after the vegetation regrows during the wet season. Consequently, while savanna burning results in instantaneous gross emissions of CO_2, it is reasonable to assume that the net carbon dioxide released to the atmosphere is essentially zero because the vegetation typically regrows between burning cycles.[19] Savanna burning does release several other important trace gases: methane (CH_4), carbon monoxide (CO), nitrous oxide (N_2O), oxides of nitrogen (NO_x, i.e., NO and NO_2) and non-methane volatile organic compounds (NMVOCs).

Estimates of global emissions of these gases due to savanna burning have been based on estimates of the annual instantaneous gross release of carbon from this activity and on ratios of the other trace gases released from burning to the total carbon released by burning. Estimates of the annual instantaneous gross release of carbon from savanna burning are highly uncertain because of lack of data on:

- the aboveground biomass density of different savannas;
- the savanna area burned annually;
- the fraction of aboveground biomass which actually burns; and
- the fraction which oxidises.

The methodology that is proposed in the next section, although conceptually quite simple, takes these factors into account. The approach allows for estimation of non-CO_2 trace gases released by savanna burning, based on default data sets and on assumptions from average literature values for various regions and types of savannas. It also allows for more accurate national estimates if data and assumptions can be developed to reflect national average conditions accurately. Nonetheless, a wide variety of technical details and open scientific issues remain important research topics.

Calculations

There are two basic components to the calculation. First, it is necessary to estimate the total amount of carbon released to the atmosphere from savanna burning. These are not considered to be net emissions, but are needed to derive non-CO_2 trace gas emissions, which are net emissions. What is required is the annual area burned for the various types of savannas, where type is based primarily upon above and below ground biomass, and perhaps climatological conditions and nutrient status. It is generally recommended for all emissions from agriculture that three-year averages of activity data (e.g., hectares burned) be used instead of a single year's data where possible. This is especially important for savanna burning which is highly variable from year to year. This variability should also be taken into account by national experts in planning data collection programmes to provide more accurate inputs to national inventory calculations. If data are not directly available, estimates can be derived based on total savanna area[20] and

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[19] If grazing pressure coupled with burning too often reduces biomass (i.e., degrades the quality of savannas), then this needs to be considered as a carbon dioxide source. This is not assumed in the basic calculations but could be included as a refinement if considered important.

[20] Most countries with significant savanna area should have national statistics on the total area, but FAO publications (e.g. FAO, 1993) also provide country-specific estimates.

average percentages of savanna burned annually, as shown in Table 4-14. Based on the area and type of savanna burned, the amount of carbon released can be calculated (a reflection of biomass densities, fractions burned, carbon contents and fractions oxidised). The second component of the calculation is the same as for other biomass burning categories – emission ratios are applied to estimate the amount of trace gas released based on the amount of carbon released (Table 4-15 provides default emission ratios).

TABLE 4-14
REGIONAL SAVANNA STATISTICS

Region[a]	Fraction of Total Savanna that is Burned Annually	Aboveground Biomass Density (t dm/ha)	Fraction of Biomass Actually Burned	Fraction of Aboveground Biomass that is Living
Tropical America	0.50	6.6 ±1.8		
Tropical Asia	0.50	4.9		
Tropical Africa	0.75	6.6 ±1.6		
Sahel zone	0.05-0.15[b]	0.5-2.5[b]	0.95	0.20
North Sudan zone	0.25-0.50[b]	2-4[b]	0.85	0.45
South Sudan zone	0.25-0.50[b]	3-6[b]	0.85	0.45
Guinea zone	0.60-0.80[b]	4-8[b]	0.9-1.0	0.55
Australia	0.05-0.70	2.1-6		

Sources: Hao et al., 1990, except where noted. These figures are growing season average biomass values, considered most appropriate for general default values

[a] Note that these are ecological zones that do not correspond directly to areas with political boundaries of the same name. For example, the North and South Sudan zones include countries other than Sudan and run East-West across the African continent.

[b] Menaut et al. (1991) These figures are maximum biomass values. For these arid sub-regions, maximums are considered the most appropriate default values.

Note: Biomass density is in tonnes of dry matter (dm) per hectare (ha).

The approach recognises that countries generally possess more than one savanna type, each with different characteristics, such as vegetative cover, that would affect trace gas emissions from burning. Also, the savanna area within a country may not be burned all at once, but rather in stages over the course of the dry season. Since the amount and nature (e.g., moisture content) of the vegetation changes during the year, factors such as biomass exposed to burning and fraction burned will vary among the savanna areas burned at different times. The data requested by this methodology focus upon country-specific types of savannas and the country-specific rate of burning for each type.[21]

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[21] If the area of savanna is not readily available, then the area of "open, broad-leaved forests," including open, broad-leaved, fallow areas, as defined by the UN Food and Agriculture Organization in FAO (1993) can be used as an estimate. This land area, corresponds to "mixed broad-leaved forest-grassland tree formations with a continuous dense grass layer in which the [woody vegetation covers] more than 10% [of the area]" (Lanly, 1982). FAO (1993) provides 1990 estimates of this area, by country, for tropical America, Asia, and Africa. Hao et al. (1990) provide an estimate of the humid savanna area in Australia, based on work by Lacey et al. (1982).

It is also recognised that national and regional estimates of the percentage of savanna area burned annually are highly uncertain. An example selection of regional estimates is included in Table 4-14. Though regional variability is great, the methodology, by focusing upon a simple classification of savanna type and the burning by type, can be implemented using data that are available to most countries. The methodology is intended to be flexible to allow users to define the savanna types and/or geographic subregions for calculations. It is strongly recommended that national experts consider dividing total savannas into woody savannas and grasslands, if possible. Many countries contain some systems which have significant aboveground woody biomass and others which have very little aboveground biomass other than grasses. These subcategories have significantly different biomass densities and fractions oxidised and should be accounted for separately rather than by averaging, if data are available. National experts are encouraged to carry out the calculations at the finest levels of detail for which credible data can be obtained. Finally, by varying assumptions about the rate and/or type of savannas burned, national experts can easily compare the sensitivity of the calculated emissions to the uncertainties in the data.

It should be noted that not all savanna burning occurs from anthropogenic causes. Some natural savanna fires obviously occur, and would occur in the absence of human intervention. As is the case with many human interactions with the biosphere, it is very difficult to establish the net effects of human intervention relative to the natural background, or the conditions which would have occurred naturally in the absence of intervention. In this situation, the recommended conservative default is that all fires are considered anthropogenic, unless they can be documented to be from natural causes. This clearly may overstate the emissions somewhat. National experts may choose to modify this assumption using expert judgement or other sources to allocate the anthropogenic and natural components, provided the rationale is clearly documented.

Step 1: Total Carbon Released From Savanna Burning

In order to calculate the carbon released to the atmosphere from savanna burning, these data are required for each category:

- Area of savanna;
- Fraction of savanna area burned annually;
- Average aboveground biomass density (tonnes dry matter/hectare) of savannas;
- Fraction of aboveground biomass which actually burns;
- Fraction of aboveground biomass that is living;
- Fraction of living and of dead aboveground biomass oxidised; and
- Fraction of carbon in living and dead biomass.

Not all of these data must be provided by the user. Initially one could pool the living and dead biomass if data are not available. More importantly, Table 4-14 provides much of the basic default data that only need to be refined for country-specific relevance. Given the data, the steps to calculate emissions are not overly difficult. One simply calculates from the area burned the total carbon released based upon the factors listed above. In addition to the data in Table 4-14, other recommended default values are included in the step-by-step discussion below.[22]

The following equations summarise the calculations to estimate the total carbon released due to the burning of savannas for each category:

EQUATION 1

Area of Savanna Burned Annually (ha)

=

Total Area of Savanna (ha) x Fraction Burned Annually

EQUATION 2

Biomass Burned (t dm)

=

Area of Savanna Burned Annually (ha)
x Aboveground Biomass Density (t dm/ha) x Fraction Actually Burned

EQUATION 3

Carbon Released from Live Biomass (t C)

=

Biomass Burned (t dm) x Fraction that is Live x Fraction Oxidised
x Carbon Content of Live Biomass (t C/t dm)

EQUATION 4

Carbon Released from Dead Biomass (t C)

=

Biomass Burned (t dm) x Fraction that is Dead x Fraction Oxidised
x Carbon Content of Dead Biomass (t C/t dm)

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[22] It is hoped that individual countries have this information since it is needed to execute the proposed methodology. Regional estimates of these statistics are provided by Menaut (1990) and Hao et al. (1990) and reproduced in a table. More country-specific research is clearly needed on this issue before accurate inventories can be developed. This research should include data on savanna area burned annually, savanna biomass densities, live fractions of biomass, burning efficiencies, and carbon contents of savanna biomass. In the meantime, default values can be used.

EQUATION 5

Total Carbon Released (t C)

=

C Released from Live Material (t C) + C Released from Dead Material (t C)

In the first equation, the savanna area in the country is multiplied by the percentage of the savanna area that is burned annually, if statistics on area burned annually are not directly available. If national experts have data on the area burned annually they should use this and begin with equation 2. In the second, area burned is multiplied by aboveground dry biomass per hectare (ha) on the savanna at the time of burning and the fraction of biomass which actually burns. Regional estimates of rates of savanna burning and biomass densities are presented in Table 4-14. The fraction actually burned accounts for the fact that when savannas are burned, not all of the biomass on each hectare is actually exposed to flame. If detailed information is not available, a general default value in the range of 0.80-0.85 is recommended (Delmas and Ahuja, 1993).

The aboveground biomass density before burning is a function of the type of savanna being burned and the time of year in which burning occurs.[23] The values for West African savannas provided in Table 4-14 correspond to mid-season fires, except for those of the Sahel where burning occurs early. If statistics on maximum biomass density and fraction of maximum biomass density present at the time of burning are not available, countries can use an average biomass density instead. According to this analysis, average savanna biomass densities are lowest in tropical Asia, at about 5 tonnes per hectare (t/ha) (Singh and Misra, 1978), average around 6.6 t/ha in tropical Africa and tropical America (San José and Medina, 1976; González-Jiménez, 1979; Coutinho, 1982; Hopkins, 1965; Haggar, 1970; Menaut and César, 1982; and Huntley and Morris, 1982). The densities range between 2 and 6 t/ha in Australia (Lacey et al., 1982). These estimates have an uncertainty of ± 30 per cent based on field measurements (Hao et al., 1990). As mentioned, these regional average densities are presented in Table 4-14 and can be used as default values if average biomass density for a specific country or savanna type is not known.

In the third and fourth equations, the living and the dead portions of aboveground biomass burned are multiplied by their respective fractions oxidised and carbon contents. Estimates of the fraction of aboveground biomass that is living for West African savannas range from 20 to 55 per cent (Table 4-14). Data suggest that for the live portion, the fraction which burns ranges between 65 and 95 per cent and for the dead portion essentially 100 per cent burns (Menaut et al., 1991). If fractions oxidised are not available, 80 per cent and 100 per cent for the living and dead portions, respectively, can be used. If country or ecosystem values are not available, then the values 0.45 t C/t dry biomass and 0.40 t C/t dry biomass can be used as default values for the carbon contents of the living and dead portions, respectively (Menaut et al., 1991).

The total carbon released from savanna burning (Equation 5) is estimated by summing the carbon released from the living and the dead biomass fractions, calculated in Equations 3 and 4.

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[23] Menaut et al. (1991) calculate this number by multiplying the maximum biomass density of the savanna (which generally is reached at the end of the growing season) by a coefficient that declines as the burning occurs later in the dry season.

Step 2: Emissions

Once the carbon released from savanna burning has been estimated, the emissions of CH_4, CO, N_2O, and NO_x can be calculated using emission ratios. (Default values are presented in Table 4-15.)[24] The amount of carbon released due to burning is multiplied by the emission ratios of CH_4 and CO relative to total carbon released to yield emissions of CH_4 and CO (each expressed in units of C). The emissions of CH_4 and CO are multiplied by 16/12 and 28/12, respectively, to convert to full molecular weights. NMVOC could also be calculated in the same way. However, default values are not available in this edition of the *Guidelines*.

To calculate emissions of N_2O and NO_x, first the carbon released is multiplied by the estimated ratio of nitrogen to carbon (N/C ratio) in savanna biomass by weight (0.006 is a general default value for savanna biomass burning (Crutzen and Andreae, 1990)). This yields the total amount of nitrogen (N) released from the biomass burned. The total N released is then multiplied by the ratios of emissions of N_2O and NO_x relative to the N released to yield emissions of N_2O and NO_x (expressed in units of N). To convert to full molecular weights, the emissions of N_2O and NO_x are multiplied by 44/28 and 46/14, respectively.[25]

The non-CO_2 trace gas emissions calculations from burning are summarised as follows:

CH_4 Emissions = (carbon released) x (emission ratio) x 16/12

CO Emissions = (carbon released) x (emission ratio) x 28/12

N_2O Emissions = (carbon released) x (N/C ratio) x (emission ratio) x 44/28

NO_x Emissions = (carbon released) x (N/C ratio) x (emission ratio) x 46/14

TABLE 4-15
EMISSION RATIOS FOR SAVANNA BURNING CALCULATIONS

Compound	Ratios	
$CH4$ [a]	0.004	(0.002 - 0.006)
CO [b]	0.06	(0.04 - 0.08)
$N2O$ [c]	0.007	(0.005 - 0.009)
NOx [c]	0.121	(0.094 - 0.148)

Sources:
[a] Delmas, 1993
[b] Lacaux, et al., 1993
[c] Crutzen and Andreae, 1990

Note: Ratios for carbon compounds, i.e., $CH4$ and CO, are mass of carbon compound released (in units of C) relative to mass of total carbon released from burning (in units of C); those for the nitrogen compounds are expressed as the ratios of mass of nitrogen compounds released relative to the total mass of nitrogen released from the fuel.

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[24] This approach is adapted from Crutzen and Andreae, 1990, with some values updated, based on more recent studies by Delmas (1993), Delmas and Ahuja (1993) and Lacaux et al. (1993).

[25] The molecular weight ratios given above for the emitted gases are with respect to the weight of nitrogen in the molecule. Thus for N_2O the ratio is 44/28 and for NO_X it is 46/14. NO_2 has been used as the reference molecule for NO_X.

4.4.3 Field Burning of Agricultural Residues

Background

Large quantities of agricultural wastes are produced, from farming systems world-wide, in the form of crop residue.[26] Burning of crop residues, like the burning of savannas, is not thought to be a net source of carbon dioxide (CO_2) because the carbon released to the atmosphere during burning is reabsorbed during the next growing season. However, crop residue burning is a significant net source of CH_4, CO, NO_x, and N_2O. This section accounts for emissions of these non-CO_2 gases from field burning of agricultural crop residues. Burning of agricultural crop residues as an energy source is covered in the *Energy* chapter, in the section on fuel Combustion.

The amount of agricultural wastes produced varies by country, crop, and management system. For example, cereal crops produce between 0.6 and 2.5 tonnes of straw per tonne of grain (Barnard, 1990; Ponnamperuma, 1984), and wetland rice cultivated under a moderate level of management in the Philippines was found to produce between 0.6 and 0.9 tonnes of straw per tonne of grain (Ponnamperuma, 1984). Approximately 3.1 billion tonnes of crop residue are produced each year, with about 60 per cent originating in the developing world, and 40 per cent in the developed world (Strehler and Stützle, 1987).

Burning of agricultural wastes in the fields is a common practice in the developing world. It is used primarily to clear remaining straw and stubble after harvest and to prepare the field for the next cropping cycle. In Southeast Asia, burning is the major disposal method for rice straw (Ponnamperuma, 1984), which accounts for about 31 per cent of the agricultural waste in the developing world. Sugar cane residues, which make up about 11 per cent of the world's agricultural waste, are primarily disposed of by burning (Crutzen and Andreae, 1990). It has been estimated that as much as 40 per cent of the residues produced in developing countries may be burned in fields, while the percentage is lower in developed countries (Barnard and Kristoferson, 1985). Another study suggests that approximately 425 Tg dry matter agricultural wastes (~200 Tg C) are burned in the fields in developing countries and that about one-tenth as much is burned in developed countries (Crutzen and Andreae, 1990).

Calculations

The methodology for estimating greenhouse gas emissions from burning of agricultural wastes is based, as in savanna burning, on 1) total carbon released, which is a function of the amount and efficiency of biomass burned and the carbon content of the biomass, and 2) the application of emission ratios of CH_4 and CO to total carbon released, and of N_2O and NO_x to total nitrogen released from biomass fires which are available from the scientific literature on biomass burning. It is generally recommended for all emissions from agriculture and land use change, that three-year averages of activity data (e.g., crop residues burned) be used if available.

Step 1: Total Carbon Released from Burning Agricultural Residues

Data required, for each crop type, to calculate the amount of carbon burned in agricultural wastes are listed below:

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[26] Barnard (1990) outlines several broad categories of crop residue: woody crop residues (coconut shells, jute sticks, etc.), cereal residues (rice and wheat straw, maize stalks, etc.), green crop residues (groundnut straw, soybean tops, etc.), and crop processing residues (bagasse, rice husks, etc.).

- Amount of crops produced with residues that are commonly burned;
- Ratio of residue to crop product;
- Fraction of residue burned;
- Dry matter content of residue;
- Fraction oxidised in burning; and
- Carbon content of the residue.

There are standard default or literature values available for many of these data. Table 4-17 provides a summary of available default data. The most important data for users to provide are the actual amount of crops produced (by type) with residues that are commonly burned. Annual crop production statistics by country for most of the crops from which residues are burned are given in the *FAO Production Yearbooks, FAO (1991)*. Users may also find the *United Nations World Trade Yearbooks* useful. Crop-specific data for each country on ratios of residue to crop, fraction of residue burned, dry matter content of residue, and carbon content of residue can be incorporated at any time to replace the default values. A potentially valuable data source is the recent BUN/UNCED study by Professor D. Hall (and others) of Kings College, London, Hall et all (1994). In this context, one should also note the book *Renewable Energy: Sources for Fuels and Electricity* edited by Johansson et al. (1992).

From production data one can estimate the actual material (in carbon units) that is burned. One simple procedure is shown below:

Total carbon released (tonnes of carbon) =

$\sum_{\text{all crop types}}$ annual production (tonnes of biomass per year),

x the ratio of residue to crop product (fraction),

x the average dry matter fraction of residue (tonnes of dry matter / tonnes of biomass),

x the fraction actually burned in the field,

x the fraction oxidised,

x the carbon fraction (tonnes of carbon / tonnes of dry matter)

It is highly desirable to use country-specific data for these values wherever possible. Example estimates of residue/crop product ratios, average dry matter fraction and carbon fraction for certain crops are presented in Table 4-17.[27] If no other data are available,

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[27] Dry matter (dm), or dry biomass, refers to biomass in a dehydrated state. According to Elgin (1991), the moisture content of crop residue varies depending on the type of crop residue, climatic conditions (i.e., in a humid environment the residue will retain more moisture than in an arid environment), and the length of time between harvesting and burning of the residue. From a simple perspective, one can use the dry matter content values in Table 4-17 to convert from total crop residue to dry matter. For example, 200 tonnes of crop residue with a moisture content of 10%, would have a dry matter content of 90%, equal to 180 tonnes of dry matter. To convert from dry matter to carbon content, an average value of 0.45 t C/t dm can be used in the cases

the following assumptions regarding the fraction of crop residue burned in the field can be used as very crude default factors: for developing countries 0.25, and for developed countries a much smaller share possibly, 0.10 or less.[28] A default value of 0.90 for fraction oxidised can be used to account for the approximate 10 per cent of the carbon that remains on the ground as a result of charcoal formation and other aspects of incomplete combustion (Seiler and Crutzen, 1980; and Crutzen and Andreae, 1990).

Step 2: Gas Emissions

Once the carbon released from field burning of agricultural residues has been estimated, the emissions of CH_4, CO, N_2O, and NO_x can be calculated based on emission ratios (default values are provided in Table 4-16).[29] The amount of carbon released due to burning is multiplied by the emission ratios of CH_4 and CO relative to total carbon to yield emissions of CH_4 and CO (each expressed in units of C). The emissions of CH_4 and CO are multiplied by 16/12 and 28/12, respectively, to convert to full molecular weights. NMVOC could also be calculated in the same way. However, default values are not available in this edition of the *Guidelines*.

To calculate emissions of N_2O and NO_x, first the total carbon released is multiplied by the estimated N-C ratio of the fuel by weight to yield the total amount of nitrogen (N) released. Some crop-specific values are given in Table 4-17 and 0.015 is a general default value for crop residues.[30] The total N released is then multiplied by the ratios of emissions of N_2O and NO_x relative to the N content of the fuel to yield emissions of N_2O and NO_x (expressed in units of N). To convert to full molecular weights, the emissions of N_2O and NO_x are multiplied by 44/28 and 46/14, respectively.[31]

The calculation for trace gas emissions from burning is summarised as follows:

CH_4 Emissions = Carbon Released x (emission ratio) x 16/12

CO Emissions = Carbon Released x (emission ratio) x 28/12

N_2O Emissions = Carbon Released x (N/C ratio) x (emission ratio) x 44/28

NO_x Emissions = Carbon Released x (N/C ratio) x (emission ratio) x 46/14

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where crop specific data are not available. The terms dry matter and dry biomass are used interchangeably in this text.

[28] Crutzen and Andreae, 1990. The estimates are very speculative and should be used with caution. The actual percentage burned varies substantially by country and crop type. This is an area where locally developed, country-specific data are highly desirable. As this issue is studied further, it may be possible to incorporate more accurate, country-and crop-specific percentages into future editions of the *Guidelines*.

[29] This approach is adapted from Crutzen and Andreae, 1990, with some values updated based on more recent studies by Delmas (1993), Delmas and Ahuja (1993) and Lacaux et al. (1993).

[30] Crop specific values are generally in the range of 0.01-0.02, from Crutzen and Andreae, 1990, so that 0.015 can be used as a generally representative value if no other information is available.

[31] The molecular weight ratios given above for the emitted gases are with respect to the weight of nitrogen in the molecule. Thus for N_2O the ratio is 44/28 and for NO_X it is 46/14. NO_2 has been used as the reference molecule for NO_X.

TABLE 4-16
EMISSION RATIOS FOR AGRICULTURAL RESIDUE BURNING CALCULATIONS

Compound	Ratios
CH_4 [a]	0.005 Range 0.003 - 0.007
CO [b]	0.06 Range 0.04 - 0.08
N_2O [c]	0.007 Range 0.005 - 0.009
NO_x [c]	0.121 Range 0.094 - 0.148

Sources:

[a] Delmas, 1993

[b] Lacaux, et al., 1993

[c] Crutzen and Andreae, 1990

Note: Ratios for carbon compounds, i.e., CH_4 and CO, are mass of carbon compound released (in units of C) relative to mass of total carbon released from burning (in units of C); those for the nitrogen compounds are expressed as the ratios of mass of nitrogen compounds relative to the total mass of nitrogen released from the fuel.

TABLE 4-17
SELECTED CROP RESIDUE STATISTICS

Product	Residue/Crop Product	Dry Matter Fraction	Carbon Fraction (% dm)	Nitrogen-Carbon (N-C) Ratio
Wheat	1.3	0.78-0.88	0.4853	0.012
Barley	1.2	0.78-0.88	0.4567	
Maize	1	0.30-0.50	0.4709	0.02
Oats	1.3			
Rye	1.6			
Rice	1.4	0.78-0.88	0.4144	0.014
Millet	1.4			0.016
Sorghum	1.4			0.02
Pea	1.5			
Bean	2.1			
Soya	2.1			0.05
Potatoes	0.4	0.30-0.60	0.4226	
Feedbeet	0.3	0.10-0.20 [a]	0.4072[a]	
Sugarbeet	0.2	0.10-0.20 [a]	0.4072[a]	
Jerusalem artichoke	0.8			
Peanut	1			

Sources: Strehler and Stützle, 1987
Sugarbeet data from Ryan and Openshaw, 1991
Nitrogen content from Barnard and Kristoferson, 1985
[a] These figures are for beet leaves.

Possible refinements of the basic calculations

The basic calculations presented above address the immediate release of non-CO_2 trace gases when savannas or crops are burned. This is believed to be the most important effect of biomass burning on GHG emissions and the best characterised at present. However, there are other issues not treated in these calculations. The effect of past burning on current emissions is one such issue. The longer-term release or uptake of these gases following burning is an important research issue and may be included in future refinements of the calculations. In particular, grassland fires (savanna burning) may perturb the soils sufficiently to release additional N_2O and NO_x. Little is known about the magnitude of this flux so these emissions are not included in the first application of the methodology. It is less likely that such delayed releases are significant after field burning of agricultural residues, but this may also require further study.

Long term changes in soil carbon are certainly possible as a result of agricultural practices. In the land use change and forestry chapter, there is a general default assumption that soil carbon is gradually lost from agricultural lands over many years after forests are cleared. In fact, depending on the specific agricultural and soil management practices (including burning) which are used, there may be a variety of effects on soil carbon. For example, repeated burning of savannas and crop residues in fields may cause an increase in the amount of carbon stored in the soil over time. This is an area which requires further research and may lead to more detailed emissions estimation methods in the future.

In addition, agricultural practices such as overgrazing, which degrade the productivity of grasslands or other agricultural lands, reduce the amount of aboveground biomass which regrows. These could be considered sources of gradual emissions of carbon dioxide. This situation is not included in the basic calculations, but could be included in more refined calculations. National experts should determine whether or not this is important for their country, and whether or not they are able to provide input data.

A similar long-term effect can be observed from savanna burning. If the savannas are burned too frequently, complete regrowth may not occur. In this situation, grasslands can degrade over time, resulting in long-term losses of both aboveground and soil carbon. If this condition is significant, national experts are encouraged to estimate the annual effect and explain the assumptions and data used.

Another possible refinement is to account for carbon that might be sequestered through the use of agricultural residues to make durable products (e.g., bricks, composite boards). The default assumption is that the carbon sequestered in such activities on an annual basis is small, and can be ignored in the calculations, that is, as long as the stocks of such products are not significantly increasing or decreasing over time, their effect on emissions/removals can be ignored. If national experts believe the effects may be significant, they are encouraged to estimate this carbon sink and document the assumptions and data used.

4.5 Greenhouse Gas Emissions from Agricultural Soils

4.5.1 Introduction

Agricultural soils may emit or remove nitrous oxide (N_2O), carbon dioxide (CO_2), and/or methane (CH_4).

This section presents the method for calculating national emissions of N_2O from agriculture soils. This N_2O method is a significant revision of the 1995 *IPCC Guidelines for National Greenhouse Gas Inventories* (IPCC, 1995). Carbon dioxide emissions from soils are described in the Chapter on Land-use Change and Forestry.

For N_2O, the 1995 Methodology was evaluated and the present method includes more sources of N_2O and more explicit information about emission factors. Three sources of N_2O are distinguished in the methodology: (i) direct emissions from agricultural soils, (ii) direct soil emissions from animal production (including stable emissions to be reported under Manure Management (Section 4.2)), and (iii) N_2O emissions indirectly induced by agricultural activities. The methodology is an approach which requires only input data available from FAO databases. To retain consistency with the source categories defined in the *IPCC Guidelines* (1995), not all N_2O emissions from these sources are reported here. Emissions from animal waste management systems (i.e., before manure is added to soils), and from manure used for fuel, are reported under Section 4.2 and Energy Chapter, respectively, whereas emissions from human sewage are reported in the Waste Chapter. However, the description of the N cycle which produces these N_2O sources is included in Section 4.5.4 to provide the reader with a coherent picture of the N cycle as related to agricultural soils.

Anthropogenic input into agricultural systems include synthetic fertiliser, nitrogen from animal wastes, nitrogen from increased biological N-fixation, and nitrogen derived from cultivation of mineral and organic soils through enhanced organic matter mineralisation. Nitrous oxide may be produced and emitted directly in agricultural fields, animal confinements or pastoral systems or be transported from agricultural systems into ground and surface waters through surface runoff, nitrogen leaching, consumption by humans and introduction into sewage systems which transport the nitrogen ultimately into surface water. Ammonia and oxides of N (NO_x) are also emitted from agricultural systems and may be transported off-site and serve to fertilise other systems which leads to enhanced production of N_2O.

Agricultural systems are considered as being the same throughout the world and this methodology does not take into account different crops, soils and climates which are known to regulate N_2O production. These factors are not considered because limited data are available to provide appropriate emission factors. Countries which have data to show that default data are inappropriate for their country should include a full explanation for the values used. The method also uses a linear extrapolation between N_2O emissions and fertiliser nitrogen application and in the indirect emissions section does not account for the probable lag time between nitrogen input and ultimate production of N_2O as a result of this nitrogen input into agricultural soils.

Using midpoint emission factors this current methodology and the FAO data (FAO, 1990a;b), global N_2O–N emissions for the year 1989 were estimated. Direct emissions from agricultural soils totalled 2.5 Tg N, direct emissions from grazing animals totalled 1.6 Tg N and indirect emissions resulting from agricultural nitrogen input into the atmosphere and aquatic systems totalled 1.9 Tg N_2O–N. These estimates show that each of the three components of agriculture considered contribute about the same amount of N_2O to the global atmospheric budget. The N_2O input to the atmosphere from agricultural production as a whole has apparently been underestimated by the previous IPCC methodology by at least 70 per cent.

4.5.2 Direct N_2O Emissions from Agricultural Soils (Including Glasshouse Systems Farming and Excluding Effects of Grazing Animals)

Anthropogenic sources of N_2O can be biogenic (e.g., enhanced N_2O production by bacteria in fertilised fields) or abiogenic (e.g., formation during burning processes). Several studies indicate that anthropogenic sources are largely biogenic, with agriculture as a major contributor (Bouwman et al., 1995; Mosier et al., 1995a;b).

Biogenic production of N_2O in the soil results primarily from the nitrification and denitrification processes. Simply defined, nitrification is the aerobic microbial oxidation of ammonium to nitrate and denitrification is the anaerobic microbial reduction of nitrate to dinitrogen gas. Nitrous oxide is a gaseous intermediate in the reaction sequences of both processes which leaks from microbial cells into the soil atmosphere. Major regulators of these processes are temperature, pH and soil moisture content.

In most agricultural soils, biogenic formation of N_2O is enhanced by an increase in available mineral nitrogen, which in turn increases nitrification and denitrification rates. Addition of fertiliser N, therefore, directly results in extra N_2O formation. Most studies on N_2O emissions from agricultural soils investigate the difference in N_2O production between fertilised and unfertilised fields. Emissions from unfertilised fields are considered background emissions. However, actual background emissions from agricultural soils may be higher than historic natural emissions as a result of enhanced mineralisation of soil organic matter. This is particularly observed in organic soils in both cold and warm climates over the globe (Bouwman and van der Hoek, 1991; Kroeze, 1994). Background emissions may also be lower than historic emissions due to depletion of soil organic matter (Groffman et al., 1993).

OVERVIEW OF SOURCES

Introduction

The following sources and sink of N_2O can be distinguished.

- Synthetic fertilisers;
- Animal excreta nitrogen used as fertiliser;
- Biological nitrogen fixation;
- Crop residue and sewage sludge application;
- Glasshouse farming;
- Cultivation of high organic content soils;
- Soil sink for N_2O.

All of these N_2O sources are included in the methodology, except for sewage sludge application and the soil sink for N_2O. These sources and sinks are not estimated because emissions are negligible or data are insufficient.

Synthetic fertilisers

Synthetic fertilisers are an important source of N_2O. The amount of synthetic fertiliser nitrogen applied to agricultural fields world-wide is well documented in the FAO database (FAO Annual Fertiliser Yearbook). Studies on fertiliser-induced N_2O emissions have been revised by Bouwman (1994,1995), Cole et al. (1995), Mosier (1994) and Mosier et al. (1995a;b). Conversion factors used by Cole et al. (1995) and Mosier et al. (1995b) are recommended. This recommendation is based on discussions conducted during the preparation of the Cole et al. (1995) document and from Bouwman (1994) who estimated that 0.0125 ± 0.01 of the applied nitrogen was directly emitted as N_2O (see Table 4-18). This range encompasses more than 90 per cent of the published field emission values summarised in Bouwman (1994). Note that most of the information in Bouwman's summary is derived from field studies conducted in temperate regions of the world since few annual flux measurements have been made in tropical agricultural systems.

TABLE 4-18

DEFAULT EMISSION FACTORS FOR DIRECT EMISSIONS OF N_2O

EF_1 (fraction of N-input, kg N_2O-N/kg N)	0.0125 (0.0025-0.0225)[a]
EF_2 (kg N_2O-N/ha/yr)	5 temperate and 10 tropical (2-15) [a]

[a] Indicates range

Animal excreta nitrogen used as fertiliser

Although the amount of nitrogen used as fertiliser from animal excreta is more uncertain than the amount of synthetic fertiliser used, estimates can be made, based on animal population and agricultural practices, as shown in Section 4.5.3 of this Chapter (Table B-1, Appendix B). To account for the loss of fertiliser from NH_3 volatilisation and emission of nitric oxide (NO) through nitrification after fertiliser is applied to fields, NH_3 volatilisation and NO emission factors are needed. Even though climate, soil, fertiliser placement and type, and other factors influence NH_3 volatilisation and NO_x emissions, a default emission factor of 0.1 (kg NH_3–N + NO_x–N emitted/kg N applied) can be used for synthetic fertilisers and 0.2 (kg NH_3–N + NO_x–N emitted/kg N applied) for animal waste fertiliser (Table 4-19) (0.2 is used for animal waste because of the potentially larger NH_3 volatilisation; see Section 4.5.4). The amount of nitrogen from these sources available for conversion to N_2O is therefore equal to 90 per cent of the synthetic fertiliser nitrogen applied and 80 per cent of the animal waste nitrogen applied (Schepers and Mosier, 1991).

Biological nitrogen fixation

Although the amount of nitrogen fixed by biological nitrogen fixation in agricultural systems can be estimated, the N_2O conversion coefficient is even less certain. Biological nitrogen fixation (BNF) supplies globally some 90 to 140 Tg N/yr to agricultural systems (Peoples et al., 1995). Although more verification of these figures is necessary, all indications are that BNF contributes more nitrogen for plant growth than the total amount of synthetic nitrogen fertilisers applied to crops each year (Danso, 1995). The *IPCC Guidelines* (1995) mention equal rates. On average, BNF supplies 50-60 per cent of the nitrogen harvested in grain legumes, 55-60 per cent of the nitrogen in nitrogen fixing trees and 70-80 per cent of the nitrogen accumulated by pasture legumes (Danso, 1995). Cultivation of grain legumes, however, often results in net soil nitrogen depletion.

In the tropics and subtropics, the use of Azolla (a genus of aquatic ferns which contains an N_2-fixing cyanobacterium) is widespread. Azolla fixes 20-25 kg N/ha (Kumarasinghe and Eskew, 1991) which is released upon decomposition. This nitrogen serves to fertilise an associated crop and eventually to stimulate N_2O formation.

Galbally et al. (1992) and Bouwman and Sombroek (1990) indicate that legumes may contribute to N_2O emission in a number of ways. Atmospheric N_2 fixed by legumes can be nitrified and denitrified in the same way as fertiliser N, thus providing a source of N_2O. Additionally, symbiotically living Rhizobia in root nodules are able to denitrify and produce N_2O (O'Hara and Daniel, 1985). Galbally et al. (1992) suggest an emission rate of 4 kg N/ha/yr from improved pastures, and Duxbury et al. (1982) suggest that legumes can increase N_2O emissions from pastures by a factor of 2 or 3. More recently Carran et al. (1995) found annual N_2O emissions ranging from 0.5 to 5 kg N_2O–N depending upon the relative fertility of the sampling location. In old and young rye grass/clover pastures Muller (personal communication) observed N_2O emissions of 0.7 and 0.3 kg N/ha/yr, respectively.

The absolute amount of nitrogen fixed by a crop is also very uncertain (Peoples et al., 1995.). Because of this uncertainty and the paucity of country data on N-fixing crops, it is difficult to assign a conversion factor for N_2O emissions derived from nitrogen fixation. Total nitrogen input (F_{BN}, Equation 1) is estimated by assuming that total crop biomass is about twice the mass of edible crop (FAO, 1990b), and a certain nitrogen content of nitrogen fixing crop ($Frac_{NCRBF}$, Table 4-19). A residue/crop ratio of 1 is assumed. For specific crops, ratios can be obtained from Table 4-17. This crop production is defined in FAO crop data bases as pulses and soybeans. The N-fixation contribution does not include N_2O produced in legume pastures. This N_2O production is at least partially accounted for in emissions from pastures that are being grazed (Section 4.5.3). Australia and New Zealand, for example, contain large areas of pasture land that includes legumes as part of the pastoral system. Little data are available for other parts of the globe (Mosier et al., 1995b).

Crop residue and Sewage sludge application

There is only limited information concerning re-utilisation of nitrogen from crop residues and nitrogen from sewage sludge applied to agricultural lands. Although the amount of nitrogen that recycles into agricultural fields through these mechanisms may add 25-100 Tg of N/yr of additional nitrogen into agricultural soils (mainly from crop residues) the amount converted to N_2O is not known. To account for the N_2O in the inventory budget the emission factor for fertilisers is used as default and the amount of nitrogen re-entering cropped fields through crop residues is calculated from the FAO crop production data. Because no appropriate estimates of sewage sludge nitrogen used as fertiliser were found, this nitrogen input is not discussed further (FAO, 1978-1981).

Nitrous oxide emissions associated with crop residue decomposition are calculated here by estimating the amount of nitrogen entering soils as crop residue (F_{CR}). The amount of nitrogen entering the crop residue pool is calculated from crop production data. Since FAO data that can be used for this purpose only represent the edible portion of the crop, these must be roughly doubled to estimate total crop biomass. We then can assume a nitrogen percentage ($Frac_{NCRBF}$ and $Frac_{NCR0}$; Table 4-19) to convert from kg dry biomass/yr to kg N/yr in crops. Some countries may have sufficient information to define the nitrogen content of crop biomass more precisely. As a default we suggest distinguishing between N-fixing crops (pulses and soybeans) and non-N-fixing crops.

Some of the crop residues is removed from the field as crop (approximately 45 per cent), and some may be burned (approximately 25 per cent of the remaining residue in developing countries), or fed to animals.

Glasshouse farming

N-fertiliser application to glasshouse-grown crops are typically high and nitrogen not recovered in the crops is frequently low (Postma et al, 1994). The available data are limited in scope, but three sets of studies indicate that N_2O emissions from glasshouse crops may be relatively small. Postma et al. (1994) quantified NH_3 and N_2O emissions from glasshouse cultivation of lettuce on a sandbed and found that NH_3 emissions and N_2O emitted directly or in drainage water totalled less than 0.01 kg N/kg of nitrogen applied. Daum (personal communication, 1995) measured N_2O emissions from soil-less culture cucumbers and found that N-loss rates as N_2O ranged between 0.004 and 0.009 kg N/kg of nitrogen input into the culture system. Pollaris (1994) measured N_2O emission in a glasshouse cultivation of tomato and lettuce and found respectively 0.007 and 0.014 kg of each kg of applied nitrogen emitted as N_2O. The importance of another factor, N_2O emission during steam disinfection of glasshouse soils, is uncertain. Postma (personal communication, 1995) found that 2-25 kg N_2O-N/ha were lost during 10-hours following soil steaming. The extent of glasshouses to which this practice is applied is not known. Overall, these data suggest that N_2O emissions from glasshouse agriculture do not need to be included separately in N_2O emission inventories but should be included only in the total fertiliser nitrogen consumed within each country. Emissions from glasshouse agriculture is, therefore, not discussed further.

Cultivation of high organic content soils

Large N_2O emissions occur as a result of cultivation of organic soils (Histosols) due to enhanced mineralisation of old, N-rich organic matter (Guthrie and Duxbury, 1978; Klemedtsson, personal communication, 1995). The rate of N-mineralisation is determined by the N-quality of the Histosol, management practices and climatic conditions. The range for enhanced emissions of N_2O due to cultivation is estimated to be 2-15 kg N_2O-N/ha/yr of cultivated Histosol. Default emission values of 5 and 10 kg N_2O-N/ha/yr are suggested for temperate and boreal, and tropical regions, respectively (Table 4-18). If better country values are available they should be used.

Soil sink for N_2O

Aerobic soils are typically sources for N_2O, but small uptake rates have been observed in isolated instances in dry soils (Duxbury and Mosier, 1993) and in wet grass pastures (Ryden, 1981; 1983). In a seasonally burned “cerrado’ in Brazil, Nobre (1994) observed occasional small but inconsistent consumption rates and concluded that this sink was very small in these soils. Anaerobic soils have a large potential for reducing N_2O to N_2 (Erich et al., 1984), since the major product of denitrification in soils is usually N_2 rather than N_2O. However, no large, constant N_2O uptake has been reported, and flooded rice fields (Parashar, 1991), for example, generally show very small emissions, depending upon the time of cropping season (Minami and Fukushi, 1984). Apparently slow rates of dissolution and transport of atmospheric N_2O in wet or flooded soils prevent this process from being a significant regulator of atmospheric N_2O. Until additional information is available to indicate that soil uptake, in aerobic or flooded soils, is important, soil uptake of atmospheric N_2O will not be included in the N_2O budget for agricultural systems.

METHODOLOGY FOR ESTIMATING DIRECT N_2O EMISSIONS FROM AGRICULTURAL FIELDS.

The *Revised IPCC 1996 Methodology* for assessing direct N_2O emissions from agricultural fields includes consideration of synthetic fertiliser (F_{SN}), nitrogen from animal waste (F_{AW}), enhanced N_2O production due to biological N-fixation (F_{BN}), nitrogen from crop residue mineralisation (F_{CR}) and soil nitrogen mineralisation due to cultivation of Histosols (F_{OS}). For futher information see: Bouwman (1994, 1995); Mosier (1994); Mosier et al. (1995a;b).

In this estimate, the total direct annual N_2O emission is:

EQUATION I

$$N_2O_{DIRECT} = [(F_{SN} + F_{AW} + F_{BN} + F_{CR}) \text{ x } EF_1] + F_{OS} \text{ x } EF_2$$

where:

N_2O_{DIRECT} = direct N_2O emissions from agricultural soils in country (kg N/yr);

EF_1 = emission factor for direct soil emissions (kg N_2O-N/kg N input) (see Table 4-18);

EF_2 = emission factor for organic soil mineralisation due to cultivation (kg N_2O-N ha/yr) (see Table 4-18);

F_{OS} = area of cultivated organic soils within country (ha of histosols in FAO data base);

F_{AW} = manure nitrogen used as fertiliser in country, corrected for NH_3 and NO_x emissions and excluding manure produced during grazing (kg N/yr);

F_{BN} = N fixed by N-fixing crops in country (kg N/yr);

F_{CR} = N in crop residues returned to soils in country (kg N/yr);

F_{SN} = synthetic nitrogen applied in country (kg N/yr);

F_{SN} = N_{FERT} x (1-$Frac_{GASF}$);

F_{AW} = (Nex x (1-($Frac_{FUEL}$ + $Frac_{GRAZ}$ + $Frac_{GASM}$));

F_{BN} = 2 x $Crop_{BF}$ x $Frac_{NCRBF}$;

F_{CR} = 2 x [$Crop_0$ x $Frac_{NCR0}$ + $Crop_{BF}$ x $Frac_{NCRBF}$] x (1-$Frac_R$) x (I-$Frac_{BURN}$);

and

N_{FERT} = synthetic fertiliser use in country (kg N/yr);

$Frac_{GASF}$ = 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);

N_{ex} = amount of nitrogen excreted by the livestock within a country (kg N/yr) (see Table 4-20);

$Frac_{FUEL}$ = fraction of livestock nitrogen excretion contained in excrements burned for fuel (kg N/kg N totally excreted)

$Frac_{GRAZ}$ = fraction of livestock nitrogen excreted and deposited onto soil during grazing (kg N/kg N excreted) country estimate;

$Frac_{GASM}$ = 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);

$Crop_{BF}$ = seed yield of pulses + soybeans in country (kg dry biomass/yr);

$Frac_{NCRBF}$ = fraction of nitrogen in N-fixing crop (kg N/kg of dry biomass) (see Table 4-19);

$Crop_0$ = production of all other (i.e., non-N fixing) crops in country (kg dry biomass/yr);

$Frac_{NCR0}$ = fraction of nitrogen in non-N-fixing crop (kg N/kg of dry biomass) (see Table 4-19);

$Frac_R$ = fraction of crop residue that is removed from the field as crop (kg N/kg crop-N) (see Table 4-19);

$Frac_{BURN}$ = fraction of crop residue that is burned rather than left on field (see Table 4-19).

The input data needed for this methodology include synthetic fertiliser use (N_{FERT}), manure-N used as fertiliser (F_{AW}), edible crop production of N-fixing crops ($Crop_{BF}$) and non-N-fixing crops ($Crop_0$), and area of cultivated organic soils (Histosols) in the country. The data for synthetic fertiliser use are available on a country basis in the FAO data base (e.g., FAO, 1990a) and the amount of nitrogen in animal waste applied to agricultural fields (F_{AW}) is calculated from the number and type of animals within a country (see Section 4.5.3) and an in-country estimate of the percentage of nitrogen excreted by farm animals that is reapplied to the field. Both synthetic fertiliser and manure used as fertiliser need to be corrected for the amount of NH_3 volatilised and NO_x emitted (10 and 20 per cent of nitrogen applied, respectively) after the material is placed in or on the soil so that the same nitrogen atom is not counted again in Section 4.5.4. The F_{AW} data also need to be carefully evaluated for each country to be sure that animal waste used to fertilise crops and animal waste deposited on pastures while animals are grazing (accounted for in Section 4.5.3) are not double counted. Crop production data for pulses and soybeans and non-N-fixing crops are listed in the FAO crop data base (FAO, 1990b). Residue remaining on the field may not always total 55 per cent of total crop biomass. If more appropriate biomass and N content numbers are available within a country they should be used. The area of histosol can be obtained from FAO Databases (e.g., FAO, 1991).

TABLE 4-19
DEFAULT VALUES FOR PARAMETERS

$Frac_{BURN}$	0.25 in developing and 0.10 or less in developed countries (kg N/kg crop-N);
$Frac_R$	0.45 kg N/kg crop-N;
$Frac_{FUEL}$	0.0 kg N/kg N excreted[a];
$Frac_{GASF}$	0.1 kg NH_3-N + NO_x-N/kg of synthetic fertiliser N applied;
$Frac_{GASM}$	0.2 kg NH_3-N + NO_x-N/kg of N excreted by livestock;
$Frac_{GRAZ}$	Table 4-21, Column Pasture range and Paddock [a];
$Frac_{NCRBF}$	0.03 kg N/kg of dry biomass;
$Frac_{NCR0}$	0.015 kg N/kg of dry biomass.

[a] Countries are recommended to obtain country specific data.

4.5.3 Direct soil emissions of N_2O from animal production (including stable emissions to be reported under Manure Management)

OVERVIEW OF SOURCES

Recent studies (e.g., Bouwman, 1995; Jarvis and Pain, 1994; Flessa et al., submitted; Mosier et al., 1995a;b) indicate that emissions from animal wastes can be significant. There are three potential sources in animal production, (a) animals themselves, (b) animal wastes during storage and treatment, and (c) dung and urine deposited on the soil by grazing animals. Although N_2O emissions from animals are described, they are not accounted for here as these are minor sources on a global scale and because such emissions are not a source of N_2O emissions from soil. The second source includes possible losses during spreading operations. Emissions from manure stored in animal waste management systems (i.e., before it is added to soils), however, must be reported under Manure Management (Section 4.2). The final source, from grazing animals, is included in the agricultural soils because the dung and urine are considered as fertilisers.

A) N_2O from animals

Animals themselves may be very small sources of N_2O. Animal fodders contain 10 to 40 g of nitrogen (N)/kg dry matter. The greater part of this nitrogen is organically bound, but as total nitrogen content increases so does the nitrate (NO^-_3) content, generally. Nitrate contents in fodders generally range from 1-10 g/kg dry matter (Spoelstra, 1985). Upon passage through the digestive track of the animal, nitrate is reduced via dissimilatory nitrate reduction to NH_3/NH_4^+. The nitrate reduction reaction may release small amounts of N_2O in the gut (Kaspar and Tiedje, 1981), which may escape to the atmosphere during rumination. Though this possible route of N_2O formation has been known for over 10 years, quantitative data in terms of N_2O release are still lacking to-date. The total amount of N_2O released by cattle is probably very small, because the gut is highly anoxic and this will favour the formation of NH_3/NH_4^+ (Tiedje, 1988). Direct losses from animals themselves are likely to be very small and are therefore not discussed further.

B) N_2O emissions from animal waste management systems

The proportion of total nitrogen intake that is excreted and partitioned between urine and faeces is dependent on the type of animal, the intake of dry matter, and the nitrogen concentration of the diet (Whitehead, 1970). The retention of nitrogen in animal products, i.e., milk, meat, wool and eggs, ranges from about 5 to 20 per cent of the total nitrogen intake, generally. The remainder is excreted via dung and urine. For sheep and cattle, faecal excretion is usually about 8g/N/kg dry matter consumed, regardless of the nitrogen content of the feed (Barrow and Lambourne, 1962). The remainder of the nitrogen is excreted in the urine and as the nitrogen content of the diet increases, so does the proportion of nitrogen in the urine. In animal production systems, where animal intake of nitrogen is high, more than half of the nitrogen is excreted as urine. For instance, sheep grazing grass/clover pastures in New Zealand have 70-75 per cent of excreted nitrogen in the urine (Haynes and Williams, 1993), whereas dairy cows in the Netherlands excrete 60-65 per cent of nitrogen in urine (Van Vuuren and Meijs, 1987).

There are small amounts of mineral nitrogen in faeces but the bulk of the nitrogen is in organic form. About 20-25 per cent of faecal nitrogen is water soluble, 15-25 per cent is undigested dietary nitrogen and the remaining 50-65 per cent is present in bacterial cells (Haynes and Williams, 1993). The organically bound nitrogen in faeces must be mineralised to NH_3/NH_4^+, before it can be the substrate of nitrifiers and denitrifiers and hence, a source of N_2O. Mineralisation of the water soluble organic nitrogen compounds and parts of the organic nitrogen compounds from bacterial cells is rapid, generally, leading to an increase in the amount of NH_3/NH_4^+ in animal wastes during storage.

The concentration of nitrogen in urine varies widely because of factors such as nitrogen content in the diet and consumption of water. Typically over 70 per cent of the nitrogen in urine is present as urea and the rest consists of amino acids and peptides. Excretions of poultry contains uric acid as dominant nitrogen compound. The hydrolysis of both urea and uric acid in urine patches to NH_3/NH_4^+ is very rapid, both in the pasture and in animal housings. Fresh urine and dung contain no nitrate; all nitrogen in wastes is in reduced forms.

Production of N_2O during storage and treatment of animal wastes can occur via combined nitrification-denitrification of ammoniacal nitrogen contained in the wastes. The amount released depends on the system and duration of waste management. As fresh dung and slurry is highly anoxic and well-buffered with near neutral pH, one would expect N_2O production to increase with increasing aeration. Aeration initiates the nitrification-denitrification reactions, and hence makes release of N_2O possible. Unfortunately, there is not enough quantitative data to derive a relationship between the degree of aeration and N_2O emission from slurry during storage and treatment.

Losses of N_2O from cattle slurry during storing for up to 6 months, with two minutes of gentle mixing twice a week, were below detection limit (10^{-4} kg N_2O N/kg nitrogen in slurry during 6 months (Oenema and Velthof, 1993). N_2O losses from a natural vented cubicle housing with 60 dairy cows and a concrete floor with frequent removal of wastes with a scarper system were also below the detection limit (2. 10^{-4} kg N per kg nitrogen excreted) (Velthof et al., submitted a;b).

Heinemeyer (unpublished results) measured N_2O losses from different housing systems for pigs. He obtained the following preliminary, results from the various housing systems: 6 10^{-4} -11 10^{-4} kg N_2O-N/kg nitrogen excreted for full and half slit floor, and reduced and deep litter; 7.4 10^{-3} kg N_2O-N/kg nitrogen excreted for deep litter combined with compost; and 3.3 10^{-3} kg N_2O-N/kg nitrogen excreted for manure piles. The slightly larger losses from the deep litter/compost and manure pile could be due to the effects of aeration. Groenestein et al. (1993) reported that more than 0.15 kg N/kg nitrogen in pig 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

$$N_2O \text{ emissions from groundwater and agricultural drainage water} = N_{LEACH} \times EF_5\text{-g}$$

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

$N_2O_{(L)}$ = N_{LEACH} x 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

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

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

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)

Appendix B

Data Underlying Nitrous Oxide Emissions from Agricultural Soils

This appendix presents the data used to calculate the manure-N excretion and N_2O emission factors in Table B-1. N_2O emissions from different Animal Waste Management Systems in different regions of the world are in Table B-2.

TABLE B-1

CALCULATION OF MANURE-N EXCRETION AND N_2O EMISSION FACTORS FOR DIFFERENT ANIMAL WASTE MANAGEMENT SYSTEMS IN DIFFERENT WORLD REGIONS. THESE ARE TO BE REPORTED UNDER MANURE MANAGEMENT, EXCEPT FOR DAILY SPREAD AND PASTURE RANGE OF PADDOCK (EMISSIONS FROM AGRICULTURAL SOILS) AND EMISSIONS AFTER USE AS A FUEL (ENERGY)

Region	Type of Animal			Emission Factor for AWMSs EF_3 (% of Manure N Excreted that is lost as N_2O)							
		Number of animals ($x10^6$)	Nitrogen excretion kg (N/animal/yr)	Anaerobic Lagoon (EF_3)	Liquid Systems (EF_3)	Daily Spread (EF_3)	Solid Storage & Drylot (EF_3)	Pasture range Paddock (EF_3)	Used Fuel (EF_3)	Other System (EF_3)	Total N Excreted (Tg N)
North America	Non-dairy Cattle	99.199	70	0.1	0.1	0.0	2.0	2.0	0.0	0.5	6.9
	Dairy Cattle	16.521	100	0.1	0.1	0.0	2.0	2.0	0.0	0.5	1.7
	Poultry (E)	1486.266	0.6	0.1	0.1	0.0	2.0	2.0	0.0	0.5	0.9
	Sheep	11.336	16	0.1	0.1	0.0	2.0	2.0	0.0	0.5	0.2
	Swine	66.146	20	0.1	0.1	0.0	2.0	2.0	0.0	0.5	1.3
	Other animals (F)	6.067	25	0.1	0.1	0.0	2.0	2.0	0.0	0.5	0.2
Western Europe	Non-dairy Cattle	56.618	70	0.1	0.1	0.0	2.0	2.0	0.0	0.5	4.0
	Dairy Cattle	31.099	100	0.1	0.1	0.0	2.0	2.0	0.0	0.5	3.1
	Poultry (E)	880.000	0.6	0.1	0.1	0.0	2.0	2.0	0.0	0.5	0.5
	Sheep	93.856	20	0.1	0.1	0.0	2.0	2.0	0.0	0.5	1.9
	Swine	114.959	20	0.1	0.1	0.0	2.0	2.0	0.0	0.5	2.3
	Other animals (F)	31.578	25	0.1	0.1	0.0	2.0	2.0	0.0	0.5	0.8
Eastern Europe	Non-dairy Cattle	101.447	50	0.1	0.1	0.0	2.0	2.0	0.0	0.5	5.1
	Dairy Cattle	56.800	70	0.1	0.1	0.0	2.0	2.0	0.0	0.5	4.0
	Poultry (E)	1667.000	0.6	0.1	0.1	0.0	2.0	2.0	0.0	0.5	1.0
	Sheep	188.159	16	0.1	0.1	0.0	2.0	2.0	0.0	0.5	3.0
	Swine	152.757	20	0.1	0.1	0.0	2.0	2.0	0.0	0.5	3.1
	Other animals (F)	21.558	25	0.1	0.1	0.0	2.0	2.0	0.0	0.5	0.5

TABLE B-1 (CONTINUED)
CALCULATION OF MANURE-N EXCRETION AND N2O EMISSION FACTORS FOR DIFFERENT ANIMAL WASTE MANAGEMENT SYSTEMS IN DIFFERENT WORLD REGIONS. THESE ARE TO BE REPORTED UNDER MANURE MANAGEMENT, EXCEPT FOR DAILY SPREAD AND PASTURE RANGE OF PADDOCK (EMISSIONS FROM AGRICULTURAL SOILS) AND EMISSIONS AFTER USE AS A FUEL (ENERGY)

Region	Type of Animal			Emission Factor for AWMSs EF_3 (% of Manure N Excreted that is lost as N_2O)							
		Number of animals (x 10^6)	Nitrogen excretion kg (N/animal/yr)	Anaerobic Lagoon (EF_3)	Liquid Systems (EF_3)	Daily Spread (EF_3)	Solid Storage & Drylot (EF_3)	Pasture range Paddock (EF_3)	Used Fuel (EF_3)	Other System (EF_3)	Total N Excreted. (Tg N)
Oceania	Non-dairy Cattle	27.610	60	0.1	0.1	0.0	2.0	2.0	0.0	0.5	1.7
	Dairy Cattle	4.441	80	0.1	0.1	0.0	2.0	2.0	0.0	0.5	0.4
	Poultry (E)	71.000	0.6	0.1	0.1	0.0	2.0	2.0	0.0	0.5	0.0
	Sheep	228.982	20	0.1	0.1	0.0	2.0	2.0	0.0	0.5	4.6
	Swine	5.003	16	0.1	0.1	0.0	2.0	2.0	0.0	0.5	0.1
	Other animals (F)	2.579	25	0.1	0.1	0.0	2.0	2.0	0.0	0.5	0.1
Latin America	Non-dairy Cattle	272.871	40	0.1	0.1	0.0	2.0	2.0	0.0	0.5	10.9
	Dairy Cattle	37.560	70	0.1	0.1	0.0	2.0	2.0	0.0	0.5	2.6
	Poultry (E)	1259.000	0.6	0.1	0.1	0.0	2.0	2.0	0.0	0.5	0.8
	Sheep	117.312	12	0.1	0.1	0.0	2.0	2.0	0.0	0.5	1.4
	Swine	78.150	16	0.1	0.1	0.0	2.0	2.0	0.0	0.5	1.3
	Other animals (F)	71.699	40	0.1	0.1	0.0	2.0	2.0	0.0	0.5	2.9
Africa	Non-dairy Cattle	133.198	40	0.1	0.1	0.0	2.0	2.0	0.0	0.5	5.3
	Dairy Cattle	18.734	60	0.1	0.1	0.0	2.0	2.0	0.0	0.5	1.1
	Poultry (E)	646.000	0.6	0.1	0.1	0.0	2.0	2.0	0.0	0.5	0.4
	Sheep	179.171	12	0.1	0.1	0.0	2.0	2.0	0.0	0.5	2.2
	Swine	12.445	16	0.1	0.1	0.0	2.0	2.0	0.0	0.5	0.2
	Other animals (F)	162.194	40	0.1	0.1	0.0	2.0	2.0	0.0	0.5	6.5

TABLE B-1 (CONTINUED)
CALCULATION OF MANURE-N EXCRETION AND N2O EMISSION FACTORS FOR DIFFERENT ANIMAL WASTE MANAGEMENT SYSTEMS IN DIFFERENT WORLD REGIONS. THESE ARE TO BE REPORTED UNDER MANURE MANAGEMENT, EXCEPT FOR DAILY SPREAD AND PASTURE RANGE OF PADDOCK (EMISSIONS FROM AGRICULTURAL SOILS) AND EMISSIONS AFTER USE AS A FUEL (ENERGY)

Region	Type of Animal			Emission Factor for AWMSs EF_3 (% of Manure N Excreted that is lost as N_2O)							
		Number of animals (x 10^6)	Nitrogen excretion kg N/animal/yr	Anaerobic Lagoon (EF_3)	Liquid Systems (EF_3)	Daily Spread (EF_3)	Solid Storage & Drylot (EF_3)	Pasture range Paddock (EF_3)	Used Fuel (EF_3)	Other System (EF_3)	Total N Excreted (Tg N)
Near East and Mediterranean	Non-dairy Cattle	44.562	50	0.1	0.1	0.0	2.0	2.0	0.0	0.5	2.2
	Dairy Cattle	17.174	70	0.1	0.1	0.0	2.0	2.0	0.0	0.5	1.2
	Poultry (E)	656.000	0.6	0.1	0.1	0.0	2.0	2.0	0.0	0.5	0.4
	Sheep	187.502	12	0.1	0.1	0.0	2.0	2.0	0.0	0.5	2.3
	Swine	0.174	16	0.1	0.1	0.0	2.0	2.0	0.0	0.5	0.0
	Other animals (F)	81.962	40	0.1	0.1	0.0	2.0	2.0	0.0	0.5	3.3
Asia and Far East	Non-dairy Cattle	440.398	40	0.1	0.1	0.0	2.0	2.0	0.0	0.5	17.6
	Dairy Cattle	45.240	60	0.1	0.1	0.0	2.0	2.0	0.0	0.5	2.7
	Poultry (E)	3949.000	0.6	0.1	0.1	0.0	2.0	2.0	0.0	0.5	2.4
	Sheep	202.442	12	0.1	0.1	0.0	2.0	2.0	0.0	0.5	2.4
	Swine	403.231	16	0.1	0.1	0.0	2.0	2.0	0.0	0.5	6.5
	Other animals (F)	293.700	40	0.1	0.1	0.0	2.0	2.0	0.0	0.5	11.7
World Total											135.3

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

TABLE B-2
N_2O EMISSIONS FROM DIFFERENT ANIMAL WASTE MANAGEMENT SYSTEMS IN DIFFERENT WORLD REGIONS

Region	Type of Animal	N_2O Emission from Waste Management Systems (GgN)							
		Liquid System	Daily Spread	Solid Storage and Drylot	Pasture Range and Paddock	Used Fuel	Other System	Total AWMS	Total N_2O Emission (Gg N)
North America	Non-dairy Cattle	0	0	19	117	0	0	137	137
	Dairy Cattle	0	0	0	0	0	1	9	9
	Poultry (E)	0	0	0	0	0	4	4	4
	Sheep	0	0	0	3	0	0	3	3
	Swine	0	0	5	0	0	0	6	6
	Other animals (F)	0	0	0	3	0	0	3	3
Western Europe	Non-dairy Cattle	0	0	2	26	0	2	32	32
	Dairy Cattle	0	0	13	5	0	0	20	20
	Poultry (E)	0	0	0	0	0	2	3	3
	Sheep	0	0	1	33	0	1	34	34
	Swine	0	0	11	0	0	0	12	12
	Other animals (F)	0	0	0	15	0	0	15	15
Eastern Europe	Non-dairy Cattle	0	0	53	0	0	0	55	55
	Dairy Cattle	0	0	53	10	0	0	64	64
	Poultry (E)	0	0	0	0	0	4	4	4
	Sheep	0	0	0	44	0	4	48	48
	Swine	0	0	0	16	0	7	24	24
	Other animals (F)	0	0	0	10	0	0	10	10

TABLE B-2 (CONTINUED)
N_2O EMISSIONS FROM DIFFERENT ANIMAL WASTE MANAGEMENT SYSTEMS IN DIFFERENT WORLD REGIONS

Region	Type of Animal	N_2O Emission from Waste Management Systems (GgN)							
		Liquid System	Daily Spread	Solid Storage and Drylot	Pasture Range and Paddock	Used Fuel	Other System	Total AWMS	Total N_2O Emission (GgN)
Oceania	Non-dairy Cattle	0	0	0	33	0	0	33	33
	Dairy Cattle	0	0	0	7	0	0	7	7
	Poultry (E)	0	0	0	0	0	0	0	0
	Sheep	0	0	0	92	0	0	92	92
	Swine	0	0	0	0	0	0	0	0
	Other animals (F)	0	0	0	1	0	0	1	1
Latin America	Non-dairy Cattle	0	0	0	216	0	1	217	217
	Dairy Cattle	0	0	1	19	0	0	19	19
	Poultry (E)	0	0	0	6	0	2	8	8
	Sheep	0	0	0	28	0	0	28	28
	Swine	0	0	13	0	0	3	15	15
	Other animals (F)	0	0	0	57	0	0	57	57
Africa	Non-dairy Cattle	0	0	3	102	0	0	105	105
	Dairy Cattle	0	0	0	19	0	0	19	19
	Poultry (E)	0	0	0	6	0	0	7	7
	Sheep	0	0	0	43	0	0	43	43
	Swine	0	0	4	0	0	0	4	4
	Other animals (F)	0	0	0	128	0	0	129	129

TABLE B-2 (CONTINUED)
N_2O EMISSIONS FROM DIFFERENT ANIMAL WASTE MANAGEMENT SYSTEMS IN DIFFERENT WORLD REGIONS

Region	Type of Animal	N_2O Emission from Waste Management Systems (Gg N)								
		Anaerobic Lagoon	Liquid System	Daily Spread	Solid Storage and Drylot	Pasture Range and Paddock	Used Fuel	Other System	Total AWMS	Total N_2O Emission (Gg N)
Near East and Mediterranean	Non-dairy Cattle	0	0	0	0	34	0	0	35	35
	Dairy Cattle	0	0	0	1	19	0	0	19	19
	Poultry (E)	0	0	0	0	6	0	1	6	6
	Sheep	0	0	0	0	45	0	0	45	45
	Swine	0	0	0	0	0	0	0	0	0
	Other animals (F)	0	0	0	0	66	0	0	66	66
Asia and Far East	Non-dairy Cattle	0	0	0	49	102	0	0	151	151
	Dairy Cattle	0	0	0	0	13	0	0	13	13
	Poultry (E)	0	0	0	0	21	0	6	27	27
	Sheep	0	0	0	0	40	0	2	42	42
	Swine	0	0	0	68	0	0	0	71	71
	Other animals (F)	0	0	0	0	223	0	3	226	226
World Total						1609				1971

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

4.6 References

Methane and Nitrous oxide Emissions from Domestic Livestock Enteric Fermentation and Manure Management

AAC (Australian Agricultural Council) (1990), *Feed Standards for Australian Livestock. Ruminants*, Commonwealth Scientific and Industrial Research Organisation (CSIRO) Publications, East Melbourne, Victoria, Australia.

ARC (1980), *The Nutrient Requirements of Ruminant Livestock*, Commonwealth Agricultural Bureau, the Lavenham Press Ltd., UK.

Bamualim, A. and Kartiarso (1985), "Nutrition of draught animals with special reference to Indonesia." In: *Draught Animal Power for Production* J.W. Copland (ed.), Australian Centre for International Agricultural Research (ACIAR), Proceedings Series No. 10. ACIAR, Canberra, A.C.T., Australia.

Bhattacharya, S.C., H.L. Pham, R.M. Shrestha and Q.V. Vu, (1993) "CO_2 emission due to fossil/traditional fuels, residues and wastes in Asia." *AIT (Asian Institute of Technology) Workshop on Global Warming Issues in Asia.* Bangkok, Thailand.

Blaxter, K.L. and J.L. Clapperton (1965), "Prediction of the amount of methane produced by ruminants," *British Journal of Nutrition* 19:511-522.

Crutzen, P.J., I. Aselmann and W. Seiler (1986), "Methane Production by Domestic Animals, Wild Ruminants, Other Herbivorous Fauna, and Humans," *Tellus* 38B:271-284

FAO (1990), *Yearbook - Production Vol. 44*, FAO, United Nations, Rome, Italy.

Gibbs, M.J. and D.E. Johnson (1993), "Livestock Emissions." In: *International Methane Emissions*, US Environmental Protection Agency, Climate Change Division, Washington, D.C., U.S.A.

Graham, N.M. (1985), "Relevance of the British metabolisable energy system to the feeding of draught animals." In: *Draught Animal Power for Production*, J.W. Copland (ed.), ACIAR (Australian Centre for International Agricultural Research) Proceedings Series No. 10. ACIAR, Canberra, A.C.T., Australia.

Hashimoto, A. and J. Steed (1993), *Methane Emissions from Typical U.S. Livestock Manure Management Systems.* Draft report prepared for ICF Incorporated under contract to the Global Change Division of the Office of Air and Radiation, US Environmental Protection Agency, Washington, D.C., U.S.A.

IAEA (1992), *Manual on measurement of methane and nitrous oxide emissions from agriculture*. International Atomic Energy Agency Publication IAEA-TECDOC-674, Vienna, Austria.

Ibrahim, M.N.M. (1985), "Nutritional status of draught animals in Sri Lanka." In: *Draught Animal Power for Production*, J.W. Copland (ed.), ACIAR (Australian Centre for International Agricultural Research) Proceedings Series No. 10. ACIAR, Canberra, A.C.T., Australia.

Johnson, K.A., H. Westberg, M. Hyler and B. Lamb (1993), *Cattle Methane Measurement Techniques Workshop*, August 9-12, 1993, Washington State University, Pullman, WA, sponsored by the Global Change Division, Office of Air and Radiation, US Environmental Protection Agency, Washington, D.C., U.S.A.

Jossy, M. T. (1994), *Estimation of Greenhouse Gases Emission from Agricultural Wastes in Asia,* M.Eng Thesis, Asian Institute of Technology, Bangkok, Thailand (to be submitted).

Jurgens, M.H. (1988), *Animal Feeding and Nutrition*, Sixth Edition, Kendall/Hunt Publishing Company, Dubuque, Iowa, U.S.A.

Lawrence, P.R. (1985), "A review of nutrient requirements of draught oxen." In: *Draught Animal Power for Production*, J.W. Copland (ed.), ACIAR (Australian Centre for International Agricultural Research) Proceedings Series No. 10. ACIAR, Canberra, A.C.T., Australia.

Lichtman, R. J. (1983), *Biogas Systems in India*, Volunteers In Technical Assistance (VITA), Arlington, Virginia, U.S.A.

NRC (1984), *Nutrient Requirements of Beef Cattle*, National Academy Press, Washington, D.C., U.S.A.

NRC (1989), *Nutrient Requirements of Dairy Cattle*, National Academy Press, Washington, D.C., U.S.A.

OECD (1991), *Estimation of Greenhouse Gas Emissions and Sinks: Final Report from the OECD Experts Meeting, 18-21 February 1991*, OECD, Paris, France.

Robbins, C.T. and D.L. Robbins (1979), "Foetal and neonatal growth patterns and maternal reproductive effort in ungulates and sub-ungulates," *Amer. Naturalist* 114:101.

Safley, L.M. Jr. and P.W. Westerman, (1992), "Performance of a low temperature lagoon digester," *Bioresource Technology* 41:167-175.

Safley, L.M., M.E. Casada, J.W. Woodbury, and K.F. Roos (1992), *Global Methane Emissions from Livestock and Poultry Manure.* US Environmental Protection Agency, Global Change Division, Washington, D.C., February 1992, EPA/400/1091/048.

Stuckey, D.C. (1984), "Biogas: a global perspective." In: *Biogas Technology, Transfer and Diffusion*, M.M. El-Halwagi (ed.), Elsevier, New York, U.S.A., pp. 18-44.

US EPA (1993), *Anthropogenic Methane Emissions in the United States*, Global Change Division, Office of Air and Radiation, Washington, D.C.

Winrock (1978), *The Role of Ruminants in Support of Man*, Winrock International, Morrilton, Arkansas, U.S.A.

Woodbury, J.W. and A. Hashimoto (1993), "Methane Emissions from Livestock Manure." In *International Methane Emissions*, US Environmental Protection Agency, Climate Change Division, Washington, D.C., U.S.A.

Yancun, C., H. Cong and Liang Pusen (1985), "Development of a new energy village - Xinbu, China." In: *Integrated Rural Energy Planning*, Y. El Mahgary and A.K. Biswas (eds.), Butterworths Publishing, Guildford, England, pp. 99-108.

Methane Emissions from Flooded Rice Fields

Aselmann, I. and P.J. Crutzen (1989), “The global distribution of natural freshwater wetlands and rice paddies, their net primary productivity, seasonality and possible methane emission.” *J. Atm. Chem.* 8, 307-358.

Blake, D.R. and F.S. Rowland (1988), “Continuing world-wide increase in tropospheric methane, 1978 to 1987”. *Science* 239, 1129-1131.

Bouwman, A.F. (1990), "Exchange of GHGs between terrestrial ecosystems and the atmosphere." Ed. A.F. Bouwman, in *Soil and the Greenhouse Effect,* John Wiley and Sons, U.K., pp. 62-127.

Chen, Z., D. Li, K. Shao and B. Wang (1993), "Features of CH_4 emission from rice paddy fields in Beijing and Nanjing." *Chemosphere* 26, 239-245

Cai, Z.C., H. Zu, H.H. Zhang and J.S. Jin (1994), "Estimate of methane emission from rice paddy fields in Taihu region, China." *Pedosphere* 4, 297-306

China Agricultural Yearbook (1990), Agribookstore, Hampton, VA, U.S.A.

Cicerone, R.J., and J.D. Shetter (1981), "Sources of atmospheric methane; measurements in rice paddies and a discussion." *J Geophys Res* 86: 7203-7209.

Cicerone, R.J., J.D. Shetter and C.C. Delwiche (1983), "Seasonal variation of methane flux from a California rice paddy." *J. Geophys. Res.* 88, 7203-7209.

Cicerone, R.J. and R.S. Oremland (1988). "Biogeochemical aspects of atmospheric methane." *Global Biogeochem. Cycles* 2, 299-327.

Cicerone, R.J., C.C. Delwiche, S C. Tyler and P.R. Zimmerman (1992, "Methane emissions from California rice paddies with varied treatments." *Global Biogeochem. Cycles* 6 233-248.

Conrad, R. (1989), "Control of methane production in terrestrial ecosystems". In: *Exchange of Trace Gases between Terrestrial Ecosystems and the Atmosphere*, M.O. Andreae and D.S. Schimel(eds.), 39-58.

Conrad, R. (1993), "Mechanisms controlling methane emission from wetland rice fields". In: *The Biogeochemistry of Global Change: Radiative Trace Gases*, R.S. Oremland, (ed.) Chapman & Hall, New York, pp. 317-335.

Crutzen, P.J. (1985), "The role of the tropics in atmospheric chemistry." In: R Dickinson (ed.) *The Geophysiology of Amazonia*, Wiley, Chichester, pp. 107-132.

DeDatta, S.K. (1975), "Upland rice around the world." In: *Major Research in Upland Rice,* International Rice Research Institute, Los Baños, Philippines.

Denier van der Gon, H.A.C. and H.U. Neue (1994), "Impact of gypsum application on methane emission from a wetland rice field." *Global Biogeoche*m, Cycles, 8, 127-134

Ehhalt, D.H. and U. Schmidt (1978), "Sources and sinks of atmospheric methane." *Pure Appl. Geophys.*, 116, 452-464.

Grist, D.H. (1986), *Rice*. Longman, Inc., New York, U.S.A., 6th edition.

Holzapfel-Pschorn, A., and W. Seiler (1986), "Methane emission during a cultivation period from an Italian rice paddy." *J. Geophys. Res.*, 91, 11803-11814..

Husin, Y. A., D. Murdiyarso, M. A. K. Khalil, R. A. Rasmussen, M. J. Shearer, S. Sabiham, A. Sunar, A. Adijuwana, (*1995*), "Methane flux from Indonesian wetland rice : The effect of water management and rice variety", *Chemosphere*, 31(4), pp 3153-3180.

Huke, R.F. (1982), *Rice Area by Type of Culture; South, South-east and East Asia.* International Rice Research Inst., IRRI, Los Banos, Laguna, Philippines.

IPCC (1990), *Climate Change, the IPCC Scientific Assessment*, J.T. Houghton, G J Jenkins and J J Ephraums, (eds.) Cambridge University Press, UK, pp. 1-40.

IPCC (1996), *Climate Change 1995: the Science of Climate Change, WGI contribution to the IPCC Second Assessment Report*, J.T. Houghton, L G Meira Filho, B A Callander, N Harris, A Kattenberg and K Maskell. (eds.) Cambridge University Press, 1996.

Jermsawatdipong, P., J. Murase, P. Prabuddham, Y. Hasathon, N. Khomthong, K. Naklang, A. Watanabe, H. Haraguchi and M. Kimura (1994), "Methane emission from plots with differences in fertilizer application in Thai paddy fields." *Soil Sci. Plant Nutr.,* 40, 63-71.

IGAC (1994), Global Measurement Standardisation of Methane Emissions from Irrigated Rice Cultivation, A Report of the Rice Cultivation and Trace Gas Exchange Activity (RICE) of the International Global Atmospheric Chemistry (IGAC) Project (Convenors, R. Sass and H.U. Neue), IGAC Core Office, Cambridge, MA.

IRRI (1990), World Rice Statistics, 1989, International Rice Research Institute, Los Banos, Philippines.

IRRI (1993), 1993-1995 IRRI Rice Almanac, International Rice Research Institute, Manila, Philippines.

IRRI (1994), IRRI Rice Almanac, International Rice Research Institute, Manila, Philippines.

Katoh, K., P. Chairoj, W. Choltikul, H. Tsuruta and K. Yagi (1995). Mechanism of methane formation and development of methods to decrease methane emission in humid tropical paddy fields. Japan International Research Center for Agricultural Sciences.

Khalil, M.A.K. and R.A. Rasmussen (1983), "Sources, sinks, and seasonal cycles of atmospheric methane." *J Geophys Res* 88: 5131-5144.

Khalil, M.A.K., R.A. Rasmussen, M.X. Wang and L. Ren (1991), "Methane emission from rice fields in China." *Environ. Sci. Tech.,* 25:979-981.

Koyama, T. (1964), "Biogeochemical studies on lake sediments and paddy soils and the production of atmospheric methane and hydrogen." In: *Recent Researches in the Fields of Hydrosphere, Atmosphere and Nuclear Geochemistry,* Y. Miyake and T. Koyama, (eds.), Muruzen, Tokyo, Japan, pp. 143-177.

Kush, G. S., (1984), *Terminology for rice growing environments*, International Rice Research Institute, P.O Box 933, Manila, Philippines, pp 5-10.

Lindau, C.W., P.K. Bollich, R.D. DeLaune, W.H. Patrick Jr., and V.J. Law (1991), "Effect of urea fertilizer and environmental factors on CH_4 emissions from a Louisiana, USA rice field." *Plant Soil,* 136, 195-203

Lindau, C.W. and P.K. Bollich (1993), "Methane emissions from Louisiana first and ratoon rice crop." *Soil Science*, 156:42-48.

Lindau, C.W. (1994), "Methane emissions from Louisiana rice fields amended with nitrogen with fertilizers." *Soil Biol. Biochem.,* 26, 353-359

Minami, K. (1994), "Methane from rice production." *Fert. Res. 37*, pp. 169-179.

Minami, K., (1995*),* "The effect of nitrogen fertilizer use and other practices on methane emission from flooded rice." *Fert. Res.*, 40, pp. 71-84.

Mitra, A.P. (1992), *Greenhouse gas emission in India, 1991 methane campaign* Council of Scientific and Industrial Research, and Ministry of Environment and Forest. Scientific Report No. 20. NPL, India.

Mitra A. P. (1996);(ed.) *Greenhouse gas inventory for India : 1995 Update*, Global Change Series, 1996 (in press).

Neue, H. U., (1989), *Rice growing soils: constraints, utilization and research needs*, FFTC Book Series No. 39, Food and Fertilizer Technology Center for the Asia Pacific Region, Taipei, Taiwan.

Neue, H.U., P. Becker-Heidmann and H.W. Scharpenseel (1990), "Organic matter dynamics, soil properties and cultural practices in rice lands and their relationship to methane production." In: *Soils and the Greenhouse Effect* A.F. Bouwman (ed.), John Wiley and Sons, UK, pp. 457-466.

Neue, H.U. and R. Sass (1994), "Trace gas emissions from rice fields." In: *Global Atmospheric-Biospheric Chemistry*, R.G. Prinn (ed.) Plenum Press, New York.

Neue, H. U., R. S. Lantin, R. Wassmann, J. B. Aduna, M. C. R. Alberto, and M. J. F. Andales (1994), "Methane emission from rice soils of the Philippines." In: *CH_4 and N_2O Global Emissions and Controls from Rice Fields and Other Agricultural and Industrial Sources,* K. Minami, A. Mosier and R. L. Sass (eds.), NIAES, Series 2, Yokendo Publ., Tokyo, Japan, pp. 55-64.

NGGIC (1996), Australian Methodology for the Estimation of Greenhouse Gas Emissions and Sinks, Agriculture, Workbook for Non-Carbon Dioxide Gases from the Biosphere, Workbook 5.1, National Greenhouse Gas Inventory Committee 1996, Canberra, Australia.

Nugroho, S.G., J. Lumbanraja, H. Suprapto, Sunyoto, W.S. Ardjasa, H. Haraguchi and M. Kimura (1994a), "Methane emission from an Indonesian paddy field subjected to several fertilizer treatments." *Soil Sci. Plant Nutr.,* 40, 275-281

Nugroho, S.G., J. Lumbanraja, H. Suprapto, Sunyoto, W.S. Ardjasa, H. Haraguchi and M. Kimura (1994b), "Effect of intermittent irrigation on methane emission from an Indonesian paddy field" *Soil Sci. Plant Nutr.* 40, 609-615.

Parashar, D. C. et al., (1994), "Methane budget from paddy fields in India." *Current Science* 66, Report No. 12, pp 938-940.

Parashar, D. C, Gupta K. Prabhat and S. Bhataachaya (1996), "Methane emissions from paddy fields in India-An update," *J. Radio and Space Physics* (in press).

Patrick, W.H. Jr. (1981), "The role of inorganic redox systems in controlling reduction in paddy soils". In: *Proc. Symp. Paddy Soil,* 107-117, Science Press, Beijing, China, Springer-Verlag.

Sass, R.L., F.M. Fisher, P.A. Harcombe and F.T. Turner (1990), "Methane production and emission in a Texas rice field." *Global Biogeochemical Cycles* 4:47-68.

Sass, R.L., F.M. Fisher, P.A. Harcombe and F.T. Turner (1991a), "Mitigation of methane emissions from rice fields: Possible adverse effects of incorporated rice straw." *Global Biogeochemical Cycles* 5: 275-287.

Sass, R.L., F.M. Fisher, F.T. Turner and M.F. Jund (1991b), "Methane emission from rice fields as influenced by solar radiation, temperature and straw incorporation". *Global Biogeochemical Cycles* 5: 335-350.

Sass, R.L., F.M. Fisher, Y.B. Yang, F.T. Turner and M.F. Jund (1992), "Methane emissions from rice fields: the effect of flood water management". *Global Biogeochemical Cycles* 6(3): 249-262.

Sass, R L (1994), Short summary chapter for methane. In: CH_4 and N_2O: global emissions and controls from rice field and other agricultural and industrial sources, K. Minami, A. Mosier and R. L. Sass (eds.) NIAES, Series 2, Yokendo Publishers, Tokyo, pp 1-7.

Sass, R.L. and F.M. Fisher (1995), "Methane emissions from Texas rice fields: a five-year study." In: *Climate Change and Rice*. S. Peng, K.T. Ingram, H.U. Neue, L.H. Ziska (eds.) Springer, Berlin, pp.46-59.

Schütz, H., A. Holzapfel-Pschorn, R. Conrad, H. Rennenberg and W. Seiler (1989a), "A 3-year continuous record on the influence of daytime, season and fertilizer treatment on methane emission rates from an Italian rice paddy." *J. Geophys. Res*. 94, 16,405-16,416.

Schütz H., W. Seiler, and R. Conrad (1989b), "Processes involved in formation and emission of methane from rice paddies." *Biogeochem* 7: 33-53

Schütz, H., W. Seiler, and H. Rennenberg (1990), "Soil and land use related sources and sinks of methane in the context of the global methane budget." In: *Soils and the Greenhouse Effect,* A.F. Bouwman (ed), Wiley, Chichester, pp. 269-285.

Seiler, W., A. Holzapfel-Pschorn, R. Conrad and D. Scharffe (1984), "Methane emissions from rice paddies". *J. Atmos. Chem*. 1, 241-268.

Shao, (1993), Agro-Ecological Environment, Supplement Issue, 19-22 (in Chinese)

Shin, Y-K, S-H. Yun, M-E. Park (1995), "Estimation of methane emission by water management and rice straw application in paddy soil in Korea." *J. Korean Soc. of Soil Sci. and Fert*. 28 (3), 261-265.

Takai, Y. (1970), "The mechanism of methane fermentation in flooded paddy soil." *Soil Sci. Plant Nutr*. 16, 238-244.

United Nations (1992), *FAO Production Yearbook, vol. 46*. Food and Agriculture Organization of the United Nations, Rome, Italy.

Wang, M.X., A. Dai, R. Shen, H. Schütz, W. Seiler, H. Rennenberg and H. Wu (1990), "CH_4 emissions from a Chinese rice paddy field." *Acta Meteoroloogica Sinica. Scienta Atmospherica Sinica* 4, 265-274.

Wassman, R., H. Schütz, H. Papen, H. Rennenberg, W. Seiler, A. Dai, R. Shen, X. Shangguan and M. Wang (1993a), Quantification of methane emissions from Chinese rice fields (Zhejiang Province) as influenced by fertiliser treatment." *Biogeochemistry* 20:83-101.

Wassman, R., M.X. Wang, X.J. Shangguan, X.L. Xie, R. X. Shen, Y.S. Wang, H. Papen, H. Rennenberg and W. Seiler (1993b), First records of a field experiment on fertilizer effects on methane emission from rice fields in Hunan Province (PR China) *Geophys. Res. Letters* 20, 2071-2074.

Wassmann, R., H.U. Neue, R.S. Lantin, J.B. Aduna, M.C.R. Alberto, M.J. Andales, M.J. Tan, H.A.C. Denier van der Gon, H. Hoffmann, H. Papen, H. Rennenberg and W. Seiler (1994), "Temporal patterns of methane emissions from wetland rice fields treated by different modes of N application." *J. Geophys. Res. 99*, 16457-16462

Yagi, K. and K. Minami (1990a), Effect of organic matter applications on methane emissions from Japanese paddy fields. *Soil Sci. Plant Nutr*. 36, 599-610.

Yagi, K. and K. Minami (1990b), Estimation of global methane emission from paddy fields. Res. Rep. Div. Environ. Planning, *NIAES*, 6, 131-142 (in Japanese).

Yagi, K.H. Tsuruta, K. Minami, P. Chairoj and W. Cholitkul (1994a), "Methane emissions from Japanese and Thai paddy fields." In: *CH_4 and N_2O Global Emissions and Controls from Rice Fields and Other Agricultural and Industrial Sources*, K. Minami, A. Mosier and R.L. Sass (eds.), *NIAES*, Series 2, Yokendo Publ., Tokyo, Japan, pp 41 -53.

Yagi K, P.Chairoj, H. Tsuruta, W. Cholitkul, and K. Minami (1994b), "Methane emission from rice paddy fields in the central plain of Thailand." *Soil Sci. Plant Nutr.* 40: 29-37.

Yagi, K., H. Tsuruta and K. Minami (1996), "Effect of water management on methane emissions from a Japanese rice paddy field: Automated methane monitoring." *Global Biogeochem. Cycles* 10, 255-267.

Yamane, I. and K. Sato (1961), "Effect of temperature on the formation of gases and ammonium nitrogen in the water-logged soils." *Sci. Rep. Res. Inst. Tohoku Univ.* D (Agr.) 12, 31-46.

Yamane, I. and K. Sato (1964), "Decomposition of glucose and gas formation in flooded soil." *Soil Sci. Plant Nutr* 10, 127-133.

Yao, H., and Z. L. Chen (1994a), "Effects of chemical fertilizer on methane emission from rice paddies." *J. Geophys. Res.* 99, 16463-16470

Yao, H., and Z. L. Chen (1994b), "Seasonal variation of methane flux from a Chinese rice paddy in a semi-arid, temperate region." *J. Geophys. Res.* 99, 16471-16477

Agricultural Burning

Andreae, M.O. (1990), "Biomass burning in the tropics: Impact on environmental quality and global climate." Paper presented at the *Chapman Conference on Global Biomass Burning: Atmospheric, Climatic, and Biospheric Implications*, 19-23 March 1990, Williamsburg, Virginia, U.S.A.

Barnard, G.W. (1990), Use of agricultural residues as fuel. In: *Bioenergy and the Environment*. J. Pasztor and L.A. Kristoferson (eds.), Westview Press, Boulder, Colorado, U.S.A, pp. 85-112.

Barnard, G.W. and L.A. Kristoferson (1985), *Agricultural Residues as Fuel in the Third World*. Technical Report No. 5. Earthscan, London, UK.

Bolin, B., E.T. Degens, P. Duvigneaud and S. Kempe (1979), The global biogeochemical carbon cycle. In: *the Global Carbon Cycle*,. B. Bolin, E.T. Degens, P. Duvigneaud, and S. Kempe (eds.). SCOPE 13, Wiley, Chichester, UK. pp. 1-56.

Bouliere, F. and M. Hadley (1970), The ecology of tropical savannas. *Annual Review of Ecology Systems* 1:125-152.

Bucher, E.H. (1982), Chaco and Caatinga-South American arid savannas, woodlands and thickets. In: *Ecology of Tropical Savannas* B.J. Huntley, and B.H. Walker, (eds.) (Ecological Studies 42). Springer-Verlag, Berlin, Germany, pp. 48-79.

Coutinho, L.M. (1982), Ecological effects of fire in Brazilian cerrado. In: *Ecology of Tropical Savannas*. B.J. Huntley, and B.H. Walker (eds.)(Ecological Studies 42). Springer-Verlag, Berlin, Germany, pp. 273-291.

Crutzen, P.J. and M.O. Andreae (1990), "Biomass burning in the Tropics: Impact on atmospheric chemistry and biogeochemical cycles." *Science* 250:1669-1678.

Crutzen, P.J. (1983), "Atmospheric interactions – Homogenous gas reactions of C, N, and S containing compounds." In *The Major Biogeochemical Cycles and Their Interactions,* B. Bolin, and R.B. Cook (eds.),. SCOPE 21, John Wiley and Sons Inc., New York, U.S.A, pp. 67-114.

Delmas, R. (1993), "An Overview of Present Knowledge on Methane Emission from Biomass Burning." In: *Proceedings of an International IPCC Workshop: Methane and Nitrous Oxides, Methods in National Emissions Inventories and Options for Control, 3- 5 February 1993, Amersfoort, NL.* A.R, van Amstel (ed.). RIVM Report No. 481507003, Bilthoven, The Netherlands, July.

Delmas, R.A. and D. Ahuja (1993), "Estimating national methane emissions from anthropogenic biomass burning. Working group report: Methane emissions from biomass burning. In *Proceedings of an International IPCC Workshop: Methane and Nitrous Oxides, Methods in National Emissions Inventories and Options for Control, 3-5 February 1993, Amersfoort, NL.,* A.R. van Amstel (ed.), RIVM Report No. 481507003, Bilthoven, The Netherlands, July.

Elgin, J. (1991), National Programme Leader for Forages and Pastures, US Department of Agriculture, personal communication.

FAO/UNEP (1981), *Tropical Forest Resources Assessment Project.* FAO (Food and Agriculture Organization of the United Nations/United Nations Environment Programme), Rome, Italy.

FAO (1991), *Production Yearbook Volume 44.* FAO Statistics Series 94, FAO (Food and Agriculture Organization of the United Nations), Rome, Italy.

FAO (1993), *Forest Resources Assessment 1990* - Tropical Countries. FAO Forestry Paper No. 112, FAO, Rome, Italy.

Galbally, I.E. (1985), The emission of nitrogen to the remote atmosphere: Background paper. In: *The Biogeochemical Cycling of Sulfur and Nitrogen in the Remote Atmosphere,* J.N. Galloway, R.J. Charlson, M.O. Andreae, and H. Rodhe (eds.). D. Reidel, Dordrecht, The Netherlands, pp. 27-53.

González-Jiménez, E. (1979), "Primary and secondary productivity in flooded savannas." In UNESCO/UNEP/FAO (ed.). "Tropical grazing land ecosystems of Venezuela." *Nat. Resour. Res.* 16:620-625.

Haggar, R.J. (1970), "Seasonal production of *Andropogon gayanus*, 1. Seasonal changes in field components and chemical composition." *Journal of Agricultural Science,* 74:487-494.

Hall, D., F. Rosillot-Calle, R.H. William and J Wood (1994), *Network Newsletter*, Biomass users Network/UNCED. King's College, London, Campden Hill Road, London W8 7AH, UK.

Hao, W.M., M.H. Liu and P.J. Crutzen (1990), "Estimates of annual and regional releases of CO_2 and other trace gases to the atmosphere from fires in the tropics, based on the FAO statistics for the period 1975-1980." In: *Fire in the Tropical Biota, Ecosystem Processes and Global Challenges,* J.G. Goldammer, (ed.). Springer-Verlag, Berlin, Germany, pp. 440-462.

Harris, D.R. (1980), "Tropical savanna environments: Definition, distribution, diversity, and development." In: *Human ecology in savanna environments,* Harris, D.R. (ed.). Academic Press, New York, US, pp. 3-27.

Hopkins, B. (1965), “Observations on savanna burning in the Olokemeji Forest Reserve, Nigeria.” *Journal of Applied Ecology* 2:367-381.

Howden, S.M., G.M. McKeon, J.C. Scanlan, J.O. Carter and D.H. White (1996), “Methods for exploring options to reduce greenhouse gas emissions from tropical grazing systems.” *Climatic Change*. (In press.)

Huntley, B.J. (1982), “South African savannas.” In: *Ecology of Tropical Savannas,* B.J. Huntley and B.H. Walker (eds.),.(Ecological Studies 42). Springer-Verlag, Berlin, Germany, pp. 101-119.

Huntley, B.J. and J.W. Morris (1982), “Structure of Nylsvley savanna.” In: *Ecology of Tropical Savannas,* B.J. Huntley and B.H. Walker (eds.), (Ecological Studies 42). Springer-Verlag, Berlin, Germany, pp. 433-455.

Johansson, T.B., H. Kelly, A.K.N. Reddy and R.H. Williams (1992), *Renewable Energy: Sources for Fuels and Electricity*. Island Press, Washington, DC, U.SA.

Lacaux, J.P., H. Cachier and R. Delmas (1993), “Biomass Burning in Africa: An Overview of Its Impact on Atmospheric Chemistry.” In: *Fire in the Environment: The Ecological, Atmospheric and Climatic Importance of Vegetation Fires* P.J. Crutzen and J.G. Goldammer (eds.). J. Wiley & Sons Ltd, UK.

Lacey, C.J., J. Walker and I.R. Nolde (1982), Fire in Australian tropical savannas. In: *Ecology of Tropical Savannas,* B.J. Huntley, and B.H. Walker (eds.).. (Ecological Studies 42). Springer-Verlag, Berlin, U.S,A. pp. 246-272.

Lanly, J.P. (1982), *Tropical Forest Resources*. Food and Agriculture Organization of the United Nations (FAO) Paper 30, FAO, Rome, Italy.

Lashof, D.A. and D.A. Tirpak (eds.) (1990), *Policy Options for Stabilizing Global Climate*. Report to Congress, US Environmental Protection Agency, Washington, D.C., U.S.A.

Levine, J.S. (ed.) (1990), *Global Biomass Burning: Atmospheric, Climatic and Biospheric Implications*. The MIT Press, Cambridge, MA, U.S.A.

Lieth, H. (1978), *Patterns of Primary Productivity in the Biosphere*. Hutchinson Ross, Stroudsberg, PA, U.S.A..

Lobert, J.M., D.H. Scharffe, W.M. Hao and P.J. Crutzen (1990), “Importance of biomass burning in the atmospheric budgets of nitrogen-containing gases.” *Nature* 346:552-554.

Menaut, J.C., L. Abbadie, F. Lavenu, Ph. Loudjani and A. Podaire (1991), “Biomass burning in West African savannas.” In: *Global Biomass Burning*, J. Levine (ed.), MIT Press, Cambridge, MA, U.S.A.

Menaut, J.C., and J. César (1982), “The structure and dynamics of a West African savanna.” In: *Ecology of Tropical Savannas,* B.J. Huntley and B.H. Walker (eds.).(Ecological Studies 42). Springer-Verlag, Berlin, Germany, pp. 8-100.

Menaut, J.C. (1990), *Biomass burning in West African savannas*. Presentation given at the Chapman Conference on Global Biomass Burning. Williamsburg, Virginia, March 19-23. Proceedings forthcoming: MIT Press, Cambridge, MA, U.S.A.

Ponnamperuma, F.N. (1984), “Straw as a source of nutrients for wetland rice.” In: *Organic Matter and Rice*. International Rice Research Institute, Los Baños, Philippines. pp. 117-135.

San José, J.J. and E. Medina (1976), "Organic matter production in the Trachypogon savannas in Venezuela." *Tropical Ecology* 17:113-124.

Seiler, W. and P.J. Crutzen (1980), "Estimates of gross and net fluxes of carbon between the biosphere and the atmosphere from biomass burning." *Climatic Change* 2:207-247.

Singh, K.P. and R. Misra (1978), *Structure and Functioning of Natural, Modified and Silvicultural Ecosystems of Eastern Uttar Pradesh*. Technical Report MAB Research Project, Banaras Hindu University, Varanasi.

Strehler, A. and W. Stützle (1987), "Biomass residues." In: *Biomass: Regenerable Energy,* D.O. Hall, and R.P. Overend (eds.). John Wiley, Chichester, U.K., pp. 75-102.

US HEW (1970), *Air Quality Criteria for Carbon Monoxide*. US HEW, Washington, D.C., U.S.A.

Whittaker, R.H. and G.E. Likens (1975), "The biosphere and man." In: (1975), *Primary Productivity of the Biosphere,* H. Lieth and R.H. Whittaker (eds.). Ecological Studies No. 14. Springer-Verlag, New York-Heidelberg-Berlin.

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Bakken, L., K. Refsgaard, S. Christensen and A. Vatn (1994), "Energy use and emissions of greenhouse gases from grassland agricultural systems." In: *Grassland and Society,* L. 't Mannetje and J. Frame (eds.), Wageningen Pers, Wageningen, The Netherlands, pp. 361-376.

Barrow, N.J. and L.J. Lambourne (1962), *Australian Journal of Agricultural Research* 13:461-471.

Berounsky, V.M. and S.W. Nixon (1985), "Eutrophication and the rate of net nitrification in a coastal marine ecosystem." *Estuarine Coastal and Shelf Science* 20:773-781.

Berounsky, V.M. and S.W. Nixon (1993), "Rates of nitrification along an estuarine gradient in Narragansett Bay." *Estuaries* 16(4):718-730.

Bouwman, A.F. (1994), *Method to estimate direct nitrous oxide emissions from agricultural soils*. Report 773004004. National Institute of Public Health and Environmental Protection, Bilthoven, The Netherlands, p. 28.

Bouwman, A.F. (1995), "Compilation of a global inventory of emissions of nitrous oxide." Thesis Landbouwuniversiteti Wageningen. ISBN 90-5485-364-6, p. 143.

Bouwman, A.F. (1996), *Long-term scenarios of livestock-crop-land use interactions for the assessment of environmental indicators in developing countries*. FAO Report. (In press).

Bouwman, A.F., and W.G. Sombroek (1990), "Inputs to climatic change by soils and agriculture related activities: Present status and possible future trends." In: *Soils on a warmer Earth*, H.W. Scharpenseel, M. Schomaker & A. Ayoub (eds.). *Developments in Soil Science* 20. Elsevier, Amsterdam, pp. 15-30.

Bouwman, A.F. and K.W. Van Der Hoek (1991), *Analysis of soil and water borne emissions of nitrous oxide and methane in the Netherlands*. RIVM Report No. 736301010. National Institute of Public Health and Environmental Protection, Bilthoven, The Netherlands.

Bouwman, A.F., K.W. van der Hoek and J.G.J. Olivier (1995), "Uncertainties in the global source distribution of nitrous oxide." *J. Geophys. Res.*, 100:2785-2800.

Bowden, R.D., J.M. Melillo and P.A. Steudler (1991), "Effects of nitrogen additions on annual nitrous oxide fluxes from temperate forests soils in the northeastern United States." *J. Geophys. Res.* 96:9321-9328.

Brumme, R. and F. Beese (1992), "Effects of liming and nitrogen fertilization on emission of CO_2 and N_2O from a temperate forest" *J. Geophys. Res.* 97:12851-12858.

Burton, C.H., R.W. Sneath and J.W. Farrent (1993), "Emissions of nitrogen oxide gases during aerobic treatment of animal slurries." *Bioresource Technology* 5:233-235.

Carran, R.A., Theobald, P.W. and Evans, J.P. (1995), "Emission of nitrous oxide from some grazed pasture soils in New Zealand." *Australian Journal of Soil Research* 33:341-352.

Christensen, P.B. and J. Soerensen (1988), "Denitrification in sediment of lowland streams: regional and seasonal variation in Gelbaek and Rabis Beak, Denmark." *FEMS Microbiol. Ecol.* 53:335-344.

Christensen, P.B., L.P. Nielsen, N.P. Revsbech and J. Sorensen (1989), "Microzonation of denitrification activity in stream sediments as studies with a combined oxygen and nitrous oxide microsensor." *Appl. Environ. Microbiol.* 55:1234-1241.

Cole, V. et al., (1995), 'Agricultural Options for Mitigation of Greenhouse Gas Emissions'. In: *Climate Change 1995 - Impacts, Adaptations and Mitigations of Climate Change: Scientific - Technical analyses: The Second Assessment Report of the Intergovernmental Panel on Climate Change*, R. T. Watson, M. C. Zinyowera and R. M. Moss (eds.), Cambridge University Press, New York, U.S.A.

Cooper A.B. and J.G. Cooke (1984), "Nitrate loss and transformation in 2 vegetated headwater streams." *N. Z. J. Mar. Freshwater Res.* 18:441-450.

Czepiel, P., P. Crill and R. Harriss (1995), "Nitrous oxide emissions from municipal wastewater treatment." *Environ. Sci. Technol.* 29(9):2352-2356.

Danso, S.K.A. (1995), Sustainable agriculture. "The role of biological nitrogen fixing plants." In IAEA (ed.) *Nuclear Techniques in Soil-Plant Studies for Sustainable Agriculture and Environmental Preservation*, IAEA, Vienna, pp. 205-224.

Debruyn, W., G. Lissens, J. Van Rensbergen and M. Wevers (1994), "Nitrous oxide emissions from waste water." In: J. Van Ham et al. (eds.) 1994:159-165.

De Klein, C.A.M. and R.S.P. Van Logtestijn (1994), "Denitrification and N_2O emission from urine-affected grassland soil." *Plant and Soil* 163:235-242.

DeMore, W.B., S.P. Sander, D.M. Golden, R.F. Hampson, M.J. Kurylo, C.J. Howard, A.R. Ravishankara, C.E. Kolb and M.J. Molina (1994), *Chemical kinetics and photochemical data for use in stratospheric modeling*, JPL-NASA, Evaluation No. 11.

Dentener, F.J. and P.J. Crutzen. (1994), "A three-dimensional model of the global ammonia cycle." *J. Atmos. Chem.* 19:331-369.

Dowdell, R.J., J.R. Burford and R. Crees (1979), "Losses of nitrous oxide dissolved in drainage water from agricultural land." *Nature* 278:342-343.

Duff, J.H., F.J. Triska and R.S. Oremland (1984), "Denitrification associated with stream periphyton: Chamber estimates from undisrupted communities." *J. Environ. Qual.* 13(4):514-518.

Duxbury, J.M. and A.R. Mosier (1993) "Status and issues concerning agricultural emissions of greenhouse gases." In: *Agricultural Dimensions of Global Climate Change* H.M. Kaiser and T.E. Drennen (eds.). St. Lucie Press, Delray Beach, Florida, U.S.A, pp. 229-258.

Duxbury, J.M, D.R. Bouldin, R.E. Terry and R.L. Tate III (1982), "Emissions of nitrous oxide from soils." *Nature* 298:462-464.

Ecetoc (1994), *Ammonia emissions to air in Western Europe*. Technical Report No 62. Brussels, p. 196.

Erich, M.S., A. Bererie, and J.M. Duxbury (1984), "Activities of denitrifying enzymes in freshly sampled soils." *Soil Sci.* 38:25-32.

FAO (1978-1981), *Report of the Agro-Ecological Zones Project*. World Soils Resources, Project Volume 3, No. 48 South and Central America, FAO (Food and Agriculture Organization of the United Nations), Rome.

FAO (1990a), *Fertilizer Yearbook*. Volume 39. FAO statistics series No. 95. FAO, Rome.

FAO (1990b), *Fertilizer Yearbook*. Volume 43. FAO statistics series No. 94. FAO, Rome.

FAO (1991), *Agrostat PC, Land-use,* FAO, Rome.

Flessa, H., P. Doersch, F. Beese, H. Koenig and A.F. Bouwman, "Influence of cattle excrements on nitrous oxide and methane fluxes in pasture land" (submitted).

Galbally, I.E. et al., (1992), *Proceedings of IGBP Workshop No. 14*. Canberra, Oct. 3-5, 1990.

Goreau, T.J., W.A. Kaplan, S.C. Wofsy, M.B. McElroy, F.W. Valois and S.W. Watson. (1980), "Production of NO_2- and N_2O by nitrifying bacteria at reduced concentrations of oxygen." *Appl. Environ. Microbiol.* 30:526-532.

Groenestein, C.M., J. Oosthoek and H.G. van Faassen (1993), "Microbial processes in deep-litter systems for fattening pigs and emissions of ammonia, nitrous oxide and nitric oxide." In: *Nitrogen flow in pig production and environmental consequences* M.W.A. Verstegen, et al. (eds.). *Proc. First Int. Symp. on Nitrogen Flow in Pig Production and Environmental Consequences*, Pudoc Scientific Publ., Wageningen. pp. 307-312.

Groffman P.M., C.W. Rice and J.M. Tiedje (1993), "Denitrification in a Tallgrass Prairie Landscape". *Ecology* 74:855-862.

Guthrie, T.F. and J.M. Duxbury (1978), "Nitrogen mineralization and denitrification in organic soils." *Soil Sci. Soc. Am. J.* 42:908-912.

Hanaki, K., Z. Hong and T. Matsuo (1992), "Production of nitrous oxide gas during denitrification of wastewater." *Wat. Sci. Tech.* 26(5-6):1027-1036.

Haynes, R.J. and P.H. Williams (1993), "Nutrient cycling and soil fertility in the grazed pasture ecosystem." *Advances in Agronomy* 49:119-199.

Helder, W. and R.T.P. De Vries. (1983) "Estuarine nitrite maxima and nitrifying bacteria (ems-Dollard Estuary)." *Neth. J. Sea Res.* 17:1-18.

Hemond, H.F. and A.P. Duran (1989), "Fluxes of N_2O at the sediment-water and water-atmosphere boundaries of a nitrogen-rich river." *Water Resources Research* 25:839-846.

Hill, A.R. (1979), "Denitrification in the nitrogen budget of a river ecosystem." *Nature* 281:291-292.

Hill, A.R. (1981), "Nitrate-nitrogen flux and utilization in a stream ecosystem during los summer flows." *Can. Geogr.* 25(3):225-239.

Hill, A.R. (1983), "Nitrate-nitrogen mass balances for two Ontario rivers." In: *Dynamics of Lotic Ecosystems,* T.D. Fontaine and S.M. Bartell (eds.). Ann Arbor Science, Michigan.

Hong Z., K. Hanaki and T. Matsuo (1993), "Greenhouse gas-N_2O production during denitrification in wastewater treatment." *Water Sci.Technol.* 7:203-207.

Horrigan S.G. and A.L. Springer (1990), "Oceanic and estuarine ammonium oxidation: Effects of light." *Limnology and Oceanography* 4:316-328.

IPCC (1995), *IPCC Guidelines for National Greenhouse Gas Inventories*. OECD/OCDE, Paris.

Jarvis, S.C. and B.F. Pain (1994), "Greenhouse gas emissions from intensive livestock systems: their estimation and technologies for reduction." *Climate Change* 27: 27-38.

Jenkins, M.C. and W.M. Kemp (1984), "The coupling of nitrification and denitrifiation in two estuarine sediments." *Limnol. Oceanogr.* 29:609-619.

Jensen H.B., K.S. Jorgensen and J. Soerensen (1984), "Diurnal variation of nitrogen cycling in coastal marine sediments II. Nitrous oxide emissions." *Mar. Biol.* 83:177-183.

Jensen M.H., T.K. Andersen and J. Soerensen (1988), "Denitrification in coastal bay sediment: regional and seasonal variation in Aarhus Bight, Denmark." *Mar. Ecol. Prog. Ser.* 48:155-162.

Jorgensen, K.S., H.B. Jensen and J. Sorensen (1984), "Nitrous oxide production from nitrification and denitrification in marine sediments at low oxygen concentrations." *Can. J. Microbiol.* 30:1073-1078.

Kaspar, H.F. and J.M. Tiedje (1981), "Dissimilatory reduction of nitrate and nitrite in the bovine rumen: nitrous oxide production and effect of acetylene." *Appl. Environm. Microbiol.* 41:705-709.

Kaushik N.K. and J.B. Robinson (1976), "Preliminary observations on nitrogen transport during summer in a small spring-fed Ontario stream." *Hydrobiol.* 49(1):59-63.

Kemp, W.M., P. Sampou, J. Caffrey and M. Mayer (1990), "Ammonium recycling versus denitrification in Chesapeake Bay sediments." *Limnol. Oceanogr.* 35:1545-1563.

Koops, J.G., O. Oenema and van Beusichem (1995), *Stikstofverliezen door denitrificatie in grasland op veengronden,* NWI-LUW, Wageningen, p. 32.

Kroeze, C. (1994), *Nitrous Oxide; Emission inventory and options for control in the Netherlands*. Report No. 773001004. National Institute of Public Health and Environmental Protection, Bilthoven, The Netherlands, p. 163.

Kumarasinghe, K.S. and D.L. Eskew (1991), "Applications of N-15 isotope techniques in Azolla-rice studies." In: *Stable isotopes in plant nutrition, soil fertility and environmental studies* IAEA (ed.). IAEA, Vienna. pp.169-178.

Lipschultz, F., S.C. Wofsy and L.E. Fox (1986), "Nitrogen metabolism of the eutropic Delaware River ecosystem." *Limnology and Oceanography* 29:84-98.

Minami, K. and S. Fukushi (1984), "Methods for measuring N_2O flux from water surface and N_2O dissolved in water from agricultural land." *Soil Sci. Plant Nutr.* 30:495-502.

Minami, K. and A. Ohsawa (1990), "Emission of nitrous oxide dissolved in drainage water from agricultural land." In: *Soils and the greenhouse effect*, A.F. Bouwman (ed.) John Wiley & Sons, New York, pp. 503-509.

Monaghan, R.M. and B. Barraclough (1993), "Nitrous oxide and dinitrogen emissions from urine-affected soil under controlled conditions." *Plant and Soil* 151:127-138.

Mosier, A.R. (1994), "Nitrous oxide emissions from agricultural soils." *Fertilizer Research* 37:191-200.

Mosier, A.R. and W.J. Parton (1985), "Denitrification in a shortgrass prairie: A modeling approach." In: *Planetary Ecology* D.E. Caldwell, J.A. Brierley, C.L. Brierley (eds.) Van Nostrand Reinhold Co., NY, pp. 441-452.

Mosier, A.R., D.S. Schimel, D. Valentine, K.F. Bronson and W.J. Parton (1991), "Methane and nitrous oxide fluxes in native, fertilized, and cultivated grasslands." *Nature* 350:330-332.

Mosier, A.R., J.M. Duxbury, J.R. Freney, O. Heinemeyer and K. Minami. (1995a), "Nitrous oxide emissions from agricultural fields: Assessment, measurement and mitigation." *Plant and Soil.* (In press.).

Mosier, A.R., J.M. Duxbury, J.R. Freney, O. Heinemeyer and K. Minami (1995b) "Mitigating agricultural emissions of nitrous oxide." (Submitted to *Climatic Change*).

Muller, C., R.R. Sherlock and P.H. Williams (1995a), "In-field method to determine N_2O and N_2 emissions from denitrification and N_2O emissions from nitrification from urine-affected intensive pasture soil." (Submitted to *Soil Biology and Biochemistry.*)

Muller, C., R.R. Sherlock, and P.H. Williams (1995b), "Factors influencing N_2O emission from pasture soil affected by animal urine or ploughing." (Submitted *to European Journal of Soil Science.*)

Nobre, A.D. (1994), "Nitrous oxide emissions from tropical soils." Thesis. Univ. New Hampshire, Durham, NH., p. 141.

Nogita S., Y. Saito, and T. Kuge (1981), "A new indicator of the activated sludge process - nitrous oxide." *Water Sci.Technol.* 13:199-204.

Oenema, O. and G.L. Velthof (1993), "Denitrification in nitric-acid-treated cattle slurry during storage." *Netherlands Journal of Agricultural Science* 41:63-80.

O'Hara, G.W. and R.M. Daniel (1985), "Rhizobial denitrification: a review." *Soil Biol. Biochem*. 17:1-9.

Owens, N.J.P. (1986), "Estuarine nitrification: a naturally occurring fluidized bed reaction?" *Estuarine Coastal and Shelf Science* 33:31-44.

Parashar, D.C. (1991), "Measurement of greenhouse gas emissions in India." In: *Impact of Global Climate Change on Photosynthesis and Plant Productivity*, Oxford & IBH Publishing Co., New Delhi.

Peoples, M.B., D.F. Herridge and J.K. Ladha (1995), "Biological nitrogen fixation: An efficient source of nitrogen for sustainable agricultural production?" *Plant and Soil* 174:3-28.

Poggemann, S., F. Weissbach, and U. Kuenzel (1995), *N_2O-Freisetzung von Exkrementflecken auf dem Weideland. Abstract.* VDLUFA, Garmisch-Partenkirchen, p. 275.

Pollaris, L. (1994), “Emission of nitrous oxide from horticultural substrates.” (In Dutch) M.Sc. Thesis. Faculty of Agricultural and Applied Biological Sciences. University of Gent, p. 102.

Postma, R., O. Oenema, D.W. Bussink, M. Heinen and J. Van Mollenbroek (1994), “Gasvormige stikstofverliezen bij de teelt van sla op een zandbed onder glas.” *Meststoffen* pp. 28-34.

Robinson, J.B., H.R. Whiteley, W. Stammers, N.K. Kaushik and P. Sain (1979), “The fate of nitrate in small streams and its management implications,” pp. 247-259. In *Proc. 10th Annu. Agric. Waste Manage. Conf.*, Cornell Univ.

Ronen, D., M. Margaritz and E. Alman (1988), “Contaminated aquifers are a forgotten component of the global N_2O budget.” *Nature* 335:57-59.

Ryden, J.C. (1981), “N_2O exchange between a grassland soil and the atmosphere.” *Nature* 292:235-365.

Ryden, J.C. (1983), “Denitrification loss from a grassland soil in the field receiving different rates of nitrogen as ammonium nitrate.” *J. Soil Sci.* 34:355-365.

Safley, L.M., M.E. Casada, J.W. Woodbury and K.F. Roos (1992*), Global methane emissions from livestock and poultry manure*, USEPA Report No. 400/1-91/048. Office of Air and Radiation, Washington, D.C., USA, pp. 68, 145 and annexes.

Schepers, J.S. and A.R. Mosier (1991), “Accounting for nitrogen in nonequilibrium soil-crop systems.” In: *Managing Nitrogen for Groundwater Quality and Farm Profitability* R.F. Follett, D.R. Keeney and R.M. Cruse (eds.). Soil Science Society of America, Inc. Madison, Wisconsin, pp. 125-138.

Schimel, D.S., W.J. Parton, F.J. Adamsen, R.G. Woodmansee, R.L Senft and M.A. Stillwell, (1986), “The Role of Cattle in the Volatile Loss of Nitrogen from a Shortgrass Steppe.” *Biogeochemistry* 2:39-52.

Seitzinger, S.P. (1988), “Denitrification in freshwater and coastal marine ecosystems*:* ecological and geochemical implications.” *Limnol.Oceanogr.* 33:702-724.

Seitzinger, S.P. (1990), “Denitrification in aquatic sediments.” In: *Denitrification in soil and sediment* N.P. Revsbedch and J. Sorensens (eds.), Plenum Press, New York, U.S.A, pp. 301-312.

Seitzinger S.P. and C. Kroeze (1996). “Estimates of the spatial distribution of N inputs, nitrous oxide production and denitrification in freshwater and coastal marine ecosystems.” (In preparation).

Sherlock, R.R. and K.M. Goh (1983), “Initial emission of nitrous oxide from sheep urine applied to pasture soil.” *Soil Biology and Biochemistry* 15:615-617.

Sibbesen, E. and A.M. Lind (1993), “Loss of nitrous oxide from animal manure dungheaps,” Acta Agriculture Scandianavia Section B: *Soil and Plant Science*, 43:16-20.

Smith C.J., R.D. DeLaune and W.H. Patrick, Jr. (1985), “Fate of riverine nitrate entering an estuary: I. Denitrification and nitrogen burial.” *Estuaries* 8:15-21.

Spoelstra, S.F. (1985), “Nitrate in silage.” *Grass and Forage Science* 40:1-11.

Steffens, G. and H. Vetter (1990*), Neue Faustzahlen ueber Nachrstoffgehalte und Naehrstoffanfall*. Landwirtsschaftsblatt Weser-Ems. Vol. 3.

Swank, W.T. and Caskey, W.H. (1982), “Nitrate depletion in a second-order mountain stream.” *J. Environ. Qual.* 11:581-584.

Tiedje, J.M. (1988), "Ecology of denitrification and dissimilatory nitrate reduction to ammonia." In: *Biology of Anaerobic Microorganisms,* A.J.B. Zehnder (ed.), Wiley, New York, USA, pp. 179-243.

Ueda, S., N. Ogura and E. Wada (1991), "Nitrogen stable isotope ratio of groundwater N_2O." *Geophysical Research Letters* 18(8):1449-1452.

Ueda, S., N. Ogura and T. Yoshinari (1993), "Accumulation of nitrous oxide in aerobic groundwaters." *Wat. Res.* 27(12):1787-1792.

Van Cleemput, O., A. Vermoesen, C.J. De Groot, and K. Van Ryckeghem (1994), "Nitrous oxide emission out of grassland." *Environ. Monitor. Assessm.* 31:145-152.

Van der Hoek, K.W. (1994*), Berekeningsmethodiek ammoniamemissie in Nederland voor de jaren 1990, 1991 en 1992.* RIVM Report No. 773004003.

Van Kessel, J.P. (1977), "Removal of nitrate from effluent following discharge on surface water." *Water Research* 11:533-537.

Van Vuuren, A.M. and J.A.C. Meijs (1987), In: *Animal Manure on grassland and fodder crops: fertiliser or waste?* H.G.Van der Meer, R.J. Unwin, T.A. van Dijk and G.C. Ennik (eds.), Nijhoff, Dortrecht, The Netherlands, pp. 27-45.

Velthof, G.L. and O. Oenema (1993), "Orienterende lachgasemissiemetingen in RWZI's." In Capele, A.D. and E.N. Alblasserdam (eds.) *NMI-Report 93.263*. Wageningen, p. 6.

Velthof, G.L. and O. Oenema (1994), "Effect of nitrogen fertilizer type and urine on nitrous oxide flux from grassland in early spring." In *Grassland and Society, Proceedings of the 15th General Meeting of the European Grassland Federation*. L. 't Mannetje and J. Frame (eds.) pp. 458-462.

Velthof, G.L. and O. Oenema (1995), "Nitrous oxide fluxes from grassland in the Netherlands: II. Effects of soil type, nitrogen fertilizer application and grazing." (Accepted for publication in *European Journal of Soil Science*.)

Velthof, G.L., A.B. Brader, and O. Oenema (1996a), "Seasonal variations in nitrous oxide losses from managed grasslands in the Netherlands." (Submitted to *Soil Biology and Biochemistry*).

Velthof, G.L., S.C Jarvis, A. Stein, A.G. Allen, and O. Oenema (1996b), "Spatial variability of nitrous oxide fluxes in mown and grazed grasslands on a poorly drained clay soil." (Submitted to *Soil Biology and Biochemistry*).

Vetter, H., A. Klasink and G. Steffens (1989), "Mist und Guelleduengung nach Mass." *VDLUFA-Schriftenreihe* 19:41-66.

Whitehead, D.C. (1970), *The role of nitrogen in grassland productivity*. Bulletin 48 of Commonwealth Agricultural Bureau, Hurley, UK.

WHO/UNEP (World Health Organisation/United Nations Environmental Programme) (1989), *Global Pollution and Health. Results of health related environmental monitoring, World Health Organization*. Geneva and United Nations Environment Programme, Nairobi, Yale Press, London.