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Greenhouse Gas Inventories for Agriculture in the Nordic Countries

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Preface

The emission of greenhouse gases by agriculture constitutes, particularly for methane and nitrous oxide, a significant part of total anthropogenic emissions. National inventories are, however, characterized by uncertainties and differences which complicate the identification of effective mitigation options.

On 24-25th January 2002 a workshop was held in Snekkersten, Denmark. The aims of the workshop were three-fold:

- to discuss aspects of agricultural production in the Nordic countries relevant to greenhouse gas emissions;
- to compare emissions measurements and inventories for the Nordic countries with the IPCC methodology for calculating greenhouse gas emissions;
- to discuss the need for a common approach that takes specific conditions in the Nordic countries into consideration.

Experts were invited to present and discuss the current knowledge on various aspects of greenhouse gas emissions from agriculture. This report contains the papers presented at the workshop, as well as an overview of emissions inventories in the Nordic countries. The first chapter summarizes a number of conclusions derived from the presentations and from the general discussion at the workshop.

The workshop was organised by a joint Nordic working group including scientists and government officials from Iceland, Finland, Norway, Sweden and Denmark. The members of the working group were Jørgen E. Olesen (Denmark, chairman), Søren O. Petersen (Denmark, secretary), Kristin Rypdal (Norway), Rolf Adolfsson (Sweden), Birna S. Hallsdottir (Iceland), Martti Esala (Finland) and Jørgen Fenhann (Denmark). The working group and the workshop were funded by the Nordic Council of Ministers under contract 6700134-Y537.

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Dansk sammendrag af hovedkonklusioner

De aktuelle opgørelser af drivhusgasudledninger fra landbruget i de nordiske lande er alle baseret på det internationale klimapanel IPCC's retningslinier. Imidlertid er der forskel på de enkelte landes konkrete anvendelse af den anbefalede metodik. Forskellige detaljeringsgrader (tiers) benyttes, og i en række tilfælde er nationale tilpasninger af metodikken anvendt til estimering af kilder og emissionsfaktorer. Derfor findes der i øjeblikket ikke en fælles sammenhængende og gennemskuelig metodik til beregning af drivhusgasudledninger indenfor Norden.

Effektiv prioritering af tiltag til begrænsning af landbrugets drivhusgasudledninger forudsætter en metodik, som inkluderer alle relevante drivhusgasser, metan (CH_4), lattergas (N_2O) og kuldixoid (CO_2). Kulstoflagring i landbrugsjord er ikke i øjeblikket inkluderet i emissionsopgørelserne, og metodikken til estimering af CO_2-fluxe fra landbruget er under revision. Det største problem i opgørelsen af CO_2-balancen er verificérbarheden, idet CO_2-fluxen kan være høj selv ved små ændringer i jordens kulstoflager.

Der er et stort behov for forbedring af IPCC-metodikken. Den bør tilpasses lokale forhold, men på grundlag af fælles retningslinier. Klimatiske variationer indenfor Norden bør afspejle sig i opgørelserne af metanemissioner fra husdyrgødningslagre, og der er behov for en større differentiering af emissionsfaktorerne for lattergas fra både direkte og indirekte kilder. For nogle af udledningerne fra husdyrgødningshåndtering er der brug for en reevaluering af principperne bag den aktuelt anvendte IPCC-metodik.

Forbedringer af metodikken kan ikke ske uden indsigt i systemerne og de bagvedliggende mekanismer. For nogle kilder til drivhusgasser foreligger der en mængde eksperimentelle data, og tilstrækkelig med viden til at forbedringer kan gennemføres. Dette er ikke tilfældet for alle kilder, og der er et klart behov for en modelbeskrivelse af landbrugssystemet, hvorfra simple, men pålidelige metoder til forbedring af emissionsopgørelserne kan udledes.

The need for truly common Nordic guidance on greenhouse gas emissions inventories for agriculture

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Summary

Current greenhouse gas emissions inventories for agriculture in the Nordic countries are all based on the IPCC guidelines. However, there are discrepancies between countries in the application of these methodologies. Different tiers of the methodology are used in the different countries, and national adaptations of the methodology have in many cases been used for estimating activities and emission factors. There is thus currently not a common consistent and transparent methodology for greenhouse gas emission inventories at the Nordic level.

Effective uptake of mitigation options requires a methodology that properly covers all agricultural greenhouse gases, CH_4, N_2O and CO_2. Carbon storage in agricultural soils is currently not covered in the emission inventories, and the methodology for making inventories is currently under review. The main problem of estimating CO_2 fluxes from agriculture is that of verifiability, because large fluxes may occur from only small changes in the carbon stock.

There is a great need to improve this IPCC methodology and to make it more locally adapted, but based on common guidelines. The climatic variation within the Nordic countries needs be accounted for in the estimates of methane emissions from manure, and the emission factors for nitrous oxide from both direct and indirect sources should be differentiated more than what is currently the case. For some of the emissions from manure management, there is a need to re-evaluate the principles of the current IPCC methodology.

Improvement of the methodology cannot happen without insight into the systems and the mechanisms behind. For some of the emissions sources there is a lot of experimental data available, and sufficient knowledge to improve the methodology. However, this is not the case for all sources, and there is a clear need to apply systems modelling and to derive simple, but reliable methods that can improve the emissions inventories.

Introduction

Agriculture contributes significantly to greenhouse gas (GHG) emissions, primarily due to emissions of methane and nitrous oxide. The share of agricultural CH_4 and N_2O emissions of the total national GHG emission vary between the Nordic countries, from 7% in Finland to 16% in Denmark. CO_2 from soils is reported under Land Use Change and Forestry (LUCF), and this is currently not included in the national emissions totals.

The agricultural GHG emissions originate primarily from biological processes associated with enteric fermentation, handling of manure and from crop production. Many of these processes are complex and occur in a complex environment, which often is also not well defined. Accordingly, there are large uncertainties associated with emissions estimates (Rypdal, 2002), and these uncertainties also make it difficult to evaluate efficiencies of mitigation measures.

Current greenhouse gas emissions inventories in the Nordic countries are all based on the IPCC guidelines (IPCC, 1997, 2000). However, there are discrepancies between countries in the application of these methodologies. Different tiers of the methodology are used in the different countries, and national data have often been used for estimating activities and emission factors (Petersen et al., 2002).

The GHG emission inventories should be accurate, transparent, complete and consistent. The large differences between Nordic countries in the application of the IPCC methodology for emissions inventories reduces the comparability of the inventories, and this calls for some common guidance in the Nordic region on greenhouse gas emissions inventories from agriculture. Such common guidance should be so detailed that they would promote the uptake of cost-effective abatement strategies.

Climatic conditions

The average annual temperatures at sites representative for agricultural areas in the Nordic countries range from 3.9 to 7.1 °C (Petersen et al., 2002). This is considerably less than the limit of 15 °C set by the IPCC for cool regions. The temperature affects most biological processes, and in particular the emissions from manure management depend strongly on temperature during manure storage. It was thus demonstrated that methane emissions from stored slurry differ by 30 to 40% with the climate gradient in the Nordic countries (Petersen et al., 2002).

It has been demonstrated in several studies that nitrous oxide emissions may occur at high rates, even in frozen soil (Martikainen, 2002). In boreal regions winter emissions of nitrous oxide can account for more than 50% of the annual emissions. The emissions at low temperatures can be greatly enhanced by freezing-thawing cycles. The interactions between soil physics, chemistry, microbiology and N_2O production at low temperatures are still poorly understood. These winter emissions also seem to be independent of nitrogen input (Lægreid & Aastveit, 2002). It is therefore difficult to estimate how much of the N_2O emission at low temperature that is anthropogenic, i.e. attributed to agriculturally derived N or cultivation of soils. The IPCC methodology is only concerned with the emissions attributable to human interference. This calls for new experiments and studies to separate the natural from the anthropogenic influence on low temperature N_2O emissions.

Enteric fermentation

Cattle are the most important methane producing animals in the Nordic countries, and dairy cattle constitute by far the largest proportion of the cattle. The dairy cows in the Nordic countries are on a global scale very productive, and this productivity has increased considerably over time, effectively reducing the methane production per unit product. The increasing productivity of dairy cows has meant a decrease in population size. However, at the same time there has been an increase in the population of beef cattle. As these suckler cows are often fed and housed differently from the dairy cows, it becomes increasingly important to separate these two groups in the emission inventories.

The IPCC tier 2 method for estimating methane from enteric fermentation is based on estimation of energy use by the animals. The methods for estimating energy content in feed vary considerably in the Nordic countries. There are thus currently four different systems in operation. For example, in Sweden a national methodology based on feed energy requirements expressed as metabolisable energy is used to estimate emissions factors for dairy cows. It was recently revised, but still gives about 10% higher values than the methodology based on net energy recommended by the IPCC. The Swedish emission factor for dairy cows, 130 kg CH4/ head/yr, also differs considerably from the 104 kg/CH4 head/yr used in Denmark (1995). However, this difference is completely eliminated when using input data valid for Danish conditions, mainly animal weight, activity, milk yield and feed quality, into the calulations scheme for the Swedish cow polulation (Staaf unpublished). This indicates that it is possible to compare the various approaches in the Nordic countries. Anyway, there may be a need to combine the specific feed energy data used in national inventories with methane conversion rates that are adapted to the local system. Despite these difficulties, estimates of methane emissions from enteric fermentation are less uncertain than many of the other sources of methane and nitrous oxide emission from agriculture. This is due to the good statistics on cattle population and productivity and the large knowledge base on the factors affecting methane production.

There are a number of ways to reduce methane emissions from enteric fermentation. However, none of the seem to be currently feasible, either due to their costs, effect on landscape or acceptance in the public (Bertilsson, 2002). One of the only effective, acceptable and economically feasible options currently available seems to be to accept and possibly reinforce the general trend of increasing the productivity of the animals.

Manure management

Manure management in the animal houses and during storage emits both methane and nitrous oxide to the atmosphere. For all Nordic countries two gases contributes equally to the global warming, i.e. measured in CO_2 equivalents (Petersen et al., 2002). However, there is some variation between the countries with the share of nitrous oxide being only 33% in Norway, but 66% in Finland. Such differences are caused by differences in manure type and handling.

The primary manure management strategies used in the Nordic countries are slurry systems, deep litter systems and separate systems, where farmyard manure and liquid manure are collected separately. The slurry and deep litter systems are becoming the dominant manure management systems.

A distinction should be made between emissions that occur during storage inside and outside animal houses. The IPCC methodology does not make this distinction, and for example methane emissions from the deep litter mat in the animal house does not seem to be included in the emissions inventories (Hansen et al., 2002). There is a major emission of methane from deep litter mats in the animal house, but almost no emission of nitrous oxide (Sommer & Petersen, 2002). Little is known about the methane emissions from slurry stored in the house. It can, however, be assumed to be significant, in particular in insulated houses, where slurry temperature is relatively high also during winter. Nitrous oxide emissions from animal houses probably mainly occur from slatted floors in slurry systems and manure/air interfaces in tie stall systems.

The emissions of both methane and nitrous oxide from solid manure stores strongly depend on the temperature and flow in the manure heap. During composting there may be a methane emissions, whereas nitrous oxide emissions primarily occur at lower temperatures in the heap. Temperature has been found to strongly affect methane from slurry storages, but the level varies considerably between stores.

The IPCC methodology assumes that the nitrous oxide emissions from manure storages is a fixed proportion of the nitrogen excreted. However, estimates of N2O emissions from slurry stores should preferably be based on surface area, ammonium content and water balance. The emissions of both methane and nitrous oxide from solid manure heaps should consider surface area and the potential for composting (bulk density and moisture content).

It should be noted that emissions of both methane and nitrous oxide from manure management have large uncertainties, as there are only few studies in this area. There is thus a great need for further studies that may serve as basis for a revision of the IPCC methodology.

Nitrous oxide emissions from soils

Nitrous oxide emissions from agricultural soils originate from two microbial processes that both depend on the availability of nitrogen; nitrification and denitrification. The production of N_2O occurs at higher rates when the oxygen content in the soil is depleted, which may occur due to high soil moisture contents or due to locally high microbial activity in the soil. High N_2O rates are therefore often also associated with availability of easy decomposable carbon sources in the soils.

The current IPCC methodology assumes that all inputs of nitrogen lead to nitrous oxide emissions. For the input of N in biological fixation, this probably leads to double counting, since the N-fixation takes place inside the plants, and the N that contributes to N_2O emissions is the N made available to the soil microorganisms. This soil N from N fixation is counted in either crop residues or manure.

The IPCC methodology applies the same overall emission factor for all N inputs. This emission factor was originally derived from whole year studies in the USA and UK (Bouwman, 1996). This study showed a clear relationship between nitrogen application rate and nitrous oxide emissions, indicating that 1.25±1.0% of the added N was emitted as N_2O during one year. However, analysis of compilations of more and newer datasets does not give as clear a picture on the effect of N application rate on nitrous oxide emissions (Kasimir Klemedtsson & Klemedtsson, 2002; Lægreid & Aastveit, 2002).

Analysis of the available datasets on nitrous oxide emissions from soil have shown that the emissions are generally higher following application of manure compared with mineral N fertilisers. Kasimir Klemedtsson (2001) suggested an emission coefficient for mineral fertiliser of 0.8% of added N and for manure 2.5% of added N. This lower emission factor for mineral N fertiliser is supported by the study of Lægreid & Aastveit (2002). However, they found that the emission factor was higher in carbon rich soils.

The IPCC emission factor is essentially based on estimates of the initial burst of N_2O following fertilizer and manure application that may last for up to two months, while a second component is long-term and due to nitrogen in organic matter accumulating the soil. This second component is only partly covered by the IPCC methodology, i.e. through the effect of fertilizer and manure application on nitrogen returned in crop residues and other forms of N bound inorganic matter. The IPCC methodology may therefore overestimate effects of recent additions, but underestimate long-term effects. Kasimir Klemedtsson & Klemedtsson (2002) proposed a background emission of 0.5 kg N_2O-N ha^{-1} yr^{-1} for mineral soils under Nordic conditions to compensate for the lack of an emission factor for the long-term effect due to past N additions. However, this may be an overestimation since the IPCC methodology also includes the recycling of N in manure and crop residues. Also it is not clear what the true background emissions from natural ecosystems are. Here there will also be a recycling of N resulting in N_2O emissions.

There appears to be a large uncertainty regarding the estimation of amount of N in crop residues, and the methodology is also different in IPCC (2001) compared with IPCC (1997). The uncertainty can be illustrated by the fact that N_2O emissions of 6.2 kt yr^{-1} were estimated from Denmark in 1999, but only 1.3 kt yr^{-1} from Sweden, even though emissions from application of mineral fertiliser and manure only ranged from 8.4 kt N_2O yr^{-1} in Denmark to 4.6 kt yr^{-1} in Sweden (Petersen et al., 2002). These estimates were obtained using similar emission factors, and the differences must therefore be attributed to differences in the methodology for estimating N in crop residues. It seems unlikely that the amount of N in crop residues would vary by a factor of 5 between Denmark and Sweden. There is therefore a need to develop and adopt comparable methodologies on this item on a Nordic basis.

This calls for measurements of nitrous oxide emissions in long-term fertiliser and crop rotation experiments. There is also a need for use of dynamic simulation models to better understand the influence of different management factors on nitrous oxide emissions. These models may be tested against the large base of N_2O measurements, but this requires that sufficient additional data is measured and reported in these studies (Lægreid & Aastveit, 2002). Such generally applicable models may also be used to derive simpler models for use in emissions inventories. This will probably lead to different emission factors for the different inputs into the system. However, there does not currently seem to be any justification for country specific emission factors for soils.

Indirect emissions of nitrous oxide

The indirect emissions of nitrous oxide are those associated with emissions during microbial turnover after the nitrogen has left the agricultural system. This is associated with three components; nitrate lost by leaching or runoff, ammonia volatilisation, and human sewage. Agricultural management may primarily influence the two first components.

The emission factor for nitrate leaching is the highest used in the IPCC methodology, 2.5% N_2O-N of N lost by leaching. This is the sum of three components along the flow path; 1.5% from groundwater, 0.75% from rivers and 0.25% from estuaries. There are very few data in general to support emission coefficients of this type, and some data support these values, while other data suggest lower emission coefficients. However, it is clear that there is a great variation in the ability of ground water, riparian zones, wetlands, rivers and estuaries to process nitrate and produce nitrous oxide, in particular the ratio of N_2:N_2O may have tremendous spatial variation (Groffman et al., 2002). It may be critical to include this spatial variation in emissions in the methodology, in order to reduce uncertainties associated with this emission.

Sweden has in its national emission inventories adopted at considerably smaller emission factor of only 0.25% as opposed to the 2.5% of the official IPCC methodology (Kasimir Klemedtsson, 2001). Whereas there may be reasons to believe that the official emission factor may be too high, it remains questionable whether there currently is sufficiently documentation to substantially change this figure. During 2002 Sweden has reconsidered this low emission factor and decided to adopt the 2.5% recommended by IPCC for the calculation of emissions for 1990-2001 to be reported to UNFCCC during 2003. Given the magnitude of the nitrous oxide emissions from N lost by leaching, it should be a research priority to provide better estimates of this emission, both through more measurements and through the use of models that track the nitrogen on its path through the landscape. This may be obtained by adding the issue of N_2O emissions to current national measurement programs of movement of water and N across the landscape.

Carbon storage in soils

Changes in the agricultural soil carbon pool are not reported in the Agriculture chapter of the IPCC methodology, but under Land Use Change and Forestry (LUCF). The emissions and sinks reported under LUCF are currently not accounted for the national totals. However, carbon storage in cropland and grazing land are now considered in article 3.4 of the Kyoto Protocol.

The main problem of including agricultural soil carbon stock changes in the inventories of net greenhouse gas emissions is that of verifiability. The soil carbon pools are large and the changes are slow. However, even small changes in soil carbon pools may contribute significantly to national greenhouse gas emissions in the Nordic countries. Such small relative changes in soil carbon pools are very difficult to determine from soil sampling (Heidmann et al., 2002). The cost of demonstrating a change in soil carbon storage may be exceedingly large if based on soil sampling only. However, the costs may be reduced by using locally calibrated models (Andrén & Kätterer, 2002; Smith, 2002).

Mitigation options

The most cost-effective mitigation strategies for agricultural greenhouse gases simultaneously reduce the emissions of several greenhouse gases. Examples are anaerobic digestion of manure and production of biomass for energy. Both strategies reduce greenhouse emission by substituting fossil energy use. In addition production of biomass for energy may reduce nitrous oxide emissions and increase carbons storage in soils, in particular for perennial energy crops (Olesen, 2002). Anaerobic digestion reduces the methane and nitrous oxide emissions during storage, and also nitrous oxide emissions in the soil, because the amount of volatile solids (VS) in the digested slurry is lower than in the slurry (Sommer et al., 2002). The lower VS content reduces the oxygen deficiency induced by high microbial turnover rates, and this leads to lower nitrous oxide emissions.

The current IPCC methodology does not include the effects of carbon sequestration under bioenergy crops or the effects of lower VS content in the manure on N_2O emissions from soils. There is a need to include all effects of mitigation measures in the emissions inventories. Otherwise the full benefits of the mitigation options may not be obtained, or less efficient options may be selected based on erroneous assumptions. An example of this is bioenergy crops, where perennial energy crops provide the highest reductions of greenhouse gases, when all gases including carbon sequestration in soils are considered. However, annual bioenergy crops are almost as efficient when the soil carbon sequestration is ignored.

Conclusions

There are currently large differences between the Nordic countries in the application of the IPCC methodology. In some cases the estimates of activities differ considerable, e.g. the amount of N in crop residues, in other cases different emission factors were applied. The agricultural structure varies somewhat within the Nordic countries, mainly reflecting the climatic conditions, which restricts the growing season at higher latitudes. However, this cannot justify the relatively large differences in the application of the IPCC method in the different Nordic countries.

The current emission inventories are from a Nordic perspective neither transparent nor consistent. Such differences reduce the possibilities of implementing joint Nordic (or EU) schemes for mitigating agricultural greenhouse gas emissions. Many of the differences in the emission inventories arise because of uncertainties associated with the current methodology, in particular for the nitrous oxide emissions. There is a great need to improve this methodology and to make it more locally adapted, but based on common guidelines. The climatic variation within the Nordic countries should thus be accounted for in the estimates of methane emissions from manure, and the emission factors for nitrous oxide from both direct and indirect sources should be differentiated more than what is currently the case. For some of the emissions from manure management, there is a need the re-evaluate the principles of the current IPCC methodology.

Improvement of the methodology cannot happen without insight into the systems and the mechanisms behind. For some of the emissions sources there is a lot of experimental data available, and sufficient knowledge to improve the methodology. However, this is not the case for all sources, and there is a clear need to better link experimental data with the use of systems modelling, in order to improve the understanding and derive simple, but reliable methods that can improve the emissions inventories.

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General guidance and procedures for estimating and reporting national GHG emissions for agriculture

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Summary

Greenhouse gas (GHG) emissions from agriculture account for a large share of total GHG emissions in most countries. Methane from ruminants, animal manure and rice fields, and nitrous oxide from agricultural soils are among the most important sources. In general, these emission estimates also are more uncertain than most other parts of the GHG emission inventory. IPCC has developed guidelines for estimating and reporting emissions of GHG. These guidelines shall be followed to secure complete, consistent, accurate and transparent reporting of emissions. However, the recommended methodologies are tiered, and choice of methods shall preferably reflect national circumstances, the national importance of a source, and different resources to prepare inventories. A country may also apply a national methodology given that it is well documented and not in conflict with good practice. Emission data reported under the United Nation Framework Convention on Climate Change are subject to external control, and the methodologies are reviewed by experts on agricultural inventories.

Introduction

Inventories of GHG emissions are important in order to formulate cost-effective abatement strategies and as input to climate modelling. Official GHG inventories are reported annually by each country to the UNFCCC (United Nation Framework Convention on Climate Change). GHG inventories will also be used to monitor the commitments made under the Kyoto protocol.

The Kyoto protocol restricts the total GHG emissions of each signature country. The protocol also opens up for emission trading, which implies a need for high quality emission data. According to IPCC (2001) there are several requirements for GHG inventories:

- Accuracy (minimize uncertainties, eliminate bias)
- Transparency (reporting of detailed estimates with all data and assumptions documented)
- Completeness (emissions from all sources and sinks shall be estimated)
- Consistency (the same data and assumptions shall be used across sources and across all years).

GHG emissions in the Nordic countries

The GHG emissions in each Nordic country are shown in Table 1. The importance of the agricultural sector for GHG emissions is highest in Denmark, where it accounts for 16% of total emissions[1]. The share is lowest in Finland, 7% of the total. The emissions from Denmark are also highest in absolute terms. Agriculture is the main source of N_2O emissions in all countries. Agriculture is also the most important source for CH_4 emissions in all countries except Norway and Finland.

Table 1. Total and agricultural GHG emissions from the Nordic countries in 1998 (million tonnes CO_2 equivalents).

	CO_2	CH_4	N_2O	Total[1]
Denmark				
Total	60.1	6.0	9.5	75.6
Agriculture	-	3.9	8.6	12.4
% agriculture	0.0	64	91	16
Finland				
Total	63.9	4.1	7.9	76.0
Agriculture	-	1.7	4.0	5.7
% agriculture	0.0	41	50	7
Iceland[2]				
Total	2.1	0.29	0.13	2.6
Agriculture	-	0.25	0.07	0.32
% agriculture	0.0	85	52	12
Norway				
Total	41.7	7.3	5.1	54.1
Agriculture	-	2.3	2.6	4.9
% agriculture	0.0	32	51	9
Sweden				
Total	57.0	5.4	8.0	70.3
Agriculture	-	3.5	4.9	8.2
% agriculture	0.0	62	61	12

[1] Excluding emissions from fuel combustion, PFCs, HFC and SF_6 and emissions and sinks reported under LUCF.
[2] Data for 1990
Source: UNFCCC

Emissions from various agricultural sources in the Nordic countries are shown in Table 2. According to the reported figures, emission of N_2O from agricultural soils is the single most important source of GHG emissions from agriculture, while methane from enteric fermentation accounts for only half this value. It is expected that cattle, followed by sheep, contribute most. Note that there may be sources

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[1] The figures include process related emissions only.

not reported, and that CO_2 from soils is to be reported under Land Use Change and Forestry (LUCF) and not Agriculture. Emissions and sinks reported under LUCF are currently not accounted for in the national totals.

Table 2. Emissions[1] of CH_4 and N_2O from agriculture in Denmark, Finland, Iceland, Norway and Sweden in 1998, by source (million tonnes CO_2 equivalents).

	CH_4	N_2O	Fraction of total emissions from agriculture
Enteric fermentation	9.6	NA	30.5
Manure management	1.8	1.5	10.6
Agricultural soils	0	18.6	59.1
Field burning of agricultural residues	NE[2]	NE[2]	-
Other	0	0	-

[1] The table includes process related emissions only. Emissions from transport and stationary fuel combustion are not included.

[2] No figures have been reported. However, field burning is not expected to be common in the Nordic countries.

Source: UNFCCC

Uncertainties

Estimates of uncertainties in GHG emissions from agriculture will to a large extent be based on expert judgments. According to Rypdal & Winiwarter (2001), different experts may have different opinions on the uncertainties. An overview is given in Table 3. However, all studies rank the agricultural sources to have high uncertainty compared to the national total inventory uncertainty of 10-20% (excluding LUCF). All studies also conclude that nitrous oxide from agricultural soils gives the highest contribution to total inventory uncertainty. In order to decrease the overall uncertainty in total GWP weighted emissions, it is thus crucial to reduce the uncertainty of this particular source.

Table 3. Assessed uncertainties for agricultural sources of GHG in a few countries. Uncertainties are expressed as two standard deviations in percentage of source level.

	Austria	Norway	The Netherlands	UK	USA
Enteric fermentation (CH_4)	±50	±25	±25	±20	±36
Manure management (CH_4)	..	±25	±25	±30	±36
Agricultural soils (N_2O)	-68 to +934	Two orders of magnitude	±75	Two orders of magnitude	-90 to +100

Source: Rypdal & Winiwarter (2001) and references therein.

Methodologies and good practice for preparing GHG inventories

Methodologies for preparing GHG inventories for agriculture and other sources were proposed by IPCC (1997), the so-called IPCC guidelines. Further guidance is given in IPCC (2001), the so-called Good Practice guidance, which supplements IPCC (1997) and also gives some corrections to algorithms and updated emission factors. It also gives advice on livestock population characterization for use in the calculations. IPCC (2001) gives general guidance on uncertainties, verification and quality assurance/quality control.

For most sources the guidelines propose methods at different levels of sophistication (tiers). Tier 1 is a default method that can be applied by all countries without extensive data collection. The higher tiers will be more accurate, but also require more input data. The higher tiers will best reflect national circumstances. For estimation of N_2O from agricultural soils, only one method (tier) is proposed.

Many countries have developed national methodologies. This is particularly relevant for agricultural sources where, e.g., climate conditions and national practices may influence the results in a way that is not captured by the IPCC methods. The Good Practice guidelines allow for use of a national methodology, given that it is well documented (transparent) and is not in conflict with the general good practice guidance. It is an advantage if the national methodology has been published in a refereed scientific journal. More often, however, national emission factors are used in the higher tier methods to reflect national circumstances.

The Good Practice guidance also gives advice on choice of methods among the various tiers. The general rule is that if a source is not a key source (important with respect to the determination of the total emission level and trend), the simple Tier 1 method is appropriate. If it is a key source, efforts should be made to use the higher tier methods, preferably in combination with national, well-documented emission factors. For example, methane from enteric fermentation in cattle will often be considered a key source, while methane from enteric fermentation in swine is not. This means that Tier 2 should be used for cattle, but that Tier 1 is appropriate for swine.

Countries are encouraged to move to higher tiers and change emission factors if this can reduce uncertainties or improve the good practice requirements in other aspects. When methodologies are changed, it is important to ensure consistency in time-series by re-calculating back to the base year (1990). This means that published emission data can be changed.

Control and review of GHG inventories

High quality emission inventory data are essential for the implementation of the UNFCCC protocol and the Kyoto protocol. This requirement has been met by a review system of GHG inventories. The first step is a so-called Synthesis & Assessment, where aggregated emission factors are compared to emission factors for other countries. Also the time-series' consistency is checked. When outliers are detected, the country will have to explain these. In the Synthesis & Assessment, comparisons with international statistics are also made, for example with respect to animal populations.

The next step is a review. A review may take place in the reporting country (In-country review), in the country of each reviewer (Desk review) or centrally (Central review). In a review team there will be sectorial experts, as well as general inventory experts. The team will review whether the inventory methods are in accordance with good practice. They will also review assumptions and national methodologies and emission factors. The country reporting the inventory is supposed to revise its inventory based on the feedback from the reviewers.

Conclusions

The GHG emissions from agriculture are high and also uncertain. There is consequently a need to improve the estimates of emissions from some sources. Emissions of methane from cattle (enteric fermentation) and nitrous oxide from agricultural soils are, according to the present knowledge, the most important agricultural GHG sources.

Countries are encouraged to improve their methodologies to make the estimates compatible with good practice as adopted by IPCC. This also includes basing the estimates on national information. This is particularly relevant for the agricultural sector where climate and agricultural practices influence the emission level. Such national methodologies and emission factors, however, have to be well-documented, preferably published in refereed scientific journals and according to good practice.

Research is needed to better understand the factors influencing the variation in space and time of the emissions of these sources. This will help to more accurately quantify the emissions and to develop abatement strategies.

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Comparison of national and IPCC default methodologies to estimate methane and nitrous oxide emissions from agriculture

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Summary

All Nordic countries use modified versions of the methodology recommended by the IPCC. These modifications, and their importance for reported emissions of methane (CH_4) and nitrous oxide (N_2O), are summarized in this chapter. Official inventories for 1999 were compared with inventories prepared according to the IPCC default method, and the major differences are discussed. The official inventory for Iceland lacked several sources, and so the IPCC default calculations presented here represent an improved estimate. In comparison with Iceland and Finland, national data have been introduced to a larger extent in Denmark, Norway and Sweden, in some cases with large consequences for total estimates. The combined effect of using the national methodology on CH_4 emissions ranged from -12% to +13%, whereas the range for N_2O emissions ranged from -38% to +10%. National conditions may deviate systematically from the broad categories defined by the IPCC, for example with respect to climatic conditions. This was exemplified by calculations of CH_4 emissions from animal slurry storages using temperature data from seven Nordic locations. Emissions from pig slurry deviated between –35 and +12% from the original estimate, while cattle slurry deviated between –22 and +3%. The deviations were highly correlated with the average annual temperature, indicating that a simple model could lead to improved emission estimates for this source.

Introduction

Although the basis for existing inventories of greenhouse gases in all Nordic countries is the methodology recommended by the IPCC (1997), each country has adopted its own approach to the definition of some activities (sources) and emission factors for methane (CH_4) and nitrous oxide (N_2O). This chapter summarizes these modifications and evaluates their relative importance by comparing the official inventories for the year 1999 with inventories for 1999 calculated according to the IPCC default methodology. The relatively unspecific Tier 1 was used as a reference method, except that some countries have used national data on N excretion rates and manure management for the IPCC default calculations also.

National modifications may clearly improve inventories if better statistical information or empirical data are available, or if agronomic or climatic conditions deviate systematically from the average conditions defined by IPCC for all of

Western Europe. To illustrate this point, emissions of CH_4 from animal slurry storages were calculated using monthly temperatures from seven locations within the Nordic countries.

Data sources

The Danish inventory of greenhouse gas emissions from agriculture in 1999 was published by Fenhann (2001) following a critical review of the methodology (Olesen et al., 2001). The Finnish inventory was taken from a report on trends in Finland's greenhouse gas emissions 1990-1999, and methods of calculation were published by Pipatti (Pipatti, 2001). The Icelandic inventory of greenhouse gas emissions from agriculture has until now been incomplete by not taking several known sources of N_2O into account. The IPCC default data for Iceland presented in this chapter therefore represents a new and improved estimate of N_2O emissions. The 1999 inventory for Iceland was reported to IPCC, but has not been published. The inventory of greenhouse gas emissions from Norwegian agriculture was published as part of the national Norwegian emission inventory, produced by Statistics Norway and the Norwegian Pollution Control Authority (SFT). The methodologies used in the Norwegian emission inventory are described in Flugsrud et al. (2000). The Swedish inventory for 1999 was prepared using the most recent modifications of the national method as described in Sweden's National Inventory Report from 2002.

National methodologies vs. IPCC default method: Overall effects

Table 1 shows CH_4 and N_2O emissions from agriculture in the five Nordic countries in 1999 as calculated by the default method (Tier 1) of the IPCC 1996 Revised Guidelines for National Greenhouse Gas Inventories (IPCC, 1997a). The principles of calculation are described in different chapters of this report. Table 2 shows the emissions for 1999 that were officially reported by each country. This section summarizes overall effects of the national modifications, while subsequent sections about individual sources specify the background for these effects.

The official inventory for Denmark represented a 23.4 kt decrease of CH_4 emissions and a 2.5 kt increase of N_2O emissions compared to the IPCC Tier 1 default method. This corresponded to an overall reduction of 0.18 Mt CO_2 equivalents, or 0.3% of total agricultural emissions in Denmark.

Table 1. Emissions of methane and nitrous oxide in the Nordic countries for 1999 as estimated by the IPCC default methodology (Tier 1).

Compound	Source	Denmark	Finland	Iceland	Norway	Sweden
Methane	Enteric fermentation	143.2	75.4	10.3	82.7	117.6
(kt CH_4)	Manure management	51.6	14.5	0.9	11.4	22.0
	Subtotal	*194.8*	*89.9*	*11.2*	*94.1*	*139.6*
Nitrous oxide	Manure management	2.4	1.3	0.0	0.5	1.9
(kt N_2O)	Mineral fertilizers	4.5	2.9	0.2	1.9	3.2
	Applied animal manure	3.5	1.0	0.1	1.1	1.5
	Nitrogen fixation	0.8	0.01	Not estimated*	0.01	0.1
	Crop residues	6.2	0.5	Not estimated*	4.0	2.4
	Industrial and urban wastes	0.2	0.04	NA	0.03	0.04
	Cultivation of organic soils	0.1	3.8	0.1	1.4	1.9
	Cultivation of mineral soils	NA	NA	NA	NA	NA
	N deposited during grazing	1.0	0.6	0.4	1.8	2.0
	Ammonia volatilization	1.3	0.5	0.1	0.6	0.7
	N leaching	5.2	2.9	0.3	2.9	4.0
	N2O from hayfields. etc.	NA	NA	NA	NA	NA
	Subtotal	*25.2*	*13.5*	*1.2*	*14.1*	*17.8*

* Considered to be negligible; NA: Not applicable.

Table 2. Emissions of methane and nitrous oxide in the Nordic countries for 1999 as estimated by the official national methodologies.

Compound	Source	Denmark	Finland	Iceland	Norway	Sweden
Methane	Enteric fermentation	134.7	74.0	10.3	85.0	143.5
(kt CH4)	Manure management	36.7	10.0	0.9	15.2	14.3
	Subtotal	*171.4*	*84.0*	*11.2*	*100.2*	*157.8*
Nitrous oxide	Manure management	2.4	1.3	Not estimated	Not estimated	1.9
(kt N2O)	Mineral fertilizers	4.9	3.2	0.2	2.0	2.2
	Applied animal manure	3.5	1.2	0.01	0.9	2.4
	Nitrogen fixation	0.8	0.0	Not estimated	0.2	0.1
	Crop residues	6.2	0.6	Not estimated	1.5	1.3
	Industrial and urban wastes	0.3	0.0	NA	Not estimated	Not estimated
	Cultivation of organic soils	0.1	3.8	Not estimated	1.4	1.3
	Cultivation of mineral soils	NA	NA	NA	NA	2.0
	N deposited during grazing	0.9	0.6	Not estimated	0.6	1.5
	Ammonia volatilization	1.2	0.05	Not estimated	0.3	0.1
	N leaching	7.4	1.5	Not estimated	1.4	0.3
	N2O from hayfields, etc.	NA	NA	NA	NA	0.5
	Subtotal	*27.7*	*12.2*	*0.2*	*8.15*	*13.5*

NA: Not applicable.

For Finland, the emissions of CH_4 from agriculture were 89.9 kt if calculated using the IPCC default emission factors, and 84 kt when using the national method. Hence, the official estimate of CH_4 emissions was 5.9 kt lower than indicated by the IPCC default method in the reference year. The official estimate for N_2O emissions was 1.3 kt lower than the IPCC default estimate. These differences together correspond to a reduction of 0.5 Mt CO_2 compared to the IPCC default method. Although not considered in this chapter, cultivation of organic soils is a key source in Finland's inventory. Using national emission factors and soil classification increased the CO_2 emission estimate compared to the IPCC default method by 65% (data not shown).

Methane emissions for Iceland were already calculated according the IPCC Tier 1 default method and so were unchanged in the comparison. For N_2O, the officially reported emission in 1999 was 0.2 kt and thus several times lower than the 1.1 kt N_2O estimated by the IPCC default method. In terms of CO_2 equivalents, the official inventory was 0.31 Mt lower than the IPCC default estimate.

The official 1999 inventory for Norway did not include emissions of N_2O from manure management or from field application of industrial and urban wastes. The official inventory estimate increased total Norwegian CH_4 emissions by 6.04 kt and decreased total N_2O emissions by 5.9 kt compared to the IPCC default method. Using the national methodology thus decreased the Norwegian emission estimate by 1.71 Mt CO_2 equivalents relative to the IPCC default method.

The official Swedish inventory included some sources of N_2O (cultivation of mineral soils, hayfields) which are not considered by IPCC. Industrial and urban wastes were not accounted for, and reindeer were excluded from both estimates. The total effect of using the national methodology instead of the IPCC default method was to increase CH_4 emissions by 18.2 kt, while N_2O emissions were reduced by 4.3 kt. Altogether these deviations represent a reduction corresponding to 0.95 Mt CO_2 equivalents.

The relative differences between nationally reported emissions of, e.g., CH_4 from a given source i ($CH_{4\ nat,i}$) and the emission as calculated by the IPCC default method ($CH_{4\ IPCC,\ i}$) were calculated using the total emissions of that gas, as determined by the IPCC default method, as reference:

$$\%\ difference = \frac{CH_{4nat,i} - CH_{4IPCC,i}}{\sum_{i=1}^{i=n} CH_{4IPCC}} \times 100\%$$

For CH_4, the adoption of national methodologies did not dramatically change the emission estimates, the differences ranging from –12 to +13% (see Table 3). In contrast, the national approaches to calculating N_2O emissions had a significant impact on emission estimates for Norway and Sweden which were, respectively, 38 and 24% lower than the IPCC default estimates.

Table 3. The effect of national methodologies on the emission from individual sources are presented as percentual deviations from total emissions of that compound as calculated by the IPCC default methodology (Table 1; see formula in text).

Compound	Source	Denmark	Finland	Iceland	Norway	Sweden
Methane	Enteric fermentation	-4.4	-1.6	0.0	2.4	18.6
(kt CH4)	Manure management	-7.6	-5.0	0.0	4.0	-5.5
	Subtotal	*-12.0*	*-6.6*	*0.0*	*6.4*	*13.0*
Nitrous oxide	Manure management	0.0	0.0	NA	NA	0.0
(kt N2O)	Mineral fertilizers	1.5	2.2	0.0	0.9	-5.6
	Applied animal manure	0.0	1.5	-7.3	-1.6	5.0
	Nitrogen fixation	0.0	0.0	NA	1.1	-0.1
	Crop residues	0.0	0.6	NA	-17.2	-6.1
	Industrial and urban wastes	0.3	0.0	NA	NA	NA
	Cultivation of organic soils	0.0	0.0	NA	0.0	-3.8
	Cultivation of mineral soils	NA	NA	NA	NA	11.0
	N deposited during grazing	-0.3	0.0	NA	-8.4	-2.9
	Ammonia volatilization	-0.3	-3.5	NA	-2.2	-3.4
	N leaching	8.7	-10.9	NA	-10.6	-20.9
	N2O from hayfields, etc.	NA	NA	NA	NA	2.9
	Subtotal	*10.0*	*-9.9*	*-7.3*	*-38.1*	*-23.9*

NA: Not applicable.

National methodologies vs. IPCC default method: Individual sources

Methane from enteric fermentation

In the official inventory, Denmark used the IPCC Tier 2 method for cattle, and the IPCC Tier 1 method for other animal categories. This decreased total CH_4 emissions by 4.4% compared to the IPCC default method (see Table 3). Finland used IPCC Tier 2 for cattle and Tier 1 for all other animal categories. This resulted in a small reduction in the CH_4 emissions estimate in comparison with the default Tier 1 method. Norway used the IPCC Tier 1 method throughout, but included also ostrich and domesticated deer and reindeer. Emission factors for these animal categories were estimated from emission factors for horses, cattle and goats/sheep, respectively, by scaling according to average body weight. Including these three animal groups increased CH_4 emissions from enteric fermentation for Norway by 2.4% compared with the IPCC Tier 1 default method. For Sweden, emission factors for cattle were based on a national methodology similar to IPCC Tier 2 (Swedish EPA, 1992), while other animal categories were treated according to IPCC Tier 1. The national method (reindeer excluded) gave an 18.6% higher estimate than the IPCC default method, mainly due to higher CH_4 production rates for dairy cattle and beef cows than proposed by IPCC.

Methane from manure management

The major part of CH_4 emissions from manure management in Denmark comes from pigs. Using the Tier 2 method for the official inventory had considerable influence on emission factors for cattle and pigs. For cattle the total effect on CH_4 emissions was limited, while a much lower emission factor for the category ´Other pigs' gave a reduction in total CH_4 emission from manure management of 7.6% compared to the IPCC default method. With both methods, biogas plants reduced total emissions from manure management by 0.9%. In Finland, IPCC Tier 2 was used, which reduced total CH_4 emissions by 5% relative to Tier 1 (Tab. 3). In Norway, cattle are the most important source of CH_4 emissions from manure management. The IPCC Tier 2 method was used to calculate emissions in the official inventory, but emission factors were estimated jointly by Statistics Norway and the Agricultural University of Norway[2]. This increased total emissions of CH_4 by 4%. The official Swedish estimate of CH_4 from manure management, including manure deposited on pasture, was 40% lower than that of the IPCC default method. The difference was due to the use of the IPCC Tier 2 method, and by use of national values for manure production, manure management systems and housing periods. Lower national estimates of manure production partly explained the difference, but the most important factor was a greater fraction of manure management systems with low CH_4 emission potentials (solid manure and daily spread). Relative to the total CH_4 emission estimate of the IPCC default method, the overall effect was a 5.5% reduction.

Nitrous oxide from manure management

In Denmark, the amount of manure N produced was calculated from official norms for the amounts and composition of excreta from the different animal categories and manure management systems. These norm values were also used for the IPCC default method, i.e., default N production values proposed by the IPCC were not adopted. In the official inventory, N_2O emissions were calculated without correction for NH_3 volatilization, as in the IPCC default method, and the IPCC default emission factors[3] of 0.1% for liquid manure and 2% for solid manure were used. Consequently, there was no difference between the IPCC default method and the official inventory for Denmark. In the official inventory for Finland, N_2O emissions from this source were calculated according to IPCC default method. In Iceland, this source was not taken into account in the official inventory. There is limited knowledge about the composition of excreta or the amounts handled by

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[2] Institute of Chemistry and Biotechnology, Section for Microbiology.

[3] Fraction of total N lost as N_2O.

the different manure management systems. With the IPCC default method, the Tier 1 approach was therefore used, resulting in emissions of 0.04 Gg N_2O (Table 1). In the Norwegian inventory, emissions of N_2O prior to field application were not taken into account (Aakra & Bleken 1997). This lowered the total estimate of N_2O emissions by 3.8% relative to the IPCC default methodology (see Tab. 3). In the future, this source of N_2O will be included according to the IPCC guidelines, but with Norwegian factors for N excretion from the different animal categories. In Sweden, national data on N excretion and manure management systems were used as input in both calculations. Hence, the methods resulted in identical estimates for this source.

Nitrous oxide from mineral fertilizers

The official Danish inventory used the IPCC default N_2O emission factor of 1.25% for nitrogen applied as synthetic fertilizers. Still, the total emission was slightly higher than with the IPCC default method due to a lower estimate of NH_3 volatilization (see below). Finland also used the IPCC default emission factor of 1.25% for mineral fertilizer N, and again the difference in Tab. 3 was due to a lower estimate for NH_3 volatilisation. The official Icelandic inventory did not correct for NH_3 volatilization, and the emission factor used was 1%. However, using the IPCC default method by taking NH_3 losses into account and using an emission factor of 1.25% resulted in the same N_2O emission from mineral fertilizers. Norway also used the IPCC default emission factor of 1.25% for this source, and a national estimate of NH_3 volatilization which is based on type of fertilizer used. Like for Denmark and Finland, this approach increased N_2O emissions from mineral fertilizers slightly compared to the IPCC default method. In Sweden, the official inventory for 1999 used a national estimate of NH_3 volatilization that was lower that the IPCC default value, leaving more N for direct emissions of N_2O. However, a national emission factor of only 0.8% was used which worked in the opposite direction. The overall result was that the official estimate of N_2O emissions from mineral fertilizers was 5.6% below the IPCC default estimate.

Nitrous oxide from applied animal manure

The Danish inventory for 1999 used the IPCC default value for NH_3 volatilization of 20%, as well as the IPCC default emission factor for N_2O of 1.25%, so there was no difference between methods for this sector. Finland's official inventory used a lower estimate for NH_3 volatilisation from manure (3%) compared to the IPCC default method, thus leading to higher direct emissions of N_2O from manure application. The IPCC emission factor 1.25% was applied. In Iceland, the official inventory for 1999 used national data on the amount and N content of manure from each animal type, but manure management system was not considered, and an emission factor of 0.1% was used. Emissions calculated according to the IPCC default method were ten times higher, but probably overestimated since domestic animals in Iceland are generally smaller than in other parts of Europe. In the official Norwegian inventory, the amount of manure was estimated from animal numbers (Agricultural Statistics) and national excretion factors for each animal category. This reduced the amount of manure used as fertilizer to 56.4 kt N as opposed to 82.6 kt N using the IPCC default method. There is considerable uncertainty connected to the allocation of manure between what is used as fertilizer and excreted during grazing, respectively. The proportion excreted during grazing was estimated for 1994 (Aakra & Bleken 1997), and this proportion is used every year. Emissions of N_2O from field-applied manure were calculated using the IPCC emission factor of 1.25%. The official inventory corrected for NH_3 volatilization during manure application using values from Statistics Norway's ammonia model. The N_2O emissions from applied animal manure corresponded to a 1.6% reduction in total N_2O emissions compared with the IPCC default method. The official inventory for Sweden had a higher estimate of NH_3 volatilization from applied manure than the IPCC default method, which would reduce direct emissions of N_2O from this source. However, the national emission factor was higher, 2.5 as opposed to 1.25%, and this worked in the opposite direction, resulting in N_2O emissions estimates that were quite similar. Since national values on N excretion were used in both methods, the national estimate of this source increased total emissions according to the default method by 5% (Table 3).

Nitrous oxide from N fixation

For this source, the official Danish inventory used a national estimate of symbiotic N fixation and the default IPCC emission factor for N_2O of 1.25%. This estimate also included N fixation in clover and grass-clover, whereas the IPCC default method includes only N fixation by pulses[4]. Finland used the IPCC default method for the official inventory. In Iceland this source of N_2O is considered negligible and was thus not estimated. The official estimate of biological N fixation in Norway was around 8 kt N per year (Aakra & Bleken 1997), with clover as the most important N-fixing crop. In contrast, the IPCC default method refers to production data from FAO, which means that for Norway green beans and peas are the only N-fixing crops included, corresponding to an N fixation of around 0.3 kt N in 1999. In both cases the emission factor of 1.25% was used. This national ap-

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[4] The report *Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories* (IPCC, 2001) also includes N fixation by forage crops.

proach resulted in a 1.1% higher total N_2O emission in the official inventory compared to the IPCC default method. Sweden's official inventory used national values of N fixation by peas and beans as a fraction of crop product (Växtnäringsbalans i jordbruket, 1996). An emission factor of 1.25% was used in both cases. The N_2O emissions estimated by the national method and by the IPCC default method were rather similar, and the effect on total N_2O emissions was insignificant. Sweden also included a separate contribution from hay fields in the official inventory that increased the total N_2O emission by 2.9% relative to the IPCC default method. The N-fixation was calculated county-wise with a computer programme (NPK-FLO) using area and yield of different grassland types as in-data.

Nitrous oxide from crop residues

The emission of N_2O from crop residues was the second largest in the official Danish inventory, and the most uncertain. The IPCC default method was used in that the amount of N in crop residues was assumed to be identical to the N content of harvested crops. The N content of crops was taken from Grant et al. (1998). Finland used the IPCC default values for different crop parameters for the official inventory, but some additional crop types were included and national estimates for these crops (mixed cereals, turnip rape and clover seed) were used. The result was a minor increase of total N_2O emissions from this source compared to the IPCC default method. Iceland considered this source of N_2O to be negligible and thus it has not been estimated. In the official inventory for Norway, N_2O emissions associated with crop residue decomposition were calculated from N in harvested crops, as recommended by the IPCC default method, but using national estimates of crop yields. The national estimate of crop production was 61% lower than indicated by FAO data due to differences in crop amounts and N contents, i.e., 78 kt N as opposed to an IPCC default estimate of 201 kt N. This gave an official estimate of N_2O from crop residues which reduced total N_2O emissions by 17% (Tab. 3). The official inventory for Sweden used national information about cultivated area and crop yields along with IPCC default values for dry matter content, where available, and a N_2O emission factor of 1.25% for N in crop residues. The national method accounted for residues removed, in contrast to the IPCC default method. No detailed analysis has been carried out to explain the difference between the IPCC default estimate and the national estimate, but probably the main part can be accounted for by differences in the N contents assumed in the IPCC default method (3% for N fixing crops and 1.5 % for other crops) compared with generally lower national values. The official estimate of N_2O from this source was only half the level of the IPCC default method, and using this lower emission factor the estimate of total N_2O emissions from Swedish agriculture was lowered by 6.1%.

Nitrous oxide from industrial and urban wastes

Denmark applied the IPCC default emission factor for N_2O of 1.25% for this source. Only N in sewage sludge was included with the IPCC default method, not N in industrial waste applied to fields. In Finland, industrial wastes were not taken into account in the official inventory, and so it followed the IPCC default method in this respect. For urban wastes the IPCC default of 1.25% was used. In Iceland there is no field application of organic wastes. The official Norwegian inventory presently does not account for N_2O emissions arising from the application of industrial and urban wastes on agricultural fields, and little information is available about the amount of industrial waste applied as a nutrient source. Nitrous oxide emissions derived from field-applied sewage sludge was calculated with the IPCC default method, using the IPCC default emission factor of 1.25%. The omission of this source lowered the total N_2O emission from Norwegian agriculture, as estimated by the IPCC default method, by 0.2%. Similarly, the official inventory for Sweden does not include the application of wastes on agricultural fields at present. According to official statistics, this N source amounted to about 2050 tonnes in 1999 which, estimated by the IPCC-default method, should give an emission of approximately 40 t N_2O.

Nitrous oxide from cultivation of organic soils

Organic soils are a minor source of N_2O in the Danish inventory; the IPCC default value for temperate (and cold) climates of 5 kg N_2O-N/ha was used. In Finland, the revised IPCC default of 8 kg N_2O/ha/yr was used in the official inventory. For Iceland, this is a new item in the inventory, and the IPCC emission factor of 5 kg N_2O-N /ha/year was used. The official 1999 inventory for Norway used the IPCC default emission factor of 5 kg N_2O-N/ha per year, so there was no difference between the official inventory and the IPCC default method. Sweden differentiated between annual and perennial crops, whereby organic soils cultivated with ley were considered to give a lower emission than annual crops. The official inventory estimate of N_2O from cultivated organic soils was lower than the IPCC default method, corresponding to 3.8% of total N_2O emissions. Sweden also included background emissions from cultivation of mineral soils in order to account for long-term accumulation of N in agricultural soil, in contrast to the IPCC methodology. Nitrous oxide from this source corresponded to 11% of total N_2O emissions as calculated by the default method.

Nitrous oxide from N deposited during grazing

The IPCC default method does not take NH_3 volatilization from excreta deposited during grazing into account. In contrast, the official Danish inventory for 1999 assumes NH_3 volatilization to constitute 7% of N excreted, thereby reducing the direct emission from deposited N. The IPCC default N_2O emission factor of 2.0% was used, and the difference between inventories was minor. The official inventory for Finland calculated N_2O emissions according to the IPCC default method. In Iceland, this source was not considered in the official inventory. When recalculated using the IPCC default method, N_2O from N deposited during grazing turned out to be an important source (Table 1). In Norway, N_2O emissions from animals on pastures were calculated using the IPCC default emission factor of 2%. Ammonia volatilisation from excreta deposited during grazing was accounted for using Statistics Norway's ammonia model, which is one explanation for the lower estimate compared to the IPCC default method. However, the main reason for the difference between the official estimate and the IPCC default method was that the national estimate of N deposited on pastures was only a third of the IPCC default estimate. Firstly, the IPCC estimate of total N in animal excreta was 83% higher than the Norwegian estimate, and the fraction deposited during grazing was set at 40 and 25% respectively, in the IPCC default method and the national method. The N_2O emission from this source reduced total N_2O emissions by 8.4% compared to the IPCC default method. The Swedish national methodology for estimating N_2O from grazing animals accounted for loss of NH_3 using an NH_3 emission factor of 8%. The official inventory used national data on N excretion by grazing animals that were generally lower than the IPCC default values, as well as a lower emission factor of 1%. The total effect of using the national method was a reduction of N_2O emissions corresponding to 2.9% of total N_2O emissions.

Nitrous oxide from ammonia volatilization

According to the IPCC default method, 10% of N in synthetic fertilizers, and 20% of the total N excretion by animals in a country is volatilized as NH_3. The N_2O emission factor for NH_3 upon deposition is 1%. In the official inventory for Denmark, NH_3 losses from mineral fertilizers were estimated at 2% and thus lower than the IPCC default estimate. For field-applied manure the IPCC default estimate of 20% was used. The Danish inventory also included area-based NH_3 volatilization factors for grasslands, normal crops and organically managed crops, as well as a contribution from ammonia leaching from straw. The combined effect of these differences was a slightly lower indirect N_2O emission derived from NH_3 volatilization. Finland set NH_3 emissions for mineral fertilizers and manure at 0.6 and 3%, respectively, and thus considerably lower than the default values. The resulting N_2O emission was ten-fold lower than the IPCC estimate, and this approach reduced total N_2O emissions by 3.5%. Iceland did not include this source in the official inventory for 1999, and so the adoption of the IPCC default method (Table 1) represents an improvement of the inventory for Iceland. In the Norwegian inventory, the amount of NH_3 volatilization from agriculture was calculated by Statistics Norway's ammonia model. For synthetic fertilizers, the fraction of N volatilized was 4.5% in the official inventory as opposed to 10% with the IPCC default method. Also for manure the national estimate was only half of the IPCC default estimate; the fraction volatilized from field-applied manure was 23% in the Norwegian model, and for manure deposited during grazing it was 2.79%. The IPCC default emission factor of 1% was used to calculate the amount of N_2O emitted from volatilized NH_3 upon deposition. For Sweden, the national estimate of N_2O derived from NH_3 volatilization was only about 14% of the estimate obtained by the IPCC default method. Although there were differences also in the activity data, the main reason for this deviation was the adoption of an N_2O emission factor of 0.2% in the official inventory as opposed to 1% in the IPCC default method, since most of the ammonia is deposited on acid forest soils. The resulting difference corresponded to a decrease of total N_2O emissions of 3.4%.

Nitrous oxide from N leaching

The IPCC default estimate of N leaching is 30%, i.e., 0.3 kg N is leached for each kg fertilizer N or manure N applied. The default emission factor is 2.5%. The official inventory for Denmark used an N leaching estimate based on modelled N balances, which was ca. 40% higher than the IPCC default estimate, and the IPCC default emission factor. The official Danish estimate of N_2O derived from leached N corresponded to an 8.7% increase of total emissions as calculated by the default method. In Finland, the fraction of leached N was estimated to be lower (15%) than the IPCC default of 30% (Rekolainen et al., 1995; Pipatti, 2001). Accordingly, the official estimate of indirect N_2O emissions from N leaching was only half of that obtained using the IPCC default method. The reduction corresponded to 10.9% of total N_2O emissions (see Table 3). Iceland did not consider N leaching in the official inventory for 1999, so the IPCC default estimate (Table 1) represents an improvement of the inventory for Iceland. In the official Norwegian inventory, an N leaching factor of 18% (Jordforsk, 1998) was used, together with the IPCC default emission factor of 2.5%. The Norwegian estimate of the amount of N leached was 51% lower than the estimate given by the IPCC default value, corresponding to a 10.6% reduction of total N_2O emissions. The official Swedish method gave an estimate of N_2O emissions derived from N leaching which was only 8% of the estimate given by the IPCC default method. This was explained partly by a somewhat lower estimate of N leaching (about 2/3 of the IPCC default value), but mainly by a lower national emission factor of only 0.25%. The approach taken for this source gave a 21% reduction of total N_2O emissions.

Effects of climate – a case study

The default methodology proposed by the IPCC is relatively crude in that it bulks all of Western Europe into one category with respect to emission factors, agricultural production characteristics, and climate. Current knowledge about the regulation of biological processes may not justify a further differentiation of emission factors, as discussed in the chapters by Bertilsson and Lægreid & Aastveit (but see also the chapter by Kasimir Klemedtsson & Klemedtsson). In contrast, information about animal production (e.g., feeding, age structure, housing types, manure storage conditions) and arable production (e.g., fertilizer types, manure application practices, crop residue quality), as well as climatic conditions, are relatively well-defined and might be differentiated to regional or country level in a common methodology.

This section exemplifies how the failure to take climatic gradients into account may introduce a bias in greenhouse gas emission inventories when prepared according to the current IPCC default methodology. The source investigated was CH_4 emissions from liquid manure (slurry) storages, and calculations were made for both cattle and pig slurry. A model was used which was developed as part of a case study to investigate the effects of anaerobic digestion on greenhouse gas emissions (cf. Sommer et al., this report). Input data for the model are information about feed composition (fats, protein, carbohydrate) and digestibility, storage time and air temperature. The amount of slurry in the storage tank changes across the year, and slurry temperature as well as methanogenesis is assumed to be closely correlated with the air temperature (Husted, 1994). The model was originally calibrated to give an annual emission of CH_4 that was similar to the IPCC default emission with Danish monthly mean temperatures.

In the present modelling exercise, the Danish input data on manure composition etc. were maintained, and only temperature conditions were varied. Average monthly temperatures were obtained from seven sites within the Nordic countries which span 11 degrees latitude and 45 degrees longitude (Table 4).

The resulting annual emissions of CH_4 are presented in Table 5, expressed as g CH_4 per kg volatile solids excreted. Emissions from stored pig slurry were approximately twice as high as emissions from cattle slurry, in accordance with the higher efficiency of ruminant digestion. Taking the Danish site at Ødum as reference, the emissions calculated for the other site deviated between –33.2 and

Table 4. Monthly temperatures (°C) at seven sites. The data were used for modelling CH_4 emissions from liquid manure storages.

Month	Ødum	Oslo	Trondheim	Bodø	Visby	Jokioinen	Reykjavik
	56°18′N, 10°08′E	59°56′N, 10°44′E	63°25′N, 10°27′E	67°17′N, 14°25′E	57°40′N, 18°21′E	60°49′N, 23°29′E	64°08′N, 21°56′W
January	-0.6	-4.7	-3.4	-2.1	-0.9	-7.2	-0.4
February	-0.6	-4.0	-2.9	-2.4	-1.4	-7.8	-0.1
March	1.7	-0.5	-0.7	-1.0	-0.2	-4.6	1.5
April	5.6	4.8	3.2	2.2	4.3	2.2	3.1
May	10.7	10.7	7.9	6.2	8.9	8.8	6.9
June	14.2	14.7	11.3	9.9	14.0	13.7	9.5
July	15.4	17.3	14.4	13.6	17.0	16.4	11.2
August	15.3	15.9	13.3	12.7	16.6	14.7	10.8
September	12.3	11.3	9.5	9.4	12.9	9.7	8.6
October	8.7	5.9	5.1	5.1	8.3	4.3	4.9
November	4.3	1.1	1.5	1.9	4.2	-0.1	2.6
December	1.1	-2.0	-1.0	-0.1	1.7	-3.5	0.9
Average	7.3	5.9	4.9	4.6	7.1	3.9	5.0
Min	-0.6	-4.7	-3.4	-2.4	-1.4	-7.8	-0.4
Max	15.4	17.3	14.4	13.6	17.0	16.4	11.2

Table 5. Emissions of CH_4 from pig and cattle manure during storage, and deviations from emissions at the Danish site at Ødum.

Site	Pig slurry		Cattle slurry	
	g CH_4 kg^{-1} VS	Deviation (%)	g CH_4 kg^{-1} VS	Deviation (%)
Ødum	37.4	-	18.2	-
Oslo	35.4	-5.4	16.4	-10.1
Trondheim	26.5	-29.1	14.7	-19.3
Bodø	25.0	-33.2	14.5	-20.3
Visby	39.8	6.6	18.3	0.8
Jokioinen	28.9	-22.7	14.0	-22.9
Reykjavik	28.9	-23.3	14.7	-19.1

+6.6% (pig slurry) or between –22.9 and +0.8% (cattle slurry) due to the difference in temperature alone.

The dependency of CH_4 emissions on annual minimum, maximum and average temperature was examined, and annual average temperature clearly gave the best explanation of the variation across sites (r^2 of 0.72 and 0.98 for pig and cattle slurry, respectively; see Figure 1). In contrast, there were no or much weaker correlations with minimum (r^2 of 0.01 and 0.25) and maximum temperatures (r^2 of 0.63 and 0.23).

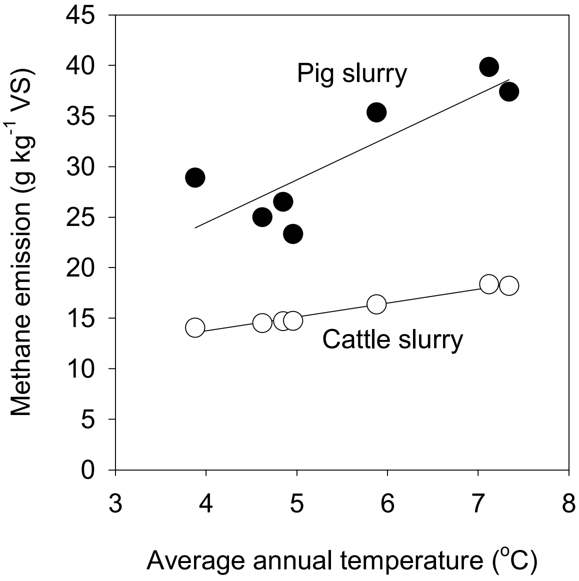

Figure 1. The relationship between average annual temperature at seven sites within the Nordic countries and CH_4 emissions from pig and cattle slurry during storage.

Conclusion

The Nordic countries constitute a relatively uniform region with respect to agricultural production. Yet, the summary of national approaches to greenhouse gas inventories for agriculture presented above shows that there are numerous country-specific approaches to estimation of individual sources. Particularly for N_2O it was found that national methodologies gave considerably lower estimates in several countries. Country-specific approaches complicate the comparison of emission data between countries. On the other hand there is no doubt that significant errors could be introduced if the current IPCC default methodology were strictly adhered to.

Activity data are likely to be improved if based on national surveys and models, rather than on regional default values. In contrast, the biological processes responsible for CH_4 and N_2O emissions are the same in all countries and should be handled in a consistent way with respect to dependency on environmental conditions.

A common methodology has the greatest potential for transparency and comparability, and future work should therefore seek to improve and refine the IPCC methodology further. The modelling exercise of CH_4 emissions from stored slurry clearly demonstrated that there is potential for improvement. Accounting for climatic gradients within the Nordic countries, i.e., the temperature dependency of biological processes, significantly changed the estimate for several locations, and the strong relationships between emissions and average annual temperature indicated that even simple models may improve emission estimates considerably.

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Methane emissions from enteric fermentation – effects of diet composition

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Summary

The formation of methane is an unavoidable result of the digestion of feeds by animals. Ruminants produce considerable amounts of methane. Cattle is the most important category of domestic ruminants in the Nordic countries. Feeding practice has considerable effect on the amount of methane produced, in general more concentrated feeds and a more intensive feeding will give less methane per kg of product (milk, beef). The separation between dairy and beef production might give a higher total methane production. Methane production can be determined experimentally using different methodologies. However, this is difficult and costly, and methane production is therefore most often calculated from knowledge about the animal production and the feeds used. Total amounts of methane produced can be decreased by manipulation of the feeding practice, but most of the methods have unacceptable consequences for environment or animal welfare.

Introduction

Cattle are by far the most important producing animals in the Swedish agricultural sector, which is characterized by a high proportion of dairy cattle within the total cattle population. Nordic dairy cows are highly productive, and the production per animal is among the highest in the world. Although grass and grass products are the basis for dairy production, grains and various by-products have increased in importance. There have been dramatic changes in the productivity of dairy cows over the last century. Today the amount of milk produced by a cow has doubled compared to 50 years ago. This has been promoted both by increases in genetic merits of cows and by changes in feeding practice. These changes have, of course, effects also on the animals' production of methane.

Beef production has up to now to a large extent been based on dairy breed animals. The number of dairy cows have, however, decreased dramatically in the last decades. This fact and also changes in agricultural policies have promoted a more specialised beef production, separate from the dairy production. In Sweden the number of dairy cows decreased from 576,000 to 448,000 between 1990 and 1999. At the same time the number of suckler cows increased from 74,000 to 165,000. As the latter animal category is often kept under different conditions compared to dairy cows, especially concerning feeding and housing, changes in the amount of methane emitted may be anticipated.

Small ruminants (sheep, goats) are few in numbers and do not have much influence on the total budget for methane produced in Northern Europe. Horses and, especially, pigs are high in numbers, but are not regarded to produce much enteric methane.

Calculation of methane production from cattle according to IPCC

According to the IPCC Good Practice guidance (IPCC, 2001), a stepwise decision tree is recommended for the calculation of methane production from livestock within a country. The steps are as follows:

- *Step1*. Divide the livestock into subgroups and characterise these.
- *Step2*. Estimate the emission factors for each of these subgroups.
- *Step3*. Multiply the emission factors from the subgroups with the number of individuals within the subgroup, and sum up over all subgroups to get the total emission.

These three steps can be performed at different levels of detail and complexity. There are two ways to approach these calculations. In the *Tier 1 metho*d (IPCC, 2001), default emission factors for each animal subgroup are used. The values to be used can be found in a tabulated form within the IPCC papers at their web-site. It is good practice to review the Tier 1 emission factors to ensure that the underlying animal characteristics - such as weight, growth rate and milk production - used to develop them are representative for the actual conditions in the country. The data should be reviewed by livestock experts, and if the underlying characteristics are significantly different from actual conditions, the emission factors should be adjusted. The *Tier 2 method* is a more complex model, which requires more specific information from the country in question, and also more specific information on the animal production. This method is recommended when the circumstances that characterise the production in a certain country are not in line with the way the standard values in *Tier 1* has been calculated. As the ways in which cattle are kept in different countries are very different, countries with large livestock populations are recommended to use the *Tier 2* methodology.

In the *Tier 1* method, the cattle are divided into two categories; Dairy cows and other cattle. The latter category includes bulls, calves, growing steers and heifers. Dairy cows are defined as adult cows producing milk in commercial quantities. In some countries it might be appropriate to divide the cows in high and low producers. The emission factor should be chosen to reflect the production circumstances within the country. The tabulated standard values are divided according to regional circumstances. For Western Europe, where high-producing cows dominate, and where feeding is based on high quality grass products and grain, the emission factor proposed is 100 kg methane per cow and year. This figure is based on an average production of 4200 kg milk per cow and year. For the category 'Other cattle', the proposed value is 48 kg methane per animal and year. Calves younger than six months are assumed not to emit any methane. The total emission is calculated by summing emission factors for each animal category multiplied with the number of animals in that category. The emission is given as gigagrams (Gg), which is the same as 1000 tons.

In the *Tier 2* method, the cattle population is divided into more categories. These are adult dairy cows (for commercial milk production), suckler cows (just producing calves), breeding bulls and young cattle. The latter category can be divided into non-weaned calves, growing heifers, steers and bulls. Knowledge is needed on average feed intake in megajoules (MJ) and kg dry matter (DM) in order to calculate the emission factor.

Methane production from cattle

The ability of ruminants to use feed products not directly usable by humans, and to convert it into products (milk, beef etc.) of high nutritional value, is unique and due to their specialized digestive system. The basic cattle feeds are carbohydrate rich feeds, e.g., grass and grain. The digestion of these feeds is, however, linked with a production of methane. The microorganisms in the rumen transform carbohydrates mainly into acetic, propionic and butyric acids. This is especially true when acetate, the dominating volatile fatty acid (VFA) in the rumen, is formed (Lindgren, 1980; Giger-Reverdin et al.,1992; Johnson & Johnson, 1995). Also, when butyric acid is formed there will be an elevated concentration of hydrogen in the rumen. For the animals, the formation of VFAs in the rumen is of vital importance, as VFAs are energy substrates for the ruminant. It is essential to get rid of surplus hydrogen in order to keep these processes going, and the most common process for this is the synthesis of methane from hydrogen and carbon dioxide. This synthesis is performed by methanogenic bacteria.

The reason why researchers in animal science have been interested in determining how methane is created in the metabolic processes of farm animals, is the fact that methane constitutes a considerable part of the energy of feeds and thus an energy loss. In the evaluation of energy in feeds it is necessary to distinguish between the most common ways of expressing feed energy.

- *Gross energy* is the energy possible to gain from a feed through total combustion.
- *Digestible energy* is the part of energy not lost through faeces. Digestibility is normally expressed in percentage. Normal energy digestibilities for cattle are

60-70% for good pastures and good hay and silages and feed diets completed with grain. For intensive production of beef cattle, like feed-lot production in America, a reasonable figure for energy digestibility might be over 75%.

- *Metabolisable energy* is digestible energy minus losses through urine and endogenous gases. These are mainly methane, but also carbon dioxide.
- *Net energy* is metabolisable energy minus energy losses from the animal in its life processes.

Different energy evaluation systems are in use in the Nordic countries. Sweden and Finland use metabolisable energy, although calculated in different ways. Denmark, Norway and Iceland are using net energy systems, but these are also different. This complicates the use of common methods for calculating methane production from ruminants in separate countries.

All factors needed to calculate the different energy expressions can be determined experimentally. It is, however, difficult, and requires specialised equipment. Because of this, emission factors are usually calculated from equations derived from experiments where methane losses have been measured. IPCC (2001) recommends to use 6% of gross energy intake for methane losses when no other figure is available. This figure is generally applicable to dairy cows. For more extensive production the figure 7% is used, while 4% is used for intensive production based on grain. Due to the differences between countries in energy evaluation systems, it is necessary to develop special equations based on the country's system.

Possibilities to influence enteric methane production

Enteric methane production is affected to a large extent by the applied feeding practice. Types of factors that influence enteric methane production from cattle are:

- *Feed intake*. Methane production, expressed as a percentage of feeds, decreases although the total methane production increases as feed intake increases. A common figure for the relative decrease in methane production is 1.6 percent units as feed intake increases by one multiple from maintenance level.
- *Type of carbohydrate*. Cell wall fibers, which are present in high amounts in roughages, give more methane production than digestible fibers such as in by-products from sugar industry, distilleries and breweries. When very high proportions of concentrates are fed (>90% of DM), the methane losses can go down to 2-3% of gross energy.

- *Changing the physical structure* of roughages by milling and pelleting decreases the methane production in the rumen.
- *Feeding fat* normally decreases the methane production in the rumen. This is due to biohydration of unsaturated fatty acids, increased production of propionic acid, and inhibition of protozoans.
- *Manipulating the rumen microflora.* Today this is mainly done by chemical agents (e.g. ionofores), but in the future genetic engineering might be a possibility.

Calculation of enteric methane production

Lindgren (1980) made calculations based on 2500 individual determinations of methane production. The average loss was 11 % of digestible energy intake. Due to large variations within the material, the author recommended that the means should not be used. The methane production is mainly due to the amount of digestible carbohydrates fed, but also to the feeding level. As the amount of digestible carbohydrates fed was not always shown in the literature, Lindgren based his equations on digestible energy. For mixed rations (roughage and concentrates), the following regression was found:

$$\text{Methane (\% of digestible energy)} = 15.7 - 0.030 \times \text{DCE} - 1.4 \times \text{L},$$

where DCE is the digestion coefficient of energy, and L the level of feeding expressed as multiples of the energetic requirement for maintenance. In Sweden, where the basis for formulating feed rations is metabolisable energy (Spörndly, 1999), it is also necessary to calculate the metabolisability of the ration expressed as metabolisable energy in percent of digestible energy. Lindgren (1980) calculated the following regression to do this.

$$\text{Metabolisability (\% of energy digested)} = 83.2 + 2.53 \times \text{L} - 0.045 \times \text{G} - 0.184 \times \text{CP},$$

where G is the percentage of roughage, and CP that of crude protein. Both are expressed as % of DM. Energy content in methane is needed in order to calculate kg of methane. The energy content is set to be 55.65 MJ kg-1 methane (IPCC, 1997).

There are many equations available in the literature based on detailed information about the feeds and especially the carbohydrates' chemical composition (Lindgren, 1980; Holter & Young, 1992; Benchaar et al., 1998). These equations are difficult to apply when the exact chemical composition is not known, which is often the case in practice. The determination coefficients are also often low (R2 ~ 0.5). Kirchgessner et al. (1991) have presented a simple model to calculate methane production in dairy cows based on milk production and live weight. His equations have reasonable correlations with determined methane production levels.

Methane (g cow^{-1} day^{-1}) = 55 + 4.5 × (kg milk cow^{-1} day^{-1}) + 1.2 × (metabolic weight)
Metabolic weight = $(\text{live weight})^{0.75}$.

Comparison of different methods to calculate enteric methane production
Some calculations concerning methane production from cattle in Sweden are shown in Tables 1 and 2 (Bertilsson, 2001). It is obvious that these models differ, and that methods 1 and 2 give considerably higher values than the IPCC default method. It is also notable that, although the total methane production from dairy cows decreased during the last decade, the total methane production tended to increase. This is due to a fast increase in the number of cattle for beef production. A similar decrease in methane production from dairy cows has been observed in Denmark, and this has been attributed to higher energy use efficiency in the milk production (Olesen et al., 2001b).

Table 1. Methane from Swedish dairy cows calculated according to different methods.

	1990	1999
Methane, kg/animal/year		
Method 1*	125	131
Method 2**	101	107
IPCC default (Tier1)	100	100
Methane, g/kg milk*		
Method 1	20.8	17.8
Method 2	16.9	14.5

*Lindgren, 1990; ** Kirchgessner et al., 1991.

A Danish study recently showed that the estimated methane emission declined with increasing proportion of concentrates in the feed ration (Olesen et al., 2001a). The largest reduction in methane emission was, however, obtained by increasing the content of fat in the ration. The reference feed ration gave approximately the same methane emission as the IPCC standard methane conversion factor. The use of a feed ration with a fat content of 7.2%, against 4.5% in the reference situation, reduced the methane emission by 34%. In CO_2 equivalents this corresponded to a reduction of 433 kt CO_2 equivalents yr^{-1} for Denmark in a scenario for the year 2010.

Table 2. Total methane production from cattle in Sweden (calculations based on Lindgren, 1980; Bertilsson, 2001, and Swedish feed tables).

Category of animals	Animal number		Per cow and year (kg CH_4)		Total amount (gG)	
	1990	1999	1990	1999	1990	1999
Dairy cows	576409	448520	124	130	71.5	58.3
Replacements	464009	361021	68	68	31.6	24.5
Suckler cows	74544	164801	100	100	7.5	16.5
Replacement	29817	66000	68	68	2.0	4.5
Other cattle	258527	404100	80	80	20.7	32.3
Total from cattle					133.3	136.1

Discussion

A more intensive dairy production will decrease the methane production per kg of milk produced (Martin & Seeland, 1999). It is doubtful whether public acceptance of an intensified production can be found. Organic production is politically and socially acceptable, but the higher proportion of roughages may increase the methane emission, and this may lead to negative environmental impacts. Use of chemical properties to decrease methane production would also be challenged by the consumers for the same reason as above, and so would the use of genetically modified organisms. Decreasing the number of animals is of course one way to deal with the problem. This would, however, have great effects on the cultural landscape in countries like Sweden, where arable land occupy less than 10% of the total land area. There are many indications that the specialisation of dairy production in total has a negative effect on the environment. A combined milk and beef production also in the future would probably be the best compromise here.

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Methane emissions from livestock manure – effects of storage conditions and climate

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Summary

Livestock manure contributes significantly to the global emission of methane (CH_4). Methane is emitted during storage of both liquid and solid manure. Part of the solid manure is produced in loose housing systems with solid floors where the manure is stored in a deep litter mat, which is a mixture of straw, urine and faeces. As anaerobic conditions are found in the lower part of the deep litter mat, significant amounts of the carbon stored in the deep litter may be emitted as CH_4. It has been estimated that a cattle deep litter mat contributed 11 to 18% of the total CH_4 (from cattle digestion and litter) emitted. This source of CH_4 does not seem to be included in the IPCC default value for solid manure. During outdoor storage of solid manure, CH_4 can be produced at a high rate in the central parts of the heap. Methane emissions have been shown to account for 0.01 to 0.2% of the total carbon content, and emissions were positively related to the bulk density of stored solid manure. Methane may be partly transformed to CO_2 during the transport from the inside of a heap towards the surface.

The emission of CH_4 from stored anaerobically digested slurry and cattle slurry has been shown to vary between <0.01 and 1.4 g C m^{-3} h^{-1}. Methane is produced in the bulk of the slurry, and it has been found that log transformed CH_4 emissions decrease linearly with the inverse temperature of the slurry. A porous surface cover on the stored liquid manure may reduce CH_4 emissions by up to 40%, probably due to CH_4 oxidation within the surface cover or at the interface between the cover and liquid in the store. The establishment of a porous cover of slurry stores could be introduced as a mitigation technique and could also be included in the IPCC guidelines for calculating CH_4 emissions from animal manure.

Introduction

The composition of animal manure varies widely between animal species and housing systems. Slurry collected below slatted floors has a low content of dry matter due to limited use of bedding materials. Slurry, therefore, mainly consists of faeces and urine. In housing systems where livestock are tied, the excretion is separated into solid and liquid manure. The solid fraction, which is usually called farmyard manure (FYM), consists of faeces, litter and some urine, and the liquid manure consists of urine and some faeces. In loose housing systems with solid floors that are strewed with straw, sawdust etc., the solid manure consists of a mixture of faeces, urine and organic strewing material, i.e., deep litter.

Slurry may be stored from one to several months inside animal houses, and for up to one year in outdoor stores. FYM is traditionally removed daily from the houses to outdoor stores, while liquid manure drains continuously through gutters to outdoor liquid manure stores. Deep litter, developing on the floors of animal houses for several months and up to about one year before it is removed, is often stored outside the animal house in manure heaps before it is applied in the field. In Denmark, approximately 80% of all the livestock manure is handled as slurry and 20% as solid manure (Poulsen et al., 2001). At present, deep litter constitutes 13% and farmyard manure 7% of the manure; however, for animal welfare reasons there is an increasing interest in loose housing systems, and these systems are expected to contribute to an increasing production of deep litter.

Livestock production contributes significantly to the increase in atmospheric methane concentration, and it has been estimated that livestock manure accounts for between 5 and 6% of the global emission of atmospheric CH4 (Hogan et al., 1991; Rotmans et al., 1992). The Danish agricultural emission of CH4 amounts to 430 kt yr-1, of which 172 kt are emitted during collection and storage of manure, and from manure applied to the soil (Petersen & Sommer, 1999). However, the emission inventories are based on a limited number of data with respect to animal manure. Therefore, for the purpose of improving the calculation of emissions and developing abatement techniques, there is a need for more knowledge about how CH4 emissions are related to the handling and type of livestock manure.

This paper describes how manure management affects emission of CH4 from livestock manure during storage inside and outside animal houses, and it discusses how this information could contribute to improve the IPCC procedures for calculating greenhouse gas emissions.

Methane emissions from solid manure

Emissions from deep litter mats

The microbial activity has proved to be very significant in the surface layers of cattle deep litter mats (Henriksen et al., 2000). The microbial activity at the surface will reduce the oxygen content of air entering the mat. Henriksen et al. (2000) found that the oxygen content in the air beneath a depth of 10-15 cm was very low, and anaerobic conditions were found in the bottom layers of the deep litter (Fig. 1). Furthermore, a combination of insulation and aerobic microbial activity caused the temperature to increase to 40-50°C at about 10-15 cm from the surface. Below this layer the temperature declined due to lower activity in the anaerobic environment.

Laboratory studies have shown that more than 80% of the total transformations of carbon take place in the aerobic 0-20 cm top layer (Henriksen et al., 2000).

From this layer, carbon was emitted in the form of carbon dioxide (CO_2). From below 15-20 cm, about 20% of the carbon transformations of the deep litter resulted in CH_4 and CO_2 production, with 5-15% of total carbon gas emissions occurring in the form of methane (Fig. 2) (Henriksen et al., 2000). No methane oxidation was observed in the aerobic top layer, therefore the suggestion that CH_4 may be oxidised during transport from bottom layers of the deep litter to the surface (Rom et al., 2001) was not confirmed. Measurements in animal houses showed that during a period of three months the daily CH_4 emission constituted 30-70 g C ton^{-1} manure, which corresponded to *ca.* 15% of the total CH_4 and CO_2 emission from the deep litter (Rom et al., 2000). The total CH_4 emission from cattle and deep litter made up *ca.* 5% of the total carbon supplied to the cattle in feed and litter, which is close to values estimated for slurry based housing systems (Jungbluth et al., 2001). The CH_4 emitted from the deep litter mat accounted for between 11 and 18% of the total CH_4 emission from cattle housing (Rom et al., 2000).

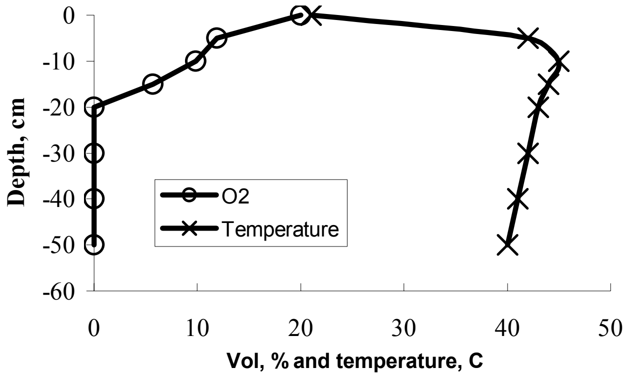

Figure 1. Profiles of temperature and oxygen concentration in a cattle deep litter mat.

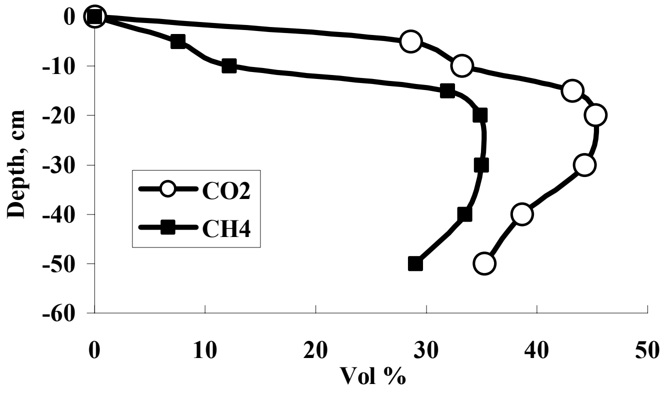

Figure 2. Profiles of concentrations of methane and carbon dioxide in a cattle deep litter mat.

Increasing the population density of animals, and thereby the excretion and compaction of the deep litter, may increase the production of CH_4. The findings above refer to a cattle deep litter mat. The methane emission from pig deep litter mats may differ significantly from these findings as pigs mix the deep litter via their behaviour and due to their sharp cloves. Therefore, a lower CH_4 emission is expected from pig deep litter mats than from cattle deep litter mats.

Solid manure heaps

When a manure heap is established, the temperature inside the heap may increase to 70oC due to aerobic microbial metabolism, i.e. composting (Fig. 3). Composting generates an upward airflow in the heap and, as a consequence, fresh air from the atmosphere will enter through the lower section of the heap.

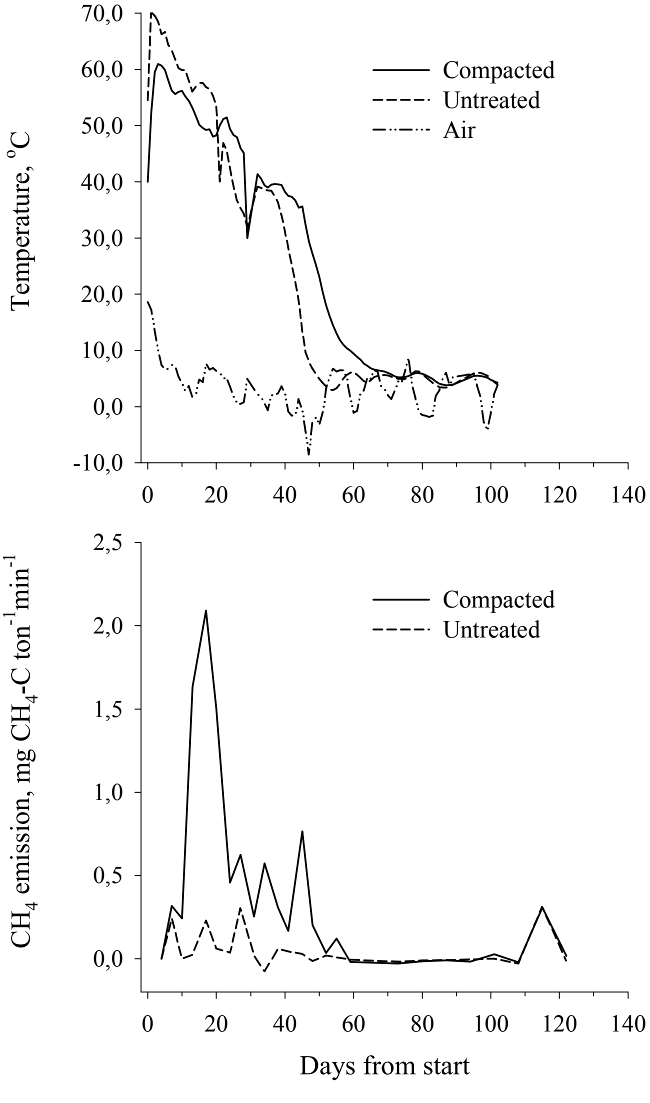

Figure 3. Dynamics of temperature and methane emission during windrow composting of cattle deep litter stored with or without compaction. Compaction was performed by means of a frontloader, which increased the bulk density from 0.42 to 0.49 t m^{-3}.

Most studies show low emissions of CH_4 immediately after the establishment of the heap. However, the CH_4 emission from composting heaps becomes significant after initiation of the phase with very high temperatures, and the emission will remain high for a period of two to four weeks, whereupon it will decline with the decreasing temperature (Hellman et al., 1997; Hellebrand & Kalk, 2000; Sommer, 2001). During the initial phase of vigorous composting, anaerobic sites may emerge due to high oxygen consumption rates. Methane production is strictly anaerobic and increases with increasing temperatures (Zeikus & Winfrey, 1976; Sommer and Møller, 2000), and the combination of anaerobic conditions and high temperatures may therefore contribute to a high emission rate during this phase (Hellman et al., 1997).

Restrictions in the air exchange of the heap will stimulate the development of sites in the heap where oxygen consumption exceeds the oxygen supply. Therefore, the CH_4 emission will be higher from a heap at high bulk density than the emission from a heap at low bulk density (Fig. 3) (Sommer, 2001). Although it was observed by Sommer (2001) that CH_4 emissions were related to CH_4 concentrations in the heap, the emission did not increase significantly until concentrations inside the heap reached *ca.* 500 ppm. It was therefore assumed that CH_4 was oxidized to CO_2 during the transport from the centre to the surface of the heap, and at low concentrations the potential for CH_4 oxidation exceeded CH_4 production rates. In studies by Sommer (2001) and Sommer & Møller (2000), the CH_4 emission accounted for between 0.01 and 0.2% of the initial carbon content of the manure heaps. Much higher values for CH_4 emission from composting manure have been reported by Hellebrand & Kalk (2000). They found that 4.6% of the carbon mineralized was released as methane, corresponding to 1.4% of the initial carbon content.

In order to reduce ammonia emissions from stored solid manure, it has been recommended to reduce the convection of air into and through the heap. The convection may be reduced with a cover of tarpaulin or through compaction of the litter. However, a negative side effect of this practice for reduction of NH_3 emissions could be an increased production of CH_4 as a result of more anaerobic conditions in the heap (Sommer & Møller, 2000; Jungbluth et al., 2001).

Methane emission from slurry

It is well established that the CH_4 emissions from anaerobically stored slurry is related to the temperature of the slurry. Laboratory studies have shown that the CH_4 production in slurry increases with increasing temperatures between 10 and ca. 30°C (Cullimore et al., 1985; Khan et al., 1997), and that below 10°C the methane production is negligible (Steed & Hashimoto, 1994).

In-house storage of slurry

Little is known about the methane production in slurry during storage inside animal houses, but from laboratory studies it is well established that the emission is related to the following factors: temperature, storage time, population of methane producing micro-organisms and content of volatile solids (VS). The temperature in slurry channels is related to the type of housing system. In Denmark the temperature in an insulated housing type is probably about 15°C during winter and 20°C during summer, in a non-insulated housing type the temperature of slurry in the slurry channels will be about 5°C during winter and 20°C during summer. If the animals are housed only during winter, then the emission of CH_4 from slurry channels will be low, as the majority of slurry then is stored during periods of low temperature.

Storage of slurry

There are only few studies of CH_4 emission from field-scale slurry stores, and therefore IPCC has based their calculations on results from laboratory experiments like the study of Steed & Hashimoto (1994). Field scale studies have found that the CH_4 emission varied between <0.01 to 1.4 g C m^{-3} h^{-1} (Husted, 1994; Khan et al., 1997; Sommer et al., 2000). Most of the variation in these studies seemed to be due to temperature variations and thus confirmed the importance of temperature observed in laboratory studies. The methane emission in the three studies quoted was related to temperature by the Arrhenius equation, i.e., the log of methane production decreased linearly with the inverse temperature. However, the parameters of the Arrhenius equation varied from study to study, probably because of variations in slurry composition or in methodology that was not taken into account in the parameterization. Thus, using data from field studies to parameterize the relation between CH_4 emission and temperature were not promising (Fig. 4).

In the study by Sommer et al. (2000), the emission of CH_4 was 40% higher from uncovered slurry than from slurry covered with a layer of straw, while the reduction with Leca nuts or a natural surface crust was intermediate (Fig. 5). The reduction of CH_4 emissions from stored slurry with a surface cover suggests that CH_4 is oxidized to CO_2 during its passage through the porous layer, in accordance with earlier observations (Husted, 1994).

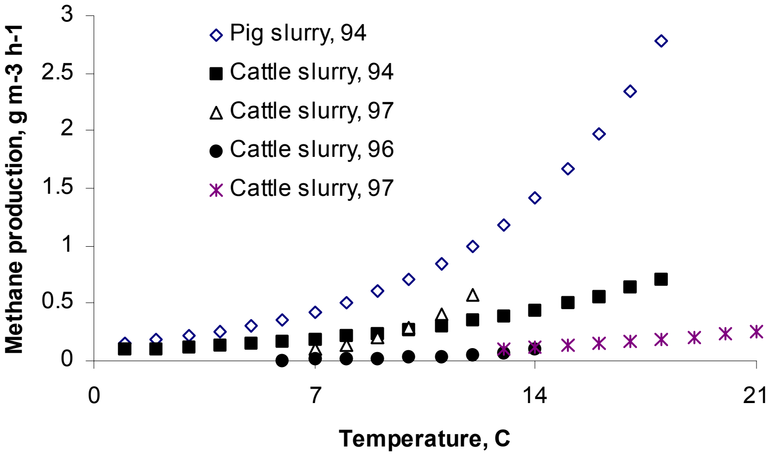

Figure 4. Influence of temperature on methane emission during field-scale storage of slurry observed in different studies (Sommer et al., 2000; Khan et al., 1997; Husted, 1994).

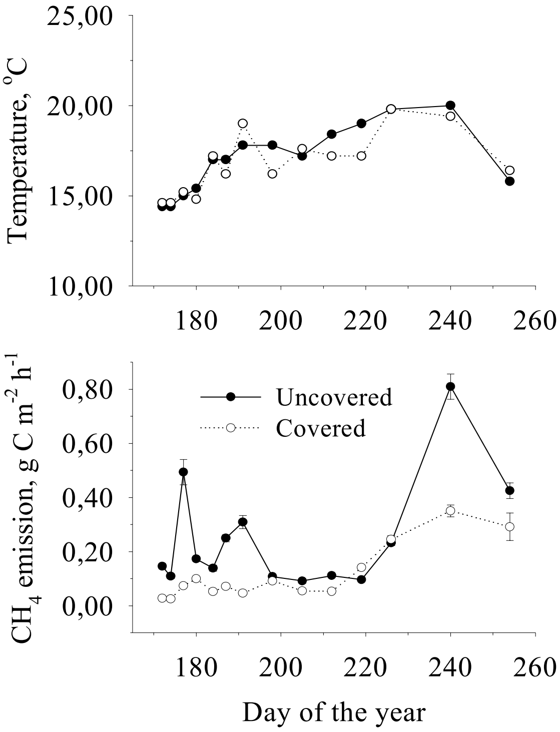

Figure 5. Temperature and methane emission from stored cattle slurry without cover and with a cover of straw.

Algorithms for estimating CH4 emissions

The IPCC methodology calculates CH4 emissions from stored manure as a fraction of the manure VS, and a distinction is made between different manure management systems.

The IPCC methodology does not include emissions from solid manure during storage in animal houses. This is a shortcoming, because emission of CH4 from deep litter mats may account for 11-18% of the methane emissions from cattle housing (Henriksen et al., 2000). Methane is only produced in the bottom layer of deep litter mats. Thus, when including this source in the methodology for calculating greenhouse gas emission, it will be necessary to evaluate whether the emission should be related to the area or the depth of litter mats, and whether type and density of livestock should be included.

Our experience is that CH4 emissions from solid manure stored outdoors in heaps are quite low as predicted by IPCC, but that new handling systems for reduction of NH3 emissions could contribute to increase the methane emission from these stores.

Anaerobic digestion of slurry has been shown to reduce CH4 emissions from stores of animal slurry (Sommer et al., 2000). However, care must be taken to retrieve CH4 for a period after the slurry has left the biogas reactor, where a combination of high temperatures and a large population of methane producing microorganisms enables intense CH4 production to continue. Inventories should also take into account that the CH4 emission from slurry stores may be reduced by a porous surface cover.

Conclusions

Methane is produced in anaerobic volumes of stored slurry and solid manure, and the production is positively related to temperature. Methane emission from slurry stores may be reduced by porous covers of a natural surface crust, straw or Leca nuts. A significant emission of methane has been observed from cattle deep litter mats. Methane emissions from solid manure heaps of low density are small, but may increase if the inflow of air is restricted by compaction or by covering of the heaps.

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A new model for calculating the reduction in greenhouse gas emissions through anaerobic co-digestion of manure and organic waste

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Summary

Biogenic emissions of methane (CH_4) and nitrous oxide (N_2O) occur during handling, storage and after field application of animal manure. The emissions are linked to decomposition of volatile solids (VS), which provide energy for microorganisms.

During anaerobic storage, turnover of VS drives the microbial processes which lead to CH_4 production. Also, turnover of VS in slurry applied to fields will consume oxygen and can thereby stimulate N_2O production. Anaerobic digestion of manure and organic wastes for biogas production removes VS prior to storage and field application, and therefore this treatment also reduces the potential for CH_4 and N_2O emissions.

A model has been developed to evaluate the effect of anaerobic co-digestion of animal manure and organic waste on CH_4 and N_2O emissions. The model estimates the reduction in VS during storage and digestion, and an algorithm for prediction of CH_4 emissions from manure during storage relates the emission to VS, temperature and storage time. Nitrous oxide emissions from field-applied slurry are calculated using VS, slurry N, soil water potential and application method as input variables, thus linking C and N turnover. The amount of fossil fuel that is substituted by CH_4 produced during digestion is also calculated in order to estimate the total effect of anaerobic digestion on greenhouse gas emissions from slurry.

Model calculations show the potential of manure digestion to modify the emission of greenhouse gases from agriculture. The experience from application of the model to different scenarios is that the emission of greenhouse gases and their reduction must be calculated with dynamic and integrated models. Specifically, the results indicate that digestion of slurry and organic wastes could reduce Danish greenhouse gas emissions by as much as 3%.

Introduction

Anthropogenic emissions of the greenhouse gases carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) have increased significantly during the last century. Measures to reduce global warming due to the greenhouse effect tend to focus on CO_2 emissions from combustion of fossil fuels. Relative to CO_2, the amounts of

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[§] This work has also been presented at the Third International Symposium on Non-CO_2 greenhouse Gases (NCGG-3) Scientific understanding, control options and policy aspects. Maastrict, The Netherlands 21-23 January 2002.

CH_4 and N_2O in the atmosphere are low, but their global warming potentials (GWP) are, respectively, 21 and 310 times higher than that of CO_2. Globally, the emission of CH_4 and N_2O from livestock manure contributes 5-6% to the total emission of CH_4 (Hogan et al., 1991; Rotmans et al., 1992) and 7% of N_2O (Khalil & Rasmussen, 1992). Within EU, agriculture is estimated to contribute almost half of the CH_4 emissions and more than half of the N_2O emissions (EEA, 1999). Main sources of CH_4 are animal digestion and manure stores, while N_2O mainly originates from the turnover of mineral fertilizers and field applied animal manure, and from the decomposition of crop residues.

Anaerobic digestion of animal manure and organic waste materials in biogas digesters reduces the level of volatile solids (*VS*). Since *VS* drives the microbial processes that may lead to CH_4 production during anaerobic storage, the removal of *VS* in biogas digesters prior to storage can reduce the potential for CH_4 emissions to the atmosphere. Emissions of N_2O from manure applied to agricultural land can be stimulated in environments with low oxygen availability. Since turnover of *VS* in the manure leads to enhanced oxygen consumption, anaerobic digestion has a potential to reduce N_2O emissions from field-applied slurry (Petersen, 1999).

The IPCC model (IPCC, 1997) for quantifying the effect of anaerobic digestion on greenhouse gas emissions from manure will not fully account for the reduction of VS, partly because the model does not present algorithms to calculate the effect of digestion on CH4 emissions during subsequent storage, and partly because the model does not predict an effect of VS reduction on N2O emissions after field application of manure. Therefore, we developed a model designed to estimate the total reduction in greenhouse gas emissions, which results from co-digestion of animal slurry and organic waste (not including sewage sludge and organic household waste) in biogas plants.

The model

The model uses *VS* as the main driving variable to predict CH_4 and N_2O emissions before and during digestion, during storage, and after field application of untreated and digested manure and waste. The fundamental principle is to estimate the removal of *VS* in slurry and organic waste during anaerobic digestion and storage (Fig. 1).

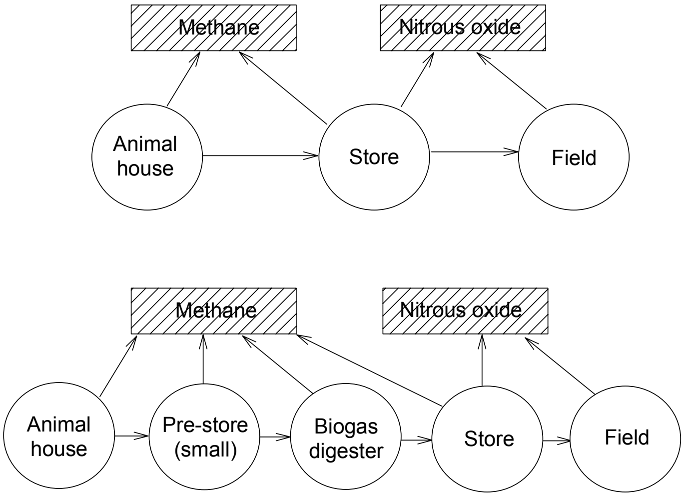

Figure 1. Sources of CH_4 and N_2O in manure management systems without (top) and with (bottom) fermentation of slurry in biogas digesters. Emissions from digesters are due to leaks.

Methane emissions from slurry channels inside animal houses and during storage are related to the content of degradable VS, storage time and temperature. Volatile solids (VS) in slurry are considered to consist of fats, protein and simple, degradable carbohydrates (designated as VS_D) and of non-degradable carbohydrates like lignocellulose (VS_{ND}). The potential CH_4 production per kg VS in cattle and pig slurry is estimated by Bushwell's equation (Symons & Bushwell, 1933), while the production of CH_4 per kg VS actually achieved in digesters is assumed to represent a 90% degradation of VS_D (Angelidaki et al., 2000). Hence, the fraction of VS that is VS_D can be calculated with the following equation:

$$\frac{VS_D}{VS} = \frac{1}{0.9} \times \frac{CH_4(digestor)}{CH_4(potential)} \tag*{Eq. 1}$$

The amount of VS_D contributed by carbohydrate is calculated assuming that all fats and protein, but only a fraction of the carbohydrate, is readily degradable:

$$VS_{D,Carbohydrate} = VS_D - (VS_{D,fats} + VS_{D,protein}) \tag*{Eq. 2}$$

Using measurements of CH_4 production in operating biogas plants and information about the overall composition of slurry, it has been calculated that the different organic species contribute to VS_D as shown in Table 1; $VS_{D,fats}$ and $VS_{D,proteins}$ are identical to VS_{fats} and $VS_{proteins}$, respectively, as we have assumed that these components are 100% degradable.

Table 1. Parameters for calculating CH_4 production and VS_D removal during storage and digestion of animal slurry and organic waste.

Biomass	Degradable			Non-degradable carbohydrate	CH_4 production
	Fat	Protein	Carbohydrate		
			%		kg CH_4/kg VS_D
Cattle slurry	9	18	21	52	0.34
Pig slurry	10	30	25	35	0.36
Organic waste	50	25	5	20	0.52

The degradation of *VS*, and the derived production of CH_4, is calculated with a model which integrates the effect of storage inside and outside the animal house. Factors for CH_4 emission during storage of slurry in animal houses are given in Table 2. At present we have no detailed measurements of temperature variation in slurry stored in-house, and therefore only two temperature regimes, i.e., summer (20°C) and winter (15°C), have been used in the calculations.

Table 2. Emission factors for CH_4 during storage inside animal houses, given in % of VS excreted and (in brackets) in g kg^{-1} VS.

	% of *VS* excreted (g CH_4 kg^{-1} VS excreted)		
	15°C	20°C	Average
Cattle	7 (24.2)	13 (44.9)	10 (34.5)
Pig	3 (10.68)	7 (24.92)	5 (17.8)

The temperature relationship of CH_4 production was calculated with the Arrhenius equation using data from field studies of Husted (1994), Khan et al. (1997) and Sommer et al. (2000):

$$\ln F(T) = VS_D \times b_1 \times (\ln A - E \times (1/RT)) + VS_{ND} \times b_2 \times (\ln A - E \times (1/RT)) \qquad \text{Eq. 3}$$

where *F* is the emission rate (g CH_4 kg^{-1} *VS* h^{-1}), VS_D and VS_{ND} are the amounts of, respectively, degradable and non-degradable *VS* (g kg^{-1} slurry), b_1 and b_2 are rate correcting factors (no dimensions), *A* is the Arrhenius parameter, *E* the activation energy, *R* the gas constant and *T* the temperature (K). The parameters used are given in Table 3. Temperature is related to air temperature and storage time according to a standard scheme for filling and emptying of in-house slurry channels and outdoor stores.

Nitrous oxide emissions from field-applied slurry are estimated on the basis of inputs of NH_4^+, VS_{ND} and VS_D. Three application strategies are defined, for which NH_3 volatilization and soil water potential at the time of application are defined (see Table 4). The model links N_2O emissions to the proportion of turnover expected to occur in oxygen deficient slurry 'hot spots', i.e., where reduction of nitrate diffusing into the hot spots replaces the aerobic decomposition of VS. The VS_{ND} is assumed to remain in the hot spot, while VS_D is partly redistributed with the slurry liquid to the surrounding soil. The VS_D distribution at equilibrium is calculated using relationships between slurry VS and water potential determined under laboratory conditions (Fig. 2). In the model, N_2O from nitrification is related to NH_4^+ in slurry hot spots and bulk soil, respectively, while N_2O from denitrification is a function of VS_D retained in slurry hot spots. Further, the fraction of VS_D degraded via denitrification (as opposed to aerobic processes) is estimated at 10% (Petersen et al., 1996). Nitrous oxide emissions derived from rainfall events are included as an area-based background. According to this model, anaerobic digestion will have little impact on N_2O derived from nitrification, but it will reduce N_2O from denitrification proportionately with VS_D removal.

Table 3. Parameters for calculating CH_4 emissions from cattle and pig slurry using Eq. 3.

Parameters		Cattle	Pig
Arrhenius parameter	ln(A)	22.60	44.00
Activation energy	E	6.3×10^7	1.13×10^8
Gas constant	R	8.314	8.314
Rate correction factor for VS_D	b1	1	1
Rate correction factor for VS_{ND}	b2	0.01	0.01

Table 4. Nitrous oxide emissions from field-applied slurry were estimated on the basis of the conditions and assumptions summarized below.

Input variables (kg ha^{-1})	Total VS, VS_D, Total N, NH_4^+-N		
Application strategies	I.	II.	III.
- time of year (proportion of slurry applied)	spring, incorporated (65%)	spring, surface applied (17%)	summer, surface applied (18%)
- NH_3 volatilization	10%	20%	10%
- soil water potential	pF 2	pF 2	pF 3
Redistribution of VS_D and NH_4^+	Function of slurry VS_{ND}, VS_D and soil water potential		
Limiting factor for estimates of:			
- nitrification	i) NH_4^+ in slurry hot spots; ii) NH_4^+ transported to the soil		
- denitrification	VS_D retained in slurry hot spots; 10% of C is metabolized via denitrification, and this is recalculated into NO_3^- reduced		
Emission factors for:			
- nitrification	i) 0.5%; ii) 0.2%		
- denitrification	2%		
Background emission	0.2 kg N ha^{-1}		

*EF=Emission factor in pct. of nitrification and denitrification.

The model handles organic waste for co-digestion like the slurry, i.e., untreated organic waste is also assumed to be stored and applied to agricultural land. The effect of the VS_D reduction on CH_4 emissions during storage, and on N_2O emissions from field-applied waste, is accounted for in the model. Furthermore, the model accounts for the substitution of fossil fuel. For this comparative study, model parameters have been selected that give emission rates for untreated pig and cattle slurry similar to those of the IPCC model.

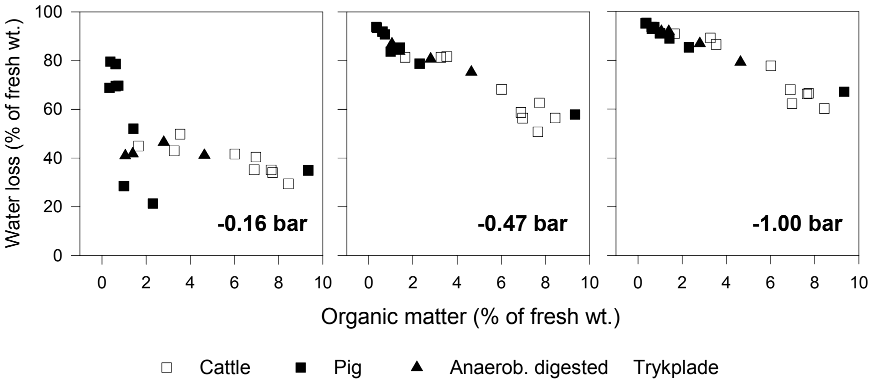

Figure 2. The relationship between slurry organic matter (VS) and water retention at three different water potentials for 9 selected cattle slurries, 9 pig slurries and 4 digested slurries. Water potentials were established with polyethylene glycol, and water was extracted by dialysis (Petersen et al., subm.).

Technology - management

Table 5 describes three sets of on-farm conditions for slurry management, for which greenhouse gas emissions were calculated (see also Fig. 1). They include a situation (Reference) with no treatment of animal slurry and organic waste, a system (Biogas I) in which slurry is digested according to the existing biogas technology, and a system (Biogas II) where the fraction of CH_4 collected for energy production is optimized by reducing the storage time prior to digestion, and by using improved biogas technology. It is assumed that the CH_4 produced will substitute natural gas for energy production.

Calculations for these three management systems indicate that digestion of pig manure can reduce greenhouse gas emissions from 1.4 kg CO_2 eq. kg^{-1} *VS* (Reference), to 0.8 kg CO_2 eq. kg^{-1} *VS* if present-day technology (Biogas I) is used, and to 0.4 kg CO_2 eq. kg^{-1} *VS* if a more efficient technology (Biogas II) were adopted. Digestion of cattle manure reduced emissions of greenhouse gases from 1.3 to 1.0 and 0.2 kg CO_2 eq. kg^{-1} *VS* for, respectively, the Biogas I and Biogas II system. Digestion would reduce emissions derived from the organic waste by about 50% with both technologies.

Table 5. Systems for handling of animal manure on livestock farms with slurry-based housing systems. The slurry is either left untreated (reference), co-fermented with organic waste using the existing biogas production technology (Biogas I), or co-fermented with organic waste using an optimized technology (Biogas II).

	Reference	Biogas I	Biogas II
Cattle slurry stored in house	30 days	30 days	1 day
Pig slurry stored in house	15 days	15 days	1 day
Cattle/pig house	20°C during summer, 15°C during winter		
Organic waste stores emptied	April		
Organic waste, composition	< 20% *VS*, organic waste; household waste and sewage sludge not included		
Methane lost via leakages and from generators		3% of methane production	1.5% of methane production
Post fermentation gas collection		Until the temperature of the fermented slurry is similar to the temperature of the environment	
Slurry store emptied	April, start of growing season		
Temperature of stored slurry	Similar to the monthly average of air temperature		

Scenarios for Denmark

The reduction in total Danish greenhouse gas emissions that may be achieved by anaerobic digestion (Biogas systems I or II) was calculated for the following three scenarios:

- 2000: A scenario where the reduction in greenhouse gas emissions due to the present-day biogas production level is calculated.
- 2012: A scenario where the reduction in greenhouse gas emission due to biogas production in the year 2012 is calculated using the official forecasting from the Danish Energy Agency that the amount of slurry digested will increase seven-fold, and that the amount of organic waste digested will double.
- Long-term: A scenario where the total production of animal slurry and organic waste (excl. household waste and sludge) is digested.

According to these calculations, Biogas I reduced the annual greenhouse gas emissions by 104 kt CO_2 equivalents in the year 2000. A reduction of 404 kt CO_2 equivalents was predicted for 2012, and a reduction of 1.331 kt CO_2 equivalents if all animal slurry and organic wastes available were to be digested using biogas technology I (Fig. 3). The more efficient biogas technology (Biogas II) could reduce the emission of greenhouse gases by 144 kt CO_2 equivalents in the year 2000, by 589 kt CO_2 equivalents in 2012, and by 2.329 kt CO_2 equivalents, if the total production of animal slurry and organic wastes were digested by this optimized technology.

The current level of slurry and organic waste digestion in biogas plants reduces total Danish greenhouse gas emissions by 0.15%. However, the potential reduction of greenhouse gases achievable is 3% assuming all animal slurry and organic waste is digested, and even 4% if it is assumed that the energy produced during digestion replaces coal.

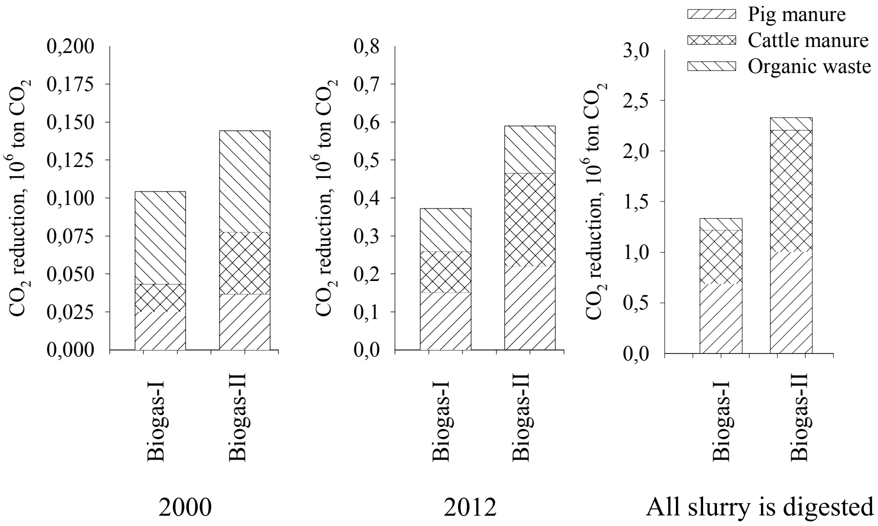

Figure 3. Reduction of annual greenhouse gas emissions for Denmark via co-digestion of animal slurry and organic waste, including reductions in CH_4 and N_2O emissions and substitution of fossil fuel energy (i.e., natural gas).

Comparison with IPCC methodology

IPCC calculates the annual emission of CH_4 from manure storages with the following equation (IPCC Reference Manual, 1997; p. 4.26):

$$EF_i = VS_i \times B_{0i} \times 0,67 \times MCF_i, \qquad \text{Eq. 4}$$

where index *i* refers to animal category, EF_i (kg CH_4) is the daily CH_4 emission rate for category *i*, VS_i is organic matter excreted (kg), B_{0i} is the potential CH_4 production rate (m^3 CH_4 kg^{-1} *VS* day^{-1}), and MCF_i is a CH_4 conversion factor. The factor 0.67 converts the amount of CH_4 from Nm^3 to kg. Table 6 gives parameter estimates for Eq. 4.

Table 6 shows that, according to IPCC, B_0 of pig manure is twice as high as that of cattle manure. However, when B_{0i} values are calculated from the data in Table 1 using Bushwell's equation, the potential CH_4 production rate is almost the same for *VS* excreted from pigs and cattle. This is compensated for in the IPCC methodology by using the same MCF factor for emission from pig and cattle slurry stored under identical storage times. Due to a shorter in-house storage time for pig slurry, the MCF is lower than for cattle slurry (Table 6). The assumption that the MCF for both animal categories is the same is not correct, because the amount of slowly degradable carbohydrates is much higher in cattle slurry than in pig slurry. Therefore, the methane conversion rate (*MCF*) will differ between the two categories. We find that it would be more correct to use the same B_{0i} for pig and cattle manure and use a smaller MCF value for cattle manure.

Table 6. IPCC standard values for calculating methane emissions from stored slurry in cold climate zones.

	MCF Methane conversion factor			B_0 Pot. CH_4 production rate
	Slurry channels	Stores outside	Total	
				$Nm^3\ CH_4\ kg^{-1}$ *VS* ($g\ kg^{-1}$ *VS*)
Pigs (15 days)	0.05	0.10	0.15	0.45 (293)
Cattle (30 days)	0.10	0.10	0.20	0.24 (150)

The Revised 1996 IPCC Guidelines (IPCC, 1997) propose that 5 to 15% of the methane produced is lost in leaking digesters. Danish data suggest that this loss is well below 5% (Møller, H.B., unpublished), probably more studies are needed to determine correct emission factors for different biogas technologies.

The IPCC model estimates the effect of storage time, and to some extent the procedure accounts for the fact that slurry in cold climates is mainly stored during winter when the cattle is housed, while little slurry is stored during summer grazing. The IPCC model may be improved by specifying the time length and climatic conditions during outside storage, which will depend on the time of transfer. Thus, slurry transferred to the outside store in February will be stored for two months, while slurry transferred in October will be stored for seven months.
The IPPC methodology does not present explicit procedures for calculating the reduction in CH_4 emission by anaerobic digestion of manure. However, one may include the effect of digestion on the CH_4 emission during subsequent storage by reducing *VS* in the fermented slurry.

With respect to N_2O emissions from field-applied manure, the link to VS_D proposed here is in contrast to the IPCC methodology, which estimates N_2O emissions from N content alone and thus will not detect any effect of removing degradable C prior to field application. The results of a recent field study suggest that removal of *VS* via anaerobic digestion may reduce N_2O emissions by 20-40% (Petersen, 1999).

We propose that revised IPCC guidelines should include explicit procedures for calculating the effect of anaerobic digestion, maybe along the lines described in this paper. This would improve the understanding of anaerobic digestion as a greenhouse gas mitigation option.

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Effects of cultivation practice on carbon storage in arable soils and grassland

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Summary

Recent estimates suggest that the carbon mitigation potential on agricultural land in Europe is considerable. Grazing lands (along with cropland) are explicitly mentioned for consideration under the Kyoto Protocol. The resulting paper includes cropland and grazing land explicitly as potential Kyoto Article 3.4 activities. Studies in the US have examined the impacts of grazing land management on soil carbon storage but in Europe, data is too limited to make regional projections. On-going EU-funded projects may help us to make better estimates in the future. Much more data is available for cropland and, according to recent estimates, some cropland management scenarios show the potential to meet Europe's 8% Kyoto emission reduction target by 2012. The ploughing of grasslands always leads to a substantial loss of soil carbon. Carbon stocks may be increased by conversion from conventional tillage to reduced- or zero-tillage systems. However, when considering zero-tillage, as well as when considering any land management change, the likely effect on other, non-CO_2 greenhouse gases needs to be considered. Recent studies have shown that as much as one half of the mitigation effect attributable to carbon sequestration under zero tillage can be reversed by an increase in N_2O emissions.

A key factor in implementing Article 3.4 of the Kyoto protocol will be demonstrating and verifying carbon stock changes or fluxes. Ultimately, the degree to which carbon stock changes can be verified depends upon how stringent the definition of verification adopted by the parties turns out to be. If the parties decide on a stringent definition of verifiability, Article 3.4 is at present, and is likely to remain in the future, unverifiable. If less stringent levels of verifiability are adopted, this might be achieved by most parties by the beginning of the first commitment period.

Introduction

Recent estimates suggest that the carbon mitigation potential on agricultural land in Europe is considerable (Smith et al., 2000). Grazing lands (along with cropland) are explicitly mentioned for consideration under the Kyoto Protocol following the 6th Conference of Parties (COP6) and details were finalised at COP7 in Marrakech. The resulting paper, the Marrakech accord, includes cropland and grazing land explicitly as potential Kyoto Article 3.4 activities. In this short paper, the likely mitigation potential of agricultural land in Europe is reviewed, and issues of verification of carbon stock changes are discussed.

Carbon mitigation potential on grazing land

A number of studies in the US have allowed the mitigation potential of US grazing lands to be assessed (Follett et al., 2001; Conant et al., 2001). In Europe, however, data is too limited to make regional projections. A number of EU-funded projects, such as GreenGrass, and an EU COST Action (627 – Carbon sequestration in grasslands) may help us to make better estimates in the future. Gross changes in grassland management (e.g., ploughing) have known effects; under such management practices, soil carbon is invariably lost. It is the subtle management changes (e.g., in livestock management) for which the impacts on soil carbon in Europe are unknown.

Carbon mitigation potential on cropland

Recent studies have shown that there is considerable potential for carbon mitigation on European cropland (Smith et al., 1997; 1998; 2000). Figure 1 shows the estimated mitigation effect of 7 cropland management options.

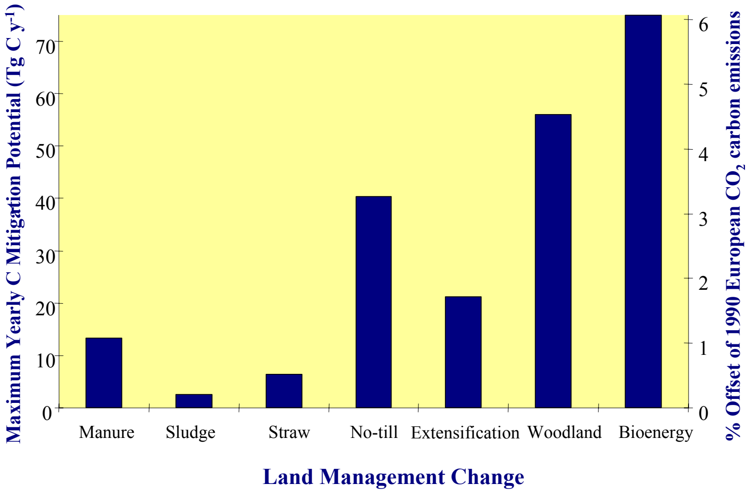

Figure 1. Carbon mitigation potential for 7 cropland management options. See Smith et al. (2000) for further details.

Furthermore, if the land-use is optimised, certain of these management practices can be combined on different areas of land. Figure 2 shows that Europe's 8% emission reduction target can be met by land management change on cropland alone.

Other recent studies include one by Vleeshouwers & Verhagen (2002). In this study, a simple crop-soil model is used to make spatially explicit estimates. The estimates are not constrained by current or likely future practice (e.g., carbon stock changes are calculated assuming that all arable land is converted to grassland), but serve to give alternative estimates of the mitigation potential on arable land.

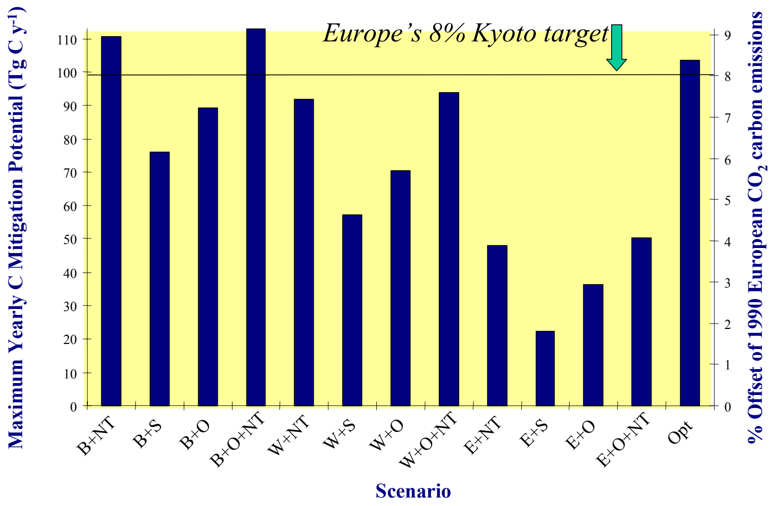

Figure 2. Carbon mitigation potential of combined changes in agricultural management (Smith et al., 2000). The letter before the first "+" in each scenario indicates the land use employed for 10% surplus arable land; B = Bioenergy crops, W = Woodland and E = Extensification. The letters after the first "+" in each scenario denote the management practice adopted on remaining portions of arable land; NT = No till, S = straw incorporation, and O = addition of organic amendments (animal manure and sewage sludge). See Smith et al. (2000) for further details.

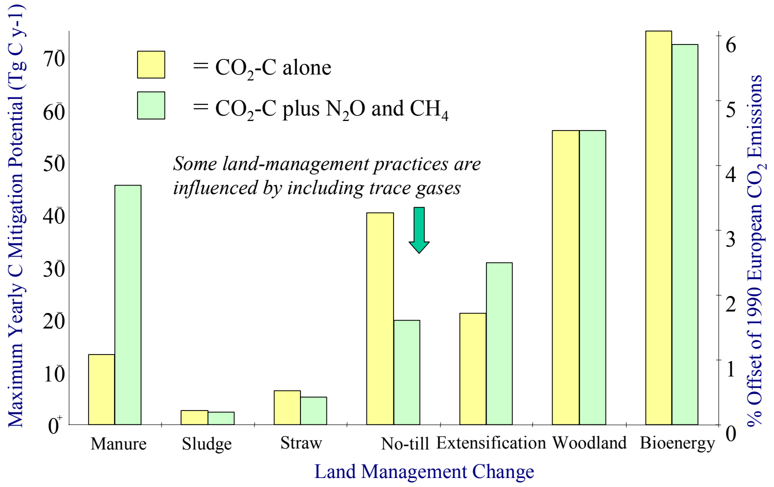

Figure 3. Carbon mitigation potential for 7 European cropland management practices when considering only CO2-carbon effects and when considering also impacts on non-CO_2 greenhouse gases, N_2O and CH_4. See Smith et al. (2001) for further details.

In any case, when considering any land management change, the likely effect on other, non-CO_2 greenhouse gases needs to be considered. For example, recent studies have shown that as much as one half of the mitigation effect attributable to carbon sequestration under zero tillage can be reversed by an increase in N_2O emissions (Smith et al., 2001). Figure 3 shows the carbon mitigation potential when considering CO_2 impacts alone and when considering also impacts on other, non-CO_2 greenhouse gases.

Verification of soil carbon changes under Article 3.4. of the Kyoto Protocol

Verification of soil carbon changes has not been given much thought until recently (Smith, 2001). A significant generic problem with the estimation of changes in terrestrial biospheric carbon stocks relates to resolution (the smallest detectable change). Because the rate of change of most biospheric pools is slow, particularly in relation to the size of the pool, resolvable changes in stock are typically not easily obtained for the larger pools.

Many Article 3.4 activities include a soil carbon component. The measurement of changes in soil organic carbon in the mineral horizons provides a good example of the difficulties faced when trying to demonstrate a stock change over a relatively short period. Such change may be difficult to measure in some soils over a 5-year commitment period because, although potentially large in absolute terms, they may be small compared with background levels. It is sometimes possible to measure the rate of change in soil organic carbon stock during a commitment period, but because of high spatial variability many sub-samples may be required to obtain a mean with an acceptable standard error.

In a recent paper, the minimum detectable difference in soil organic carbon was calculated as a function of variance and sample size for soil organic carbon changes after 5 years under a herbaceous bioenergy crop (Garten & Wullschleger, 1999). The authors showed that the smallest difference that could be detected was about 1 tonne of carbon per hectare, and this could only be done using exceedingly large sample sizes. The minimum difference that could be detected with a reasonable sample size and a good statistical power (90% confidence) was 5 tonnes of carbon per hectare. Most agricultural practices will not cause the soil to accumulate this during a 5-year commitment period (Paustian et al., 1997; Smith et al., 2000).

Cost is also a factor in verifiability. In some cases, the cost of demonstrating the change in stocks to the required level of accuracy and precision may exceed the benefits accrued from the increase in stocks. The cost of demonstrating a change in soil organic carbon stock could be decreased by developing locally calibrated models that can use more easily collected data, but there are further verification issues associated with such an approach.

Whether or not Article 3.4 is verifiable depends critically on what the parties decide verifiability is. At its most stringent, verifiability would entail the sampling of each georeferenced piece of land subject to an Article 3.4 activity at the beginning and end of a commitment period, using a sampling regime that gives adequate statistical power. Soil and vegetation samples and records would be archived and the data from each piece of land aggregated to produce a national figure. Separate methods would be required to deliver a second set of independent, verification data. Such an undertaking at the national level would be impractical and prohibitively expensive. At its least stringent, verifiability would entail the reporting of areas under a given practice (without georeferencing) and the use of default values for a carbon stock change for each practice, to infer a change for all areas shown to be under a given practice. Some scientists have argued that even the area claimed to be under a given practice will, for practical purposes, be unverifiable (Nilsson et al. 2000).

Intermediate in the range of stringency of definitions of verifiability is a scheme in which areas under a given practice are georeferenced (from remote sensing or ground survey), changes in carbon are derived from controlled experiments on representative climatic regions and on representative soils (or modelled using a well-evaluated, well-documented, archived model), and intensively studied benchmark sites are available for verification. Many of the proposed schemes for carbon accounting under Article 3.4, such as those by Australia, Canada and the US, fall into the intermediate category.

If the parties decide on a stringent definition of verifiability, Article 3.4 is at present, and is likely to remain in the future, unverifiable. If less stringent levels of verifiability are adopted, a low level of verifiability might be achieved by most parties by the beginning of the first commitment period.

Conclusions

The carbon mitigation potential on agricultural land in Europe is considerable, but we need to improve our understanding of the impacts of grazing land management on soil carbon dynamics. We also need to improve our estimates and reduce our uncertainty associated with carbon mitigation options on cropland. Critical to this will be improving our understanding of the factors controlling the flux of non-CO_2 greenhouse gases from the soil.

In implementing the Kyoto Protocol, we need to improve our methods of measuring soil carbon stock changes and in developing frameworks within which we will measure, monitor, model, report and verify changes in agricultural soil carbon stocks.

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Carbon balances for arable soils – weak data sets and strong theory

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Summary

Carbon dioxide in the atmosphere is connected to soil organic carbon. We suggest a method for calculating national soil/atmosphere carbon exchange, which is based on soil carbon pool changes. The level of complexity is slightly above standard IPCC methodology, but below most modelling exercises. The approach is based on using available theoretical knowledge and spatially low-resolution (regional) data, since crop, crop yield and soil properties are not available for each individual field.

We use the ICBM model for budgeting and for creation of what-if scenarios, e.g., due to changed land use or climate. By changing just a few of the in total five parameters used in this model we can project steady-state carbon pools as well as 30-year dynamics. Examples from Swedish arable land are given.

Introduction

The title statement may seem provocative, but we think it contains more than a grain of truth. National carbon balances for agricultural soils are not hard to calculate, provided we have enough datasets of good quality. For every hectare we need present soil carbon mass and quality, crop, yield, annual input mass and quality, soil type and future soil climate (monthly averages will do) – and then we can run a simple soil carbon model for every hectare and project what will happen to soil carbon pools and fluxes. If we do this within a GIS map, we can also produce nice, coloured maps indicating what will (or may) happen all over the country, for example by increasing the proportion of grass leys. Unfortunately, we do not have these high-resolution datasets, and so we will have to make the best of what we have. This short paper concerns what we can do to make soil carbon budgets from theoretical insights, low-resolution data, and with limited resources.

Theory

Soil carbon pool sizes are the result of differences in inputs and outputs. For simplicity, we assume that carbon input to the soil is constant at a rate of i kg year^{-1}, and that a certain fraction ($k_1 = 0.01$ year^{-1}) of the soil carbon is respired as CO_2-C each year. Thus 1 tonne of soil C will lose 10 kg C to the atmosphere each year, and 2 tonnes will lose 20 kg, i.e., the fraction but not the amount is constant. Written as a differential equation, the model looks like this:

$$dC/dt = i-k_1C \quad \text{Eq. 1}$$

In words: For each infinitesimally small time step (t) the change in soil carbon (C) is the input (i) minus the fraction parameter (k_1) times the carbon mass present in the soil (C).

At equilibrium (**steady state** of soil carbon mass), $dC/dt =0$. The conditions for equilibrium are thus:

$$i=k_1C \quad \text{Eq. 2}$$

In words: When the input (i) is equal to the respiration output (k_1C) there is no change in the amount of soil carbon, even if the inputs and outputs are large. The amount of C present in the soil at steady state thus becomes $C_{ss} = i/k_1$. For example, to maintain 1 ton of soil C with $k_1= 0.01$ year^{-1}, we need an annual input (i) of 10 kg C.

When C is large, i must be large to maintain C unless k_1 is very small. In other words, if we have increased C, we must maintain a high i to maintain the high C mass. If on the other hand we reduce i to zero, soil carbon will decrease to zero, but at a decreasing rate. If C is plotted against time, we get the familiar "exponential decay curve" which becomes less and less steep with time. Perhaps less familiar is the fact that if we start at steady state and then double the annual input, the rate of increase will gradually decrease towards a new, twice as high, steady state (Fig. 1). Note that the additional 1000 kg of input (100 x 10) have only increased soil C by 630 kg after 100 years. See Andrén & Kätterer (2001) for further discussion of basic principles.

To make the carbon mass increase linearly, we have to add carbon in a form that is not subject to decomposition, e.g., as charcoal. If we added 10 kg charcoal year^{-1}, we would actually have 100 x 10=1000 kg charcoal in the soil after 100 years. Further, after 200 years we would have 2000 kg of charcoal, and a steady state will never be reached.

Organic soil carbon is not homogeneous, and most soil carbon models take this into account by handling a number of pools of varying decomposability. Further, the varying quality of the input has to be managed, and some factor has to be included to account for differences in soil climate, e.g., a dry soil will show lower decomposition rates than a moist soil.

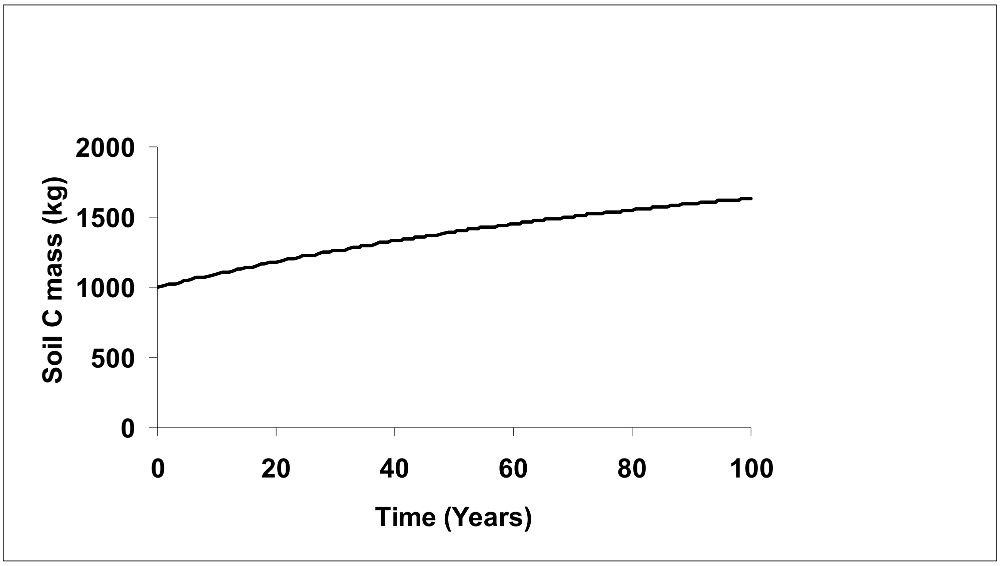

Figure 1. Soil carbon mass dynamics in a hypothetical soil. The initial mass (1000 kg soil C) is in balance when i =10 kg year^{-1} and k_1C = 0.01 year^{-1} (Eq. 2). The graph shows what will happen if we double this annual input (i =20 kg year^{-1}) and maintain this for 100 years. Note the decreasing rate of increase and that, after 100 years, the new steady-state mass (2000 kg) still is distant.

ICBM, Introductory Carbon Balance Model, was developed as a minimum approach for calculating soil carbon balances in a 30-year perspective (Andrén & Kätterer, 1997). The model is based on well-known concepts (see, e.g., Hénin & Dupuis, 1945), and it has two state variables or pools, "Young" (*Y*) and "Old" (*O*) soil carbon. Two pools were considered a minimum, since the model was intended to handle inputs of different qualities, such as wheat straw vs. farmyard manure, as well as, e.g., ploughing of grassland. ICBM has five parameters: i, k_Y, h, k_O, and r_e. (Table 1, Fig. 2).

Table 1. The parameters of the ICBM model, their typical dimensions, and the effect on total soil carbon mass of an increase in parameter value. Y and O represent young and old soil carbon, respectively.

Parameter	Symbol	Typical dimension	Effect on soil C mass of increase
Input	i	kg year^{-1}	Positive
Decomp. rate constant for *Y*	k_Y	year^{-1}	Negative
Humification coefficient	h	dimensionless	Positive
Decomp. rate constant for *O*	k_O	year^{-1}	Negative
External influence on k_Y and k_O	r_e	dimensionless	Negative

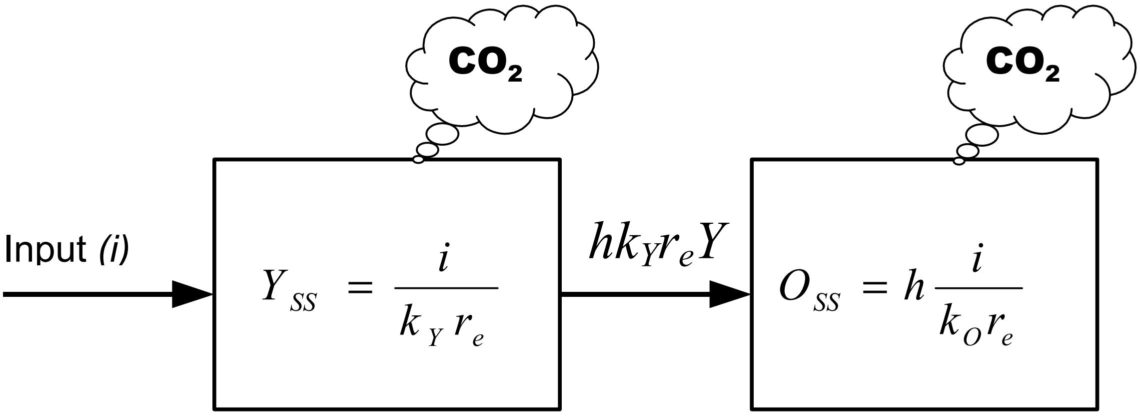

Figure 2. The ICBM model describing soil carbon balances. i = (annual) input, Y = young soil carbon, O = old soil carbon, k_Y = fraction of Y that decomposes (per year), k_O = fraction of O that decomposes (per year), h = humification coefficient, r_e = external influence coefficient. The index "SS" denotes the equation for calculating the steady-state value for that pool (Andrén & Kätterer, 2001).

The "humification coefficient" (*h*) controls the fraction of *Y* that enters *O*, and (*1-h*) then represents the fraction of the outflow from *Y* that immediately becomes CO_2–C. The parameter r_e summarizes all external influence on the decomposition rates of *Y* and *O*.

The model is analytically solved, i.e., simulation techniques are not necessary, model properties can be mathematically analysed, and the model can be run and optimised in an ordinary spreadsheet program (Excel etc.). There are also, in analogy with the one-compartment model above, equations for steady-state conditions, i.e., when the pools are constant and the inputs and outputs balance out. The steady-state equation for *Y* is:

$$Y_{ss}=i/k_Y r_e \qquad \text{Eq. 3}$$

The corresponding equation for O (when Y is at steady-state) is:

$$O_{ss}=ih/k_O r_e \qquad \text{Eq. 4}$$

Application

The crucial question is, of course, how we can use this fairly solid theoretical insight to produce soil carbon budgets for arable land at a national level, including "what if"-projections of, e.g., climate and land use change. We think the best way to approach the problem is to investigate which information is available, and how much more information can be gained with the resources available. In the following, we will use Swedish arable land as an example, but the approach can be used anywhere.

Annual agricultural statistics (crops, yields etc.) are available for Sweden (e.g, Anonymous, 1994). The statistics are grouped into eight regions, for example, "South Swedish Plains". For a dynamic carbon budget, we need the following information:

A) Crops and yields to estimate carbon inputs to the soil, but also as a factor in soil climate, since grass for hay production will create drier soil conditions than annual crops.
B) Soil types and present day carbon levels are also needed, and a recent investigation provided information that can be aggregated into the eight regions (Eriksson et al., 1997). As can be seen in the theoretical part above, the initial mass of carbon is crucial for the balance (see also Kätterer & Andrén, 1999), and the soil type will affect both soil climate and humification quotients.
C) Climate, i.e., weather station data which are readily available and can be aggregated into the eight regions. The main problem here is how to go from rainfall and air temperature data to soil climate, but fairly standardized methods are available (Allen et al., 1998).

For each of the eight regions we can calculate:

1) The area of each soil type in the region.
2) The proportion of different crops and their average yields in the region.
3) The average climate for the region.

Soil types will have to be lumped into, e.g., three categories for each region: Sandy, clayey and organic. We cannot calculate inputs from every conceivable crop in every region, since we do not know how much each crop contributes to carbon input and soil climate. Lumping the crops into three categories may be a good strategy, e.g., grass ley, cereals and sugar beet. These three categories will be fairly different in C input, residue quality and soil climate influence. From the average climate for the region we can calculate a factor that, together with the soil type and crop, will give us the soil climate for a given soil type and crop for a given region.

If we use this approach, we will have 8 regions x 3 soil types x 3 crop types = 72 different descriptions, which together add up to the total Swedish arable land. The ICBM model can easily be run with 72 different parameter sets in a single Excel spreadsheet. When the present situation is satisfactorily described, it is quite easy to project the effects on carbon pools and CO_2 fluxes of, e.g., climate change, increased grass ley areas etc.

One relevant question is where to get the parameter values, e.g., how much we should change r_e between grass ley and annual crops. The soil-pool-oriented approach has the advantage that there are numerous experimental results available, both from long-term field experiments and more specialized laboratory and small plot experiments. Particularly the long-term field experiments give integrated results for time periods relevant for projections (10 –50 years), and it is reasonably easy to estimate parameter values from well-designed experiments (see for example Andrén & Kätterer, 1997). In contrast, the now so fashionable total ecosystem flux measurements (eddy covariance towers) suffer from serious problems in calculating annual fluxes from instantaneous, highly variable flux measurements.

Currently, we have made preliminary ICBM projections using a somewhat simpler 27-parameter set spreadsheet, and some of the results are (Andrén & Kätterer, 1999, 2001):

Total carbon mass in Swedish arable topsoil (0-25 cm depth) is slightly below 300 Mt.

The mineral soils are close to steady state, but the organic soils lose about 1 Mt year^{-1}.

If we ploughed all arable land in Sweden and kept it vegetation-free for one year, we would lose more than 10 Mt during that year, which is more than half the Swedish C emission from fossil fuel combustion.

Acknowledgements

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The ICBM model may be downloaded from
www.mv.slu.se/vaxtnaring/olle/ICBM.html

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Changes in soil C and N content in different cropping systems and soil types

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Summary

Changes in the amount of C and N in cultivated Danish soils were examined using soil samples collected from 336 grid points in the nation-wide Square Grid System (7 × 7 km) in 1986-87 and again in 1997-98. Samples were taken from 0-25 cm and 25-50 cm soil depth. Information on soil types, and on soil use and management, at the farm level was available from a database at the Danish Agricultural Advisory Centre. Soils included in this study were in arable use and were mainly mineral soils.

The average C concentration for each soil type ranged from 1.5 to 2.3% at 0-25 cm soil depth and from 0.9 to 1.6% at 25-50 cm depth. The average N content ranged from 0.107 to 0.161% at 0-25 cm depth and from 0.084 to 0.106% at 25-50 cm depth. Over the 10-12 years, the C and N content at 0-25 cm depth decreased on loamy soils and increased on the coarse sandy soils. Similar changes were observed at 25-50 cm, but changes were more significant at this depth interval. Annual changes in C stocks at 0-50 cm depth ranged from –1.3 to +2.1 t C ha^{-1}, and in N stocks from –115 to +118 kg N ha^{-1}, respectively. In the dataset, the effects of soil type and soil management on the development in soil C storage appeared to be confounded. Implications for C sequestration potentials in soil are discussed and supplemented with results from long-term experiments. We conclude that national inventories of C stocks in arable soils and their role as sinks or sources for atmospheric CO2 carry less weight when based only on measurements in the tilled plough-layer.

Introduction

The soil organic matter content depends on farm management, but also on climate, geology, vegetation, drainage and topography. Short-term fluctuations in soil organic matter depend in particular on crop rotation, fertilisation, crop residue incorporation and manure application. The long-term changes in soil C and N resulting from sustained changes in management have mainly been deducted from long-term field experiments and plots representing only a few soil types (Christensen & Johnston, 1997). The advantage of these experiments is that management is well controlled and data are of high quality. Measurements of long-term changes in C and N contents in soils under normal agricultural practice are rare, but results from experimental field/plot studies can often not be directly transferred to practical agricultural conditions and other soil types, and they should thus be supplemented with farm studies.

In this study, we examined the influence of farm management on soil organic matter levels over a 10-12 year period (Heidmann et al., 2001). Furthermore, the impact of soil type on organic matter levels was investigated. The study was based on a comprehensive data set from the 7 × 7 km Square Grid covering Denmark (Fig. 1).

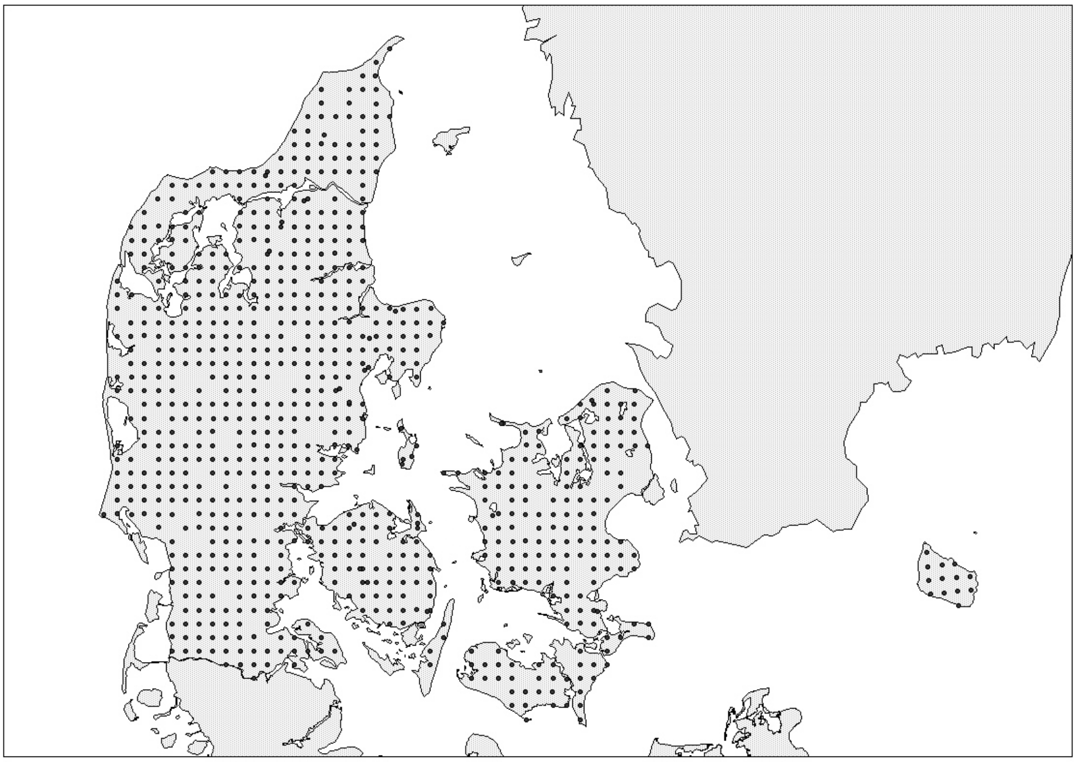

Figure 1. The Danish National Square Grid System.

Materials and Methods

Changes in the amount of organic matter of cultivated soils over a 10-12 year period were examined using soil samples from the nation-wide Square Grid System (Østergaard, 1989). For several years, the Square Grid has been used by the Danish Agricultural Advisory Centre to establish nitrogen fertiliser recommendations for Denmark. The grid includes 830 grid intersection points covering different soil types and cropping systems. Soil was sampled at two depth intervals (0-25 cm and 25-50 cm) when the grid was established in 1986-87, and again in 336 grid intersection points in 1997-98. The selected grid points represented normal agricultural practice. The dried samples were analysed for C using dry combustion in pure oxygen and estimation of CO_2 with IR-detection (Plantedirektoratet, 1994). The N content was estimated using a Dumas method (LECO FP-228 N-determinator) based on thermal conductivity measurements of elementary N (Hansen, 1989).

Information on soil use and management (crop rotations, fertilisation etc.) during the period was available from questionnaires returned by farmers. These data were stored in a database at the Danish Agricultural Advisory Centre.

The soil samples from the grid intersection points were divided into five 'fertilisation types' defined by the manure application practice during the 10-12 year period, i.e., ('Mineral', 'Cattle', 'Pig', 'Mixed' and 'Other'). The fertilisation type was defined as 'Mineral' when manure was not applied. The other types were defined when at least 90 % of the dry matter content in the manure came from cattle, pig or mixed pig/cattle. The category 'Other' included mainly manure from poultry and fur production.

The soil types were defined on the basis of the texture in the surface horizon and then classified with a JB number according to the Danish Classification System (Table 1). The System includes 12 soil types, but most grid points were located on JB1-7 soils. The distribution of grid points according to fertilisation regime and soil type is shown in Table 2. Only a few (13) points were located on JB5, whereas most points were found on JB4 (80) and JB6 (94).

Table 1. Definition of soil types for soil mapping in Denmark (Danish Soil Classification System). The system includes 12 JB No., but only JB 1-7 are shown.

Soil type	JB No.	Percentage by weight					
		Clay > 2 µm	Silt 2-20 µm	Fine sand 20-200 µm	Total sand 20-2000 µ	Humus 58.7 % C	CaCO$_3$
Coarse sand	1	0-5	0-20	0-50	75-100	≤ 10	≤ 10
Fine sand	2			50-100			
Loamy sand	3	5-10	0-25	0-40	65-95	≤ 10	≤ 10
	4			40-95			
Sandy loam	5	10-15	0-30	0-40	55-90	≤ 10	≤ 10
	6			40-90			
Loam	7	15-25	0-35		40-85	≤ 10	≤ 10

Each fertilisation type included about 60-80 grid points. The exception was the group 'Other', which included only 38 points. The fertilisation types were unevenly distributed across soil types. 'Mineral' grid points were predominantly located on loamy soils (72 %), and 'Cattle' grid points on sandy soils (74 %).

Table 2. The distribution of grid intersection points between different fertilisation and soil types. The sandy soils include JB1-4 and loamy soils JB5-6.

	Fertilisation type					
	Mineral	Cattle	Pig	Mixed	Other	All
JB No.	Number					
1	5	21	6	13	4	49
2	0	10	5	9	4	28
3	1	7	6	5	1	20
4	13	22	19	15	11	80
5	2	2	4	2	3	13
6	34	14	24	13	9	94
7	14	5	15	4	6	45
All	69	81	79	61	38	329
% on sand	28	74	46	69	53	54
% on loam	72	26	54	31	47	46

Table 3. Explanatory variables included in the regression analysis.

Clay + silt (%)
Fine sand (%)
Coarse sand (%)
Field capacity (mm)
Wilting capacity (mm)
Years with grass crops during the period
Years with straw incorporation during the period
Years with catch crops during the period
Years with leys during the period
Number of times with manure application during the period
Total dry matter in manure applied during the period (t/ha)
Total amount of C in manure applied during the period (t/ha)
Total amount of N in manure applied during the period (kg/ha)
Total amount of manure (fresh weight) during the period (t/ha)
Total amount of N in fertiliser applied during the period (kg/ha)
Total amount of P in fertiliser applied during the period (kg /ha)
Total amount of K in fertiliser applied during the period (kg/ha)
Soil temperature below grass (average over 12 months, 1961-88) (°C)
Soil temperature below bare soil (average over 12 month, 1961-88) (°C)
Precipitation (average annual sums, 1961-88) (mm)

Changes in C and N were defined as the difference between C and N contents in 1997-98 and 1986-87. The change in the amount of C and N (t ha-1) at 0-50 cm depth was calculated from changes in element concentrations (%) at the two depth intervals, and from average bulk densities defined for each JB number. The average bulk densities were calculated using data from a soil database (Larsen & Sørensen, 1996). Statistical analysis of changes in C and N was performed for both layers, but including only soil samples from JB1-7. The changes were related to fertilisation type and soil type (JB number) in a two-sided variance analysis. In addition, regression analysis including several management variables (Table 3) were performed.

Results

On average, the soil C content increased significantly at 25-50 cm soil depth over the 10-12 years, and decreased insignificantly in the top soil (0-25 cm) (Table 4). In contrast, the soil N content decreased significantly in the topsoil (-0.004 %), but was almost unchanged in the deeper layer. When changes in the amounts of C and N were calculated for 0-50 cm of the soil profile, there was no significant overall change in soil C and N content over the 10-12 year period.

Table 4. Changes in soil C and N content, average for all grid points.

Soil depth (cm)	Unit	C change	N change
0-25	%	-0.03	-0.004*
25-50	%	+0.10*	+0.001
0-25	t ha^{-1}	2.0	-0.02
25-50	kg ha^{-1} year^{-1}	179	-2

* Significant difference at P = 0.05.

Within each soil type, the average C concentration ranged from 1.5 to 2.3% at 0-25 cm depth, and from 0.9 to 1.6% at 25-50 cm depth (Table 5). The average N content ranged from 0.107 to 0.161% at 0-25 cm depth, and from 0.084 to 0.106% at 25-50 cm depth. Over the 10-12 years, the C and N content tended to increase on sandy soils (JB1-4) and decrease on loamy soils (JB5-7) (Fig. 2). The changes were larger in the 25-50 cm depth interval than in the top soil layer (Table 5).

Table 5. Changes in soil C and N content (%) in two depth intervals during the 10-12 year period.

	Soil depth (cm)	Soil type						
		JB1	JB2	JB3	JB4	JB5	JB6	JB7
Carbon	0-25	0.13*	-0.07	0.11	-0.00	-0.12	-0.09*	-0.13*
	25-50	0.36*	0.36*	0.25*	0.18*	-0.10	-0.08	-0.18*
Nitrogen	0-25	0.013*	-0.004	0.005	-0.003	-0.001	-0.010*	-0.014*
	25-50	0.017*	0.023*	0.011	0.007	-0.006	-0.010*	-0.024*

* Significant difference at P = 0.05.

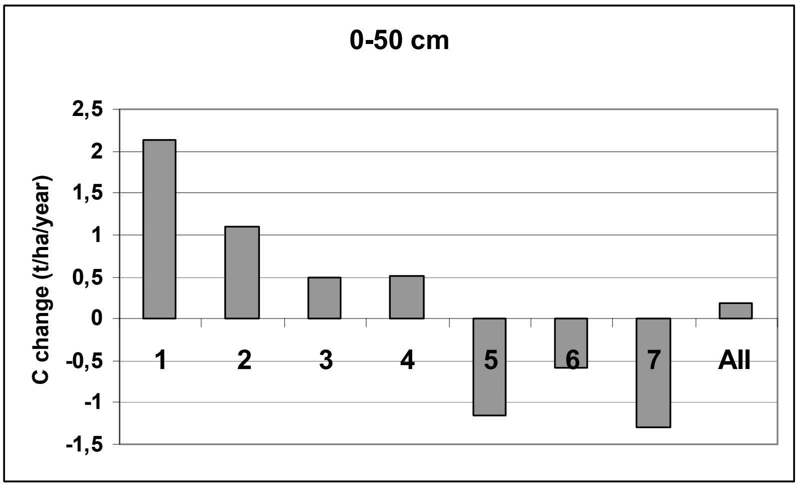

Figure 2. Change in C content (0-50 cm) in mineral soils from 1986/87 to 1997/98 depending on soil type, indicated by JB no.'s.

The fertilisation type had a significant effect on changes in the N content of the top soil. Significant decreases in N of –0.013 and –0.008% were found for the fertilisation types 'Mineral' and 'Pig', respectively. When the results were converted to N stocks based on measurements at 0-25 cm depth, a significant increase of 34 kg N ha^{-1} year^{-1} was found for the fertilisation type 'Cattle', and a decrease of 31 kg N ha^{-1} year^{-1} for the fertilisation type 'Mineral' (Fig. 3). When 25-50 cm soil depth also was included in the calculation of N stocks, no significant changes were observed. The effects of fertilisation type on changes in the average C content were large but not significant (Fig. 4). However, the variability in results was also very large. The average annual change (0-50 cm) in soil C storage ranged from –378 kg ha^{-1} ('Pig') to +897 kg ha^{-1} ('Cattle').

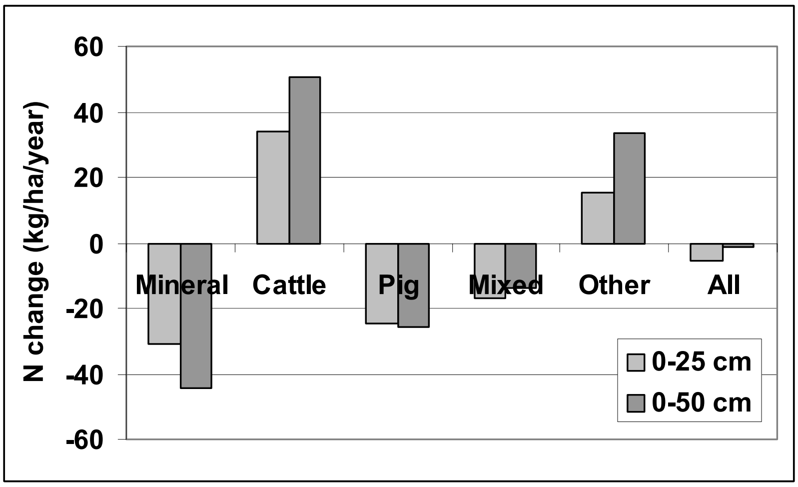

Figure 3. Changes in N content in the 0-25 cm and 0-50 cm of soil samples from 1986/87 to 1997/98 depending on fertilisation type. See text for explanation of fertilisation types.

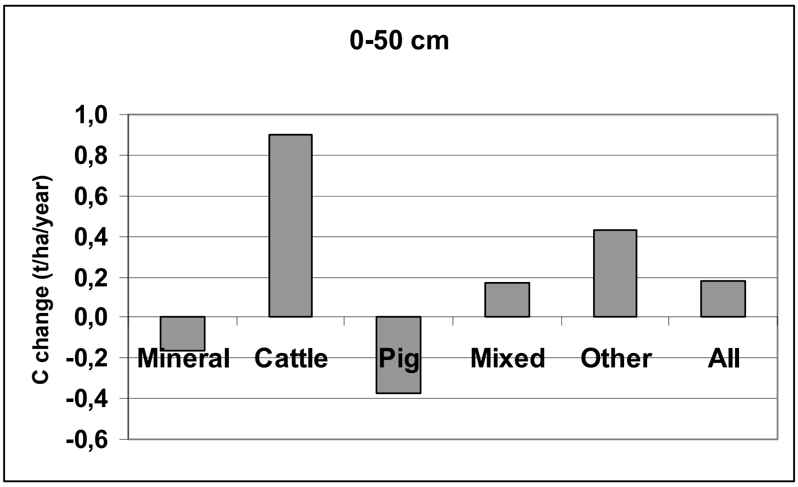

Figure 4. Change in C content (0-50 cm) in soil samples from 1986/87 to 1997/98 depending on fertilisation type (see text for explanation of fertilisation type).

The regression analysis showed that farm management was important for the storage of soil carbon. The management factors: number of years with grass crops on the field, number of manure applications, and amount of mineral fertiliser applied during the 10-12 years had significant positive impacts on the content of organic matter in the soil. The initial value of organic matter was also important for the subsequent development in soil organic matter, indicating that the potential to store organic matter in soils will depend on the starting point. The storage potential increased with decreasing initial values for all soil types. Besides, C and N content was inversely related to the normal temperature below grass.

Discussion

Changes in soil C and N pools occur slowly, and it cannot be expected that the full effect of the different management regimes can be verified during a 10-12 year period. The management factors: amount of mineral fertiliser, number of manure applications, and number of years with grass had positive effects on storage of C and N in the soil. The soil N content increased significant for grid points receiving cattle manure and decreased for points receiving mineral fertiliser or pig manure. The same trends were found for the soil C content, although the changes were not significant. It seems that manure from pigs (mainly slurry) was not as efficient as cattle manure in increasing soil C storage. Christensen (1990) found an average decrease over 30 years of 23-33 kg N ha^{-1} $year^{-1}$ at 0-20 cm soil depth for different crop rotations. This corresponds to the decreases calculated for the fertilisation types 'Pig' and 'Mixed' observed in the present study. Including the 25-50 cm layer produced larger changes, except for the fertilisation type 'Mixed'.

Vitosh et al. (1997) found that the C content was 0.46% C higher in soils receiving cattle manure for 20 years compared with soils receiving mineral fertiliser.

In our study, the difference between the fertilisation types ‘Cattle’ and ‘Mineral’ was 0.17% C at 0-25 cm depth over the 10-12 year period. Voroney & Angers (1995) found an annual C storage at 0-20 cm depth of 600 kg C ha^{-1} after 10 years with 30 t cattle manure ha^{-1} year^{-1}, and an annual decrease of 402 kg C ha^{-1} when only 10 t ha^{-1} year^{-1} was applied. In our study, the average supply of cattle manure to the fertilisation type ‘Cattle’ was about 22 t ha^{-1} year^{-1}, resulting in an annual increase (0-50 cm) of 897 kg C ha^{-1}. Christensen (1990) found decreases (0-20 cm) of 269-362 kg C ha^{-1} year^{-1} in plots receiving mineral fertiliser during a 30 year period, while a decrease of 166 kg C ha^{-1} year^{-1} was observed in our study. The average fertiliser rate was, however, higher in the Square Grid points.

There was a clear tendency for C and N to increase on sandy soils, and to decrease on loamy soils. The potential for organic matter storage was expected to be larger on loamy than on sandy soils (Johnston, 1986). However, in the present study it was not possible to separate the effect of soil texture from the effect of agricultural practice. The ‘Cattle’ points were predominantly represented on sandy soils (72% on JB1-4), and the ‘Mineral’ points predominantly on loamy soils (72% on JB5-7). Several management factors with an expected positive effect on the C and N storage were therefore more frequent on sandy than on loamy soils (Fig. 5), i.e., number of years with grass, leys, and catch crops during the period. An exception was the number of times with straw incorporation, where the negative effect probably resulted from a negative correlation with the frequency of grass crops.

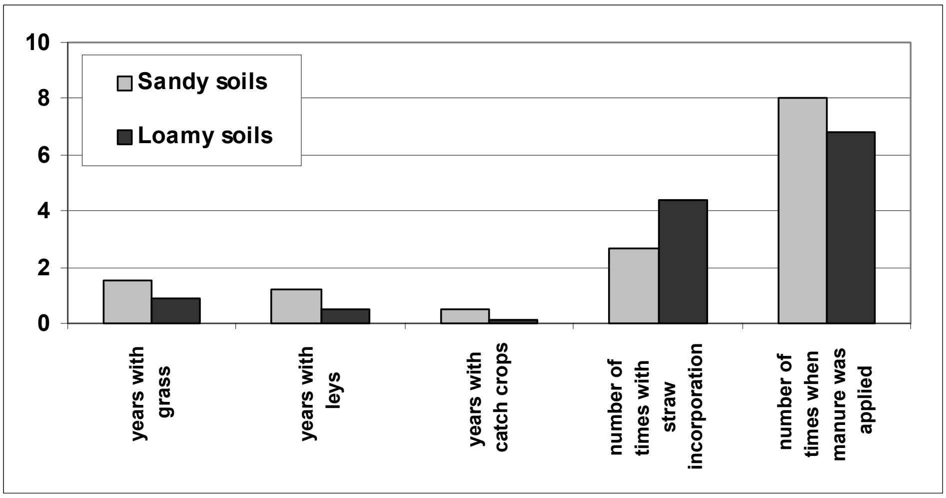

Figure 5. Distribution of selected management variables on sandy (JB1-4) and loamy soils (JB5-7).

Most often only the topsoil is studied, when effects of farm management on C and N stocks are considered (Christensen, 1990; Voroney & Angers, 1995). This study showed that changes at 25-50 cm depth could be significant (Fig. 3 and Table 5). This observation confirms that it is important to include deeper soil layers when estimating changes in soil C and N in response to management.

Conclusions

The effects of soil type and farm management on soil organic matter are not easily separated in farm level studies. Referring also to studies with controlled experimental conditions it can be concluded, however, that management has an effect on soil organic matter storage, and that this effect depends on soil type. It appears that in Danish agriculture, storage of soil organic matter will mainly occur on sandy soils dominated by dairy and cattle farms with abundant manure input and frequent grass crops. In contrast, organic matter appears to be lost on loamy soils dominated by intensive cereal production and pig farming. It was confirmed that deeper soil layers need to be included in calculations of C and N balances. We conclude that national inventories of carbon stocks in arable soils and their role as sinks or sources to atmospheric CO_2 carry less weight when based only on measurements in the tilled plough layer.

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Energy crops as a strategy for reducing greenhouse gas emissions

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Summary

The current Danish energy plan stipulates a production of 5 PJ from energy crops in 2010. This may be attained through growing of either annual (e.g., cereals) or perennial energy crops (e.g., willow or Miscanthus).

Existing Danish data and the IPCC methodology was used to calculate nitrous oxide emissions from and carbon sequestration in soils cropped with an annual energy crop (triticale) or a perennial energy crop (Miscanthus). The calculations for Miscanthus were performed separately for harvest in November or April, since the harvest time affects both yields and emissions. The estimates for Miscanthus were based on a 20-year duration of the cultivation period. The energy use for growing the crops was included in the energy budgets, as was the reduction in CO_2 *emission that will result from substitution of fossil fuel (natural gas). The calculations were performed for both a coarse sandy soil and a loamy sand. The results were compared with current (reference) practice for growing cereals. There were only minor differences in production data and emissions between the two soil types.*

The area required to produce 5 PJ was smallest for Miscanthus harvested in November (c. 25,000 ha), and about equal for triticale and Miscanthus harvested in April (c. 32,000 ha). The reduction in nitrous oxide emissions compared with cereal production was smallest for triticale (20 kt CO_2 *equivalents [eq]* yr^{-1}*) and about equal for Miscanthus at the two harvest times (30-36 kt* CO_2 *eq* yr^{-1}*). Growing Miscanthus resulted in a carbon sequestration, with the highest rates (100 kt* CO_2 *eq* yr^{-1}*) for Miscanthus harvested in April. The energy use for production of triticale was slightly lower than for normal cereal growing, whereas growing Miscanthus for harvest in April resulted in a smaller energy use which corresponded to an emission reduction of 20 kt* CO_2 yr^{-1}*. The substitution of fossil fuel corresponded to 285 kt* CO_2 yr^{-1}*. Summing all items, growing 5 PJ worth of Miscanthus harvested in April resulted in an emission reduction of 447 kt* CO_2 *eq* yr^{-1}*, growing Miscanthus for harvest in November gave a reduction of 355 kt* CO_2 *eq* yr^{-1}*, and growing triticale gave a reduction of 265 kt* CO_2 *eq* yr^{-1}*. Hence, taking nitrous oxide emissions, C sequestration and energy use into account slightly reduced the value of triticale, but significantly increased the value of Miscanthus as a* CO_2 *mitigation option.*

Introduction

The emission of greenhouse gases from agriculture constitutes about 22% of the total anthropogenic emissions in Denmark (Olesen et al., 2001b). The efforts in Denmark to reduce nitrogen losses from agriculture to the environment have and will also in the future contribute to reduce emissions of nitrous oxide, in particular by reducing the amount of nitrogen that is cycled in the system. Agriculture has in a number of areas possibilities for further reducing the total Danish emissions of greenhouse gases. This may work by reducing direct emissions of the gases, including reductions in energy use and reduced emissions of methane and nitrous oxide (Smith et al., 2000). It may also work through the adoption of alternative farming systems that offer possibilities for substituting fossil fuel use and for carbon sequestration in the soil (Olesen et al., 2001a).

Agricultural production of biomass for energy will result in substitution of fossil fuels in addition to the substitution that already occurs from combustion of the existing surplus of straw. In the most recent Danish energy plan, Energy 21, energy crops are assumed to contribute to the energy supply from the year 2005, increasing to 45 PJ yr^{-1} in 2030. The existing estimates do not account for the effect of growing energy crops on nitrous oxide emissions or on carbon sequestration in the soil. Also, the fact that different energy crops imply different levels of energy use during production is not accounted for. This study has attempted to quantify effects of energy crop type and management on total greenhouse gas emissions from energy crop production.

Methods

Growing of annual or perennial energy crops was, for two soil types, compared with ordinary cereal cropping. The soils were a coarse sandy soil and a loamy sand. The reference cereal was assumed to be spring barley on the sandy soil and winter wheat on the loamy sand. Triticale was selected as the annual energy crop and Miscanthus as the perennial energy crop. For Miscanthus, two different harvest times (November and April) were included in the analysis. With reference to the energy plan, Energy 21, it was assumed that the energy crops should contribute 5 PJ by year 2010.

Crop production data

For the ordinary cereal production systems it was assumed that half of the straw was removed for agricultural uses, e.g. bedding material. Spring barley was grown with a catch crop of ryegrass every year. Nitrogen fertilisation was based on mineral fertilisers. On the coarse sandy soil, irrigation was applied to both the spring barley and to the annual energy crop, but not to the perennial energy crop. The irrigation was set to 75 mm for spring barley and 105 mm for triticale (Landbrugets Rådgivningscenter, 1990). No irrigation was applied on the loamy sand soil.

The basic data of crop production and nitrogen use and losses are shown in Tables 1 and 2. Grain yields and N application for cereals were based on norms for the particular soils (Plantedirektoratet, 2000). However, yields and N application in triticale were reduced by 10% to account for the lower input level in bioenergy production (Nielsen, 1999). The straw yields in barley and wheat were set to 55 and 65% of the grain yields, respectively (Landbrugets Rådgivningscenter, 1999). The grain yield was set to 45% of total above-ground biomass (Olesen et al., 2000). The biomass in roots was set at 27% of above-ground biomass. The ryegrass catch crop grown with the spring barley was assumed to have contributed an additional 1 t DM ha^{-1}. Data on N contents in grain as straw were taken from Møller et al. (2000).

Miscanthus is a perennial crop with a slow growth during the establishment phase (1-3 years). Different values of crop production and of inputs were therefore used for each of the first three years, followed by a fixed value for the following years. A total production period of 20 years was used, and Tables 1 and 2 show the production and nitrogen data weighted for this 20-year production period. The data were based on experiments at two sites in Denmark, i.e., Jyndevad (coarse sand) and Foulum (sandy loam) (Jørgensen, 1997; Kristensen, 1998; Jørgensen & Kjeldsen, 2000; Jørgensen & Mortensen, 2000). There was a higher production on the coarse sandy soil, primarily caused by warmer conditions at this site.

Table 1. Annual biomass production (t DM ha^{-1}). The total biomass includes both above- and below-ground biomass. The Miscanthus data are averaged over a 20-year production period.

Soil	Crop	Total	Harvested	Returned
Coarse sand	Spring barley	12.7	5.8	8.0
	Triticale (biomass)	13.8	9.3	4.6
	Miscanthus (November)	20.9	15.3	5.6
	Miscanthus (April)	20.9	10.8	10.1
Loamy sand	Winter wheat	16.8	7.9	8.9
	Triticale (biomass)	13.8	9.3	4.6
	Miscanthus (November)	16.8	12.2	4.6
	Miscanthus (April)	16.8	7.3	9.5

Table 2. Data on annual nitrogen inputs and losses (kg N ha^{-1}). The Mischanthus data are averaged over a 20-year production period.

Soil	Crop	Fertiliser	Returned in crop residues	Ammonia volatilisation	Nitrate leaching
Coarse sand	Spring barley	136	79	8	69
	Triticale (biomass)	118	24	6	63
	Miscanthus (November)	81	38	5	18
	Miscanthus (April)	56	74	4	18
Loamy sand	Winter wheat	166	48	8	62
	Triticale (biomass)	106	24	6	44
	Miscanthus (November)	81	39	5	14
	Miscanthus (April)	56	86	4	14

Nitrate leaching from the cereal crops was estimated using an empirical model (Simmelsgaard, 1998). Ammonia volatilisation was estimated as 2% of the fertiliser nitrogen input plus an additional volatilisation from the crops (Andersen et al., 1999).

Energy consumption in the production

The energy consumption for the cereal crops was calculated separately for each soil type using the ØKOBÆR model (Dalgaard et al., 2001). This model was also used for Miscanthus, but the management was set to vary over the 20 year growing period, and no separation was made between the two soil types. The energy used for transportation of biomass to the power plant was estimated assuming an average transport distance of 50 km (Nielsen & Mortensen, 2000). The energy use was converted to CO_2 emissions using the following emission factors (Dalgaard et al., 2000): diesel, 74.0 kg CO_2 GJ^{-1}; electricity and machinery, 95.0 kg CO_2 GJ^{-1}; and fertiliser, 56.9 kg CO_2 GJ^{-1}.

Fossil fuel substitution

Estimates of energy content in the biomass were based on the combustion value, which accounts for contents of ashes and water in the biomass (Videncenter for Halm- og Flisfyring, 1993, 2000). The energy content of triticale (15% water) was 16.8 MJ kg^{-1} DM. The energy content of Miscanthus harvested in November (55% water) was 14.9 MJ kg^{-1} DM, and the energy content of Miscanthus harvested in April (15% water) was 17.5 MJ kg^{-1} DM.

It was assumed that energy crops will substitute natural gas in the energy supply (Audsley, 1997). An emission factor of 56.9 kg CO_2 GJ^{-1} was used for natural gas. It was assumed that the conversion efficiency of energy in biomass was the same as for natural gas, but in reality the efficiency will often be lower for biomass.

Carbon sequestration in soils

The carbon turnover model described below was used to estimate changes in soil carbon storage. A fixed initial content of 70 t C ha^{-1} in the top 30 cm was used, which corresponds to the average soil carbon content measured on Danish arable farms (Heidmann et al., 2000). However, the effect of crop type on carbon stock changes were independent of initial C content in the soil. The development in soil carbon content (incl. roots and rhizomes) was calculated over a 20-year period by numerical integration of Eq. 4 and 5 (see below). The carbon sequestration was then estimated as the average annual increase over the 20 years.

The carbon turnover in soils was described by a first-order differential equation:

$$\frac{\mathrm{d}C}{\mathrm{d}t} = hA - kC \qquad \text{Eq. 1}$$

where C is the soil content of organic carbon (t C ha^{-1}), t is time (years), k is the turnover rate (yr^{-1}), h is the humification coefficient, and A is the added organic carbon (t C ha^{-1} yr^{-1}).

The development of soil carbon content without addition of organic matter was described by:

$$C_t = C_0 \exp(-kt) \qquad \text{Eq. 2}$$

where C_t is the carbon content at time t, and C_0 er is soil carbon content at time 0.

The turnover rate k was estimated at 0.0136 using data for development in carbon content in the bare soil plots of the Askov long-term experiments (Christensen, 1990). The estimation was performed using Eq. 2 and the procedure NLIN of SAS (SAS Institute, 1988).

This estimate of turnover rate represented a system with annual soil cultivation. Balesdent et al. (1990) found that mineralisation in undisturbed soil was only 47% of the mineralisation in normally tilled soils. Smith et al. (1998) found for a range of North European experiments with minimum tillage that avoiding soil tillage caused an annual increase in soil C content of 0.73% of the total carbon content. These results imply that growing perennial energy crops without annual soil tillage will reduce the carbon turnover rate by 50% to $k = 0.0068$.

The humification coefficient was calculated using experiments with different rates of straw application. These included three experiments from Denmark continuing for between 9 and 23 years (Thomsen, 1995; Christensen & Olesen, 1998), and one experiment from Sweden carried out over 35 years (Kirchmann et al., 1994). The difference in soil carbon content at 0-25 cm depth between plots with and without straw application (ΔC_t) at time t was modelled as:

$$\Delta C_t = \sum_{i=1}^{t} (hA + \Delta C_{i-1}) \exp(-k) \qquad \text{Eq. 3}$$

The application of carbon in straw was estimated assuming a dry matter content in straw of 85% and a carbon content in dry matter of 45%. For the Danish experiments the humification coefficient was estimated at 0.27, and for the Swedish experiment at 0.23 using Eq. 3 and the procedure NLIN in SAS. An average humification coefficient of $h = 0.25$ was used in the model estimations.

Crop production will not only lead to carbon additions from straw and other above-ground crop residues, but also from below-ground residues. The amount of below-ground crop residues was estimated using data for the difference in carbon content in experimental treatments with removal of all above-ground crop residues and an experimental treatment with bare soil in a Swedish experiment running for 35 years (Kirchmann et al., 1994). The experimental treatment with removal of crop residues was part of a cereal-based crop rotation with calcium nitrate as fertiliser. The carbon input was estimated at 1.2 t C ha^{-1} yr^{-1} using Eq. 3. The average annual above-ground cereal dry matter yield in the experiment was 7.0 t ha^{-1}, or 3.2 t C ha^{-1}. The carbon input from below-ground residues thus constituted ca. 27% of the total carbon uptake by the crop.

The Miscanthus-derived C content at 0-30 cm soil depth was estimated to constitute 4.6 t C ha^{-1} after 9 years, and 14.1 t C ha^{-1} after 16 years of continuous Miscanthus cropping on the basis of ^{13}C-content in soil from an experiment at Hornum in Denmark (Hansen & Christensen, 2001). There were also considerable amounts of roots and rhizomes in the soil, and these constituted 6.7 t C ha^{-1} and 7.3 t C ha^{-1} after 9 and 16 years of Miscanthus growth, respectively.

To model soil organic matter in soil where Miscanthus is grown, a carbon pool of active roots and rhizomes (C_a) was introduced. This pool had a constant turnover rate (m):

$$\frac{dC}{dt} = h\left(mC_a + hA_o\right) - kC \qquad \text{Eq. 4}$$

$$\frac{dC_a}{dt} = A_u - mC_a \qquad \text{Eq. 5}$$

where C is the soil carbon content (without active roots and rhizomes). A_o is the input of carbon from above-ground plant residues (t ha^{-1} yr^{-1}), and A_u is the annual input of carbon to active roots and rhizomes (t ha^{-1} yr^{-1}).

The Miscanthus crop in the experiment at Hornum was harvested in spring, and it was assumed that the carbon inputs (A_o and A_u) were constant from the fourth year after crop establishment. The inputs in years 1, 2 and 3 were assumed to constitute 1/6, 1/3 and 1/2 of the final input level, respectively. The final input level of above-ground plant residues was assumed to be 3.6 t C ha^{-1} yr^{-1}. It was also assumed, as previously argued, that the turnover rate of soil carbon was only half of the standard value of 0.0136, since no soil tillage was performed. The annual inputs to roots and rhizomes were assumed to be a fixed percentage of the above-ground dry matter production which, based on data from experiments at Foulum, was set to 7.2 t C ha^{-1} yr^{-1} after four years of cropping. Using these assumptions and the NLIN procedure of SAS, m in Eq. 4 and 5 was estimated at 0.12, and A_u was estimated to be 16% of the above-ground carbon production.

Nitrous oxide emissions

The emission of nitrous oxide was calculated using the IPCC methodology (IPCC, 1997). The emission factors were 0.0125 kg N_2O-N kg^{-1} N for nitrogen in both fertiliser and crop residues. The emission factor was 0.010 kg N_2O-N kg^{-1} N for ammonia volatilisation, and 0.025 kg N_2O-N kg^{-1} N for nitrate leaching. The emissions vary over time for Miscanthus, and the estimates were therefore calculated as the average of a 20-year period.

Results and discussion

The calculated annual nitrous oxide emissions and changes in soil carbon stocks derived from growing different crops are shown in Table 3. There were large differences between crops, but only small differences between the two soil types. The highest emissions reductions were obtained for the Miscanthus crops for all emissions categories.

Table 3. Annual nitrous oxide emissions, energy use and substitution of fossil energy use by growing various crops. All values are expressed at t CO_2-eq yr^{-1}.

	Cereal	Triticale biomass	Miscanthus November	Miscanthus April
Coarse sandy soil				
Nitrous oxide emission	2.19	1.66	0.97	1.03
CO_2 emission from changes in soil C	0.19	1.42	-1.87	-3.59
Energy use in crop production	1.11	1.08	0.97	0.51
Substitution of fossil fuel	0.00	-8.85	-12.93	-10.75
Total emission	3.49	-4.69	-12.86	-12.80
Loamy sand soil				
Nitrous oxide emission	2.10	1.36	0.92	1.05
CO_2 emission from changes in soil C	-0.14	1.42	-1.21	-3.09
Energy use in crop production	1.04	0.75	0.97	0.51
Substitution of fossil fuel	0.00	-8.85	-10.30	-7.27
Total emission	3.00	-5.32	-9.62	-8.80

Table 4. Land area (equal mixture of coarse sand and loamy sand) required for production of 5 PJ worth of biomass based on combustion value, and emission reductions achieved compared with conventional cereal production. All emission reductions are shown in kt CO_2 eq yr^{-1}.

	Triticale biomass	Miscanthus November	Miscanthus April
Area (ha)	32140	24812	32797
Nitrous oxide emission reduction	20	30	36
Soil carbon sequestration	-45	37	108
Reduced energy use	5	3	18
Substitution of fossil fuel	285	285	285
Total emission reduction	265	355	447

Table 4 shows the land area required to grow the different bioenergy crops and the associated reductions in greenhouse gas emissions. The area required to produce 5 PJ was smallest for Miscanthus harvested in November, and about equal for triticale and Miscanthus harvested in April. The reduction in nitrous oxide emission compared with cereal production was smallest for triticale (20 kt CO_2 eq yr^{-1}) and about equal for Miscanthus at the two harvest times (30-36 kt CO_2 eq yr^{-1}). Growing Miscanthus, but not triticale, resulted in soil carbon sequestration, with the highest rate of 108 kt CO_2 yr^{-1} for Miscanthus harvested in April. The energy use for production of triticale was slightly lower than for normal cereal growing, whereas growing Miscanthus for harvest in April results in a smaller energy use corresponding to an emission reduction of 20 kt CO_2 yr^{-1}. The substitution of fossil fuel corresponded to 285 kt CO_2 yr^{-1} with all energy crops. Summing all items, growing 5 PJ worth of Miscanthus harvested in April resulted in an emission reduction of 447 kt CO_2 eq yr^{-1}, while growing Miscanthus harvested in November gave a reduction of 355 kt CO_2 eq yr^{-1}, and growing triticale gave a reduction of 265 kt CO_2 eq yr^{-1}. Hence, taking nitrous oxide emissions, C sequestration and energy use into account slightly reduced the value of triticale, but significantly increased the value of Miscanthus as a CO_2 mitigation option.

The uncertainties associated with these estimates are probably mainly associated with the calculation of root-derived carbon and nitrogen, as these were determined indirectly. Also, only carbon in the upper 30 cm of the soil was included in the calculations. There are probably differences in the root depth of the different varieties and thus in the sequestration of carbon below this depth. The effect of tillage on soil carbon turnover between the perennial and annual crops was also important, and further studies on this are needed.

The reference crop used in these calculations was a cereal crop. There is currently an option in the EU regulation to grow energy crops as an alternative to set-aside. It is expected that the requirement for set-aside will be removed. Set-aside crops of permanent grass will differ considerably from cereal crops with respect to nitrous oxide emissions and carbon sequestration. Further investigations into the effect of reference crop for the estimated benefits of bioenergy crops with respect to greenhouse gas mitigation are thus needed.

There appears to be a number of environmental advantages of growing perennial as opposed to annual energy crops. However, the promotion of perennial energy crops requires a long-term and coordinated strategy involving both technical and political aspects. The individual farmer needs a clear political signal that energy crops are given priority also in future changes of the agricultural policy. This is required because perennial energy crops occupy land areas for an extended period of time.

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Nitrous oxide emissions from manure handling – effects of storage conditions and climate

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Summary

Stored animal manure and manure applied in the field contributes an estimated 20% to the total anthropogenic emissions of nitrous oxide (N_2O) in Denmark. Manure composition, handling and climatic conditions may all influence the emission level during storage, but there are relatively few experimental data on emissions of N_2O from manure management, including animal houses, slurry stores and manure heaps.

Among animal housing systems, very high emission rates have been found with pig deep litter, and N_2O emissions are further stimulated by mechanical mixing. Slurry stores are anaerobic, but a recent study showed that N_2O can be produced in porous surface covers such as natural surface crusts, straw or leca pebbles, while no N_2O was emitted from uncovered slurry. The emission was significantly related to the water balance, i.e., the difference between evaporation and rain, during dry periods; during wet periods no N_2O was emitted. For solid manure, previous studies have typically found that less than 1% of total N is emitted as N_2O. Nitrous oxide may be produced throughout the manure heap, provided an environment with both aerobic and anaerobic pockets exists. Profiles from an experimental heap indicated that most of the N_2O emitted from solid manure was produced near the surface of the heap. Increasing density appears to stimulate N_2O emissions up to a point, where the air exchange is significantly impeded.

The IPCC methodology calculates N_2O emissions from manure on the basis of total N content (that is, on the basis of volume) and climate region only. Possibly, estimates of N_2O emissions from slurry stores could be improved by considering surface area, ammonium content and water balance as input variables. Emissions from solid manure heaps should consider surface area and the potential for composting, as reflected in bulk density and moisture content.

Introduction

More than 50% of the N excreted by pigs and cattle is in the urine (Safley et al., 1986), and typically 70-90% of this is urea-N (Bristow et al., 1992; Petersen et al., 1998a), which is rapidly hydrolyzed. From poultry, the main source of inorganic N is ureic acid (>70% of total N content), which is transformed to NH_4^+ via urea (Koerkamp, 1994). In suitable environments with both aerobic and anaerobic phases, sequential nitrification and denitrification can convert the NH_4^+ to NO_3^-, and NO_3^- to N_2, respectively. Both processes can lead to formation of nitrous oxide (N_2O), which may escape to the atmosphere (Müller et al., 1997).

In animal houses with slatted floors, slurry is stored in channels below the slats for up to a month, and outside, in slurry stores, for up to a year. In houses with solid floor covered by a bedding material like straw or sawdust, the resulting deep litter is typically transferred to a store two to three times a year and, thus, the litter is stored for several months both inside and outside the animal house. In housing systems where livestock are tied, the excreta are separated between solid manure (or farmyard manure, FYM) mainly containing faeces and straw, and liquid manure, which is a mixture of urine, water and dissolved faecal components. The liquid manure is continuously trickling through gutters to an outside store, and the FYM is typically removed from the animal house on a daily basis. In Europe, the typical storage time for manure may vary between a few weeks in the UK to as much as nine months in Denmark (Bloxham & Svoboda, 1996; Sommer et al., 1996). Therefore, the emissions from manure stores can differ significantly between countries.

Animal manure collected during housing is typically stored for a period to ensure timely spreading of the manure nutrients to the field, i.e., in connection with the growing season. Stored manure is a source of NH_3 and N_2O to the atmosphere. Currently, management strategies to reduce N losses during storage focus on NH_3 (in itself an indirect source of N_2O). In order to reduce NH_3 volatilization from slurry stores, a surface cover is required in Denmark (Sommer et al., 1993). The cover may consist of slurry organic matter forming a natural surface crust, a layer of straw, or floating leca pebbles (burned montmorillonitic clay). Solid manure may compost during storage, which will enhance NH_3 losses due to increased temperatures and ventilation of the manure heap (Sommer, 2001). In consequence, the government has in new regulations proposed that composting shall be reduced through covering with a gas-impermeable material (Ministry of the Environment, 2001). Both strategies may affect also the potential for direct N_2O emissions.

In this presentation, the potential for N_2O emissions from liquid and solid manure stored inside and outside the animal house is discussed, in particular effects of management and climate on the emission potential. The main focus will be on liquid manure systems, which are the most abundant, and on deep litter systems, the number of which may increase for welfare reasons.

Slurry stores

Inside the house

Slurry stored in channels is not a significant source of N_2O, NO_x or N_2, because little NH_4^+ is oxidized in this predominantly anaerobic liquid environment. The soiled surface of slats and the floor, in contrast, has a large interface between air and slurry that may be a source of N_2O. This was shown in a study by Thelosen et al. (1993), who measured a yearly emission of 0.2 kg N_2O-N per pig place. Assuming that the Danish norm of ca. 10 kg N excreted per pig place and year (Poulsen et al., 2001) is representative for Western European conditions, this corresponds to an N_2O loss of 2% of N excreted.

A review of mainly German studies by Jungbluth et al. (2001) confirms that there may be emissions of N_2O from housing systems with slatted floors. However, more studies are needed to show whether the emissions are significant and how they could be mitigated. One may assume that the emissions are related to the area fouled by the animals, i.e., floor construction, and one option would then be to reduce this area either by reducing the slatted area, by tying the animals, or by removing the urine immediately after excretion. New techniques for measuring gaseous emissions from housed animals have been developed that may throw more light on the regulation of N_2O emissions during housing of livestock (Monteny & Erisman, 1998).

Stored slurry is anaerobic, so there can be no nitrification in the liquid phase, and therefore no denitrification (Sommer, 1997). However, a natural or artificial surface crust on top of the stored slurry can become a mosaic of anaerobic and aerobic sites under drying conditions, thus creating an environment where N_2O can be produced (Hüther et al., 1997; Sommer et al., 2000). Emission rates of up to 25 mg N_2O-N m^{-2} h^{-1} were measured by Sommer et al. (2000), while Jungbluth et al. (2001) quoted field and laboratory scale studies in which emissions ranged from 0.2 to 5.4 mg N_2O-N m^{-2} h^{-1}.

Sommer et al. (2000) observed that from slurry covered with a porous material, N_2O emissions increased with decreasing water balance during periods where evaporation exceeded rainfall (Fig. 1). Drying will enhance convective transport of liquid upward through the cover. Dissolved NH_4^+ can be oxidized by nitrifying bacteria in oxic zones, while in anoxic pockets the products of nitrification can be denitrified. During periods with rain, inorganic N in the surface cover will be leached downward, the concentration of NH_4^+ at the top of the liquid slurry phase will be reduced by dilution, and the air-filled porosity will decline. In this environment the potential for nitrification (and therefore denitrification) is reduced. Accordingly, no emission of N_2O was detected in periods with a positive water balance (Sommer et al., 2000).

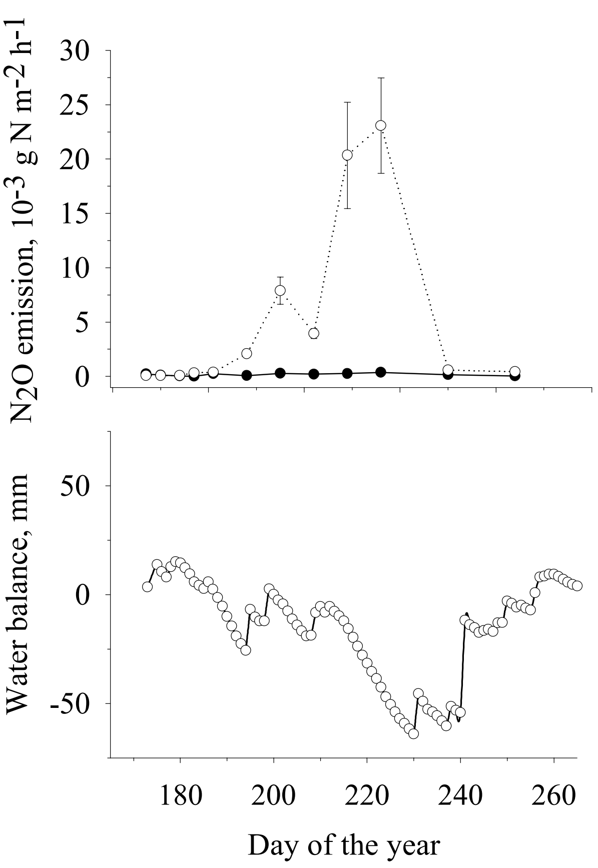

Figure 1. Top: Nitrous oxide emissions from cattle slurry covered with surface crust (open symbol) and uncovered slurry (closed symbol). Bottom: The water balance (rain-evaporation) during the experiment (adapted from Sommer et al., 2000).

Outside storage

No relation between N_2O emissions and the temperatures of slurry or air have been observed (Willers et al., 1993; Sommer et al., 2000). The interface between the liquid slurry and the outside atmosphere will be located at some depth within the surface cover, and due to the insulating effect of the cover it is likely that the temperature in this environment will differ from the bulk slurry temperature, as well as from the ambient air temperature.

Solid manure stores

Inside the house

Deep litter is a mixture of excreta and straw, in which the ratio between inorganic N and organic N is related to excretion rates, strewing rates, and microbial transformations of N. It is estimated that NH_4^+ constitutes 25% of the total N in deep litter, and that in FYM, NH_4^+ constitutes between 25 and 35% of total N (Poulsen et al., 2001). In cattle deep litter, a high proportion of the NH_4^+ derived from urine was found >10 cm below the surface (Henriksen et al., 2000).

Aerobic microbial activity in cattle deep litter may cause a temperature rise to 40-50°C at 10 cm depth. In this layer, oxygen in the air entering the mat is depleted. In a recent study, the N_2O and N_2 production in cattle deep litter was low, probably because nitrification and denitrification processes were inhibited by a combination of low oxygen partial pressure, high temperatures, and a high NH_3 concentration (Henriksen et al., 2000). The hoofs of housed cattle will compact the deep litter, whereas pigs on deep litter will tend to spread the bedding material. Therefore, there is a greater potential for production of N_2O in deep litter of pig houses. Nitrous oxide losses of 5-21% were observed with pig deep litter which was also mechanically mixed once a week (Thelosen et al., 1993; Groenestein et al.,1993; Groenestein and van Faassen, 1996). In pig houses where the deep litter was left untreated, emissions of 0.05-3.73 kg N_2O place^{-1} year^{-1} were recorded in different studies (Jungbluth et al. 2001), corresponding to 0.3-24% of total N.

From tie-stall systems there will also be interfaces between manure and air, which are potential sources of N_2O. Jungbluth et al. (2001) referred to studies which had found significant N_2O emissions from tie-stall dairy houses with animals.

Deep litter stored outside houses

During a period ranging from a few days to several weeks after storage, the temperature of stored solid manure and deep litter may increase to between 60 and 70°C (cf. Fig. 2) due to aerobic microbial metabolism, i.e., composting. Following a rapid increase, the temperature will slowly decline. Composting generates an upward airflow inside the heap and, consequently, gases are effectively transported to the outside atmosphere. Further, composting causes an increase in pH, which increases the NH_3-to-NH_4^+ ratio (Karlsson & Jeppson, 1995), and the vapor pressure of NH_3 is increased by 40-60% for every 10°C increase in temperature (Petersen et al., 1998*b*). Both factors stimulate volatilization of NH_3 from the heap. In solid manure with a low straw content, such as solid cattle manure, the air exchange is low and composting will normally not occur (Forshell, 1993).

During the initial phase of storage, before the temperature increases, there can be a production and emission of N_2O from the heap (Fig. 2). During the composting phase, little N_2 and N_2O is produced, partly because NH_3 volatilization depletes the pool of NH_4^+, and partly because nitrifying and denitrifying microorganisms are not thermophilic (Hellman et al., 1997). After the temperature decline, conditions suitable for nitrification-denitrification may be re-established, which can lead to a secondary increase in N_2O emissions (Fig. 2).

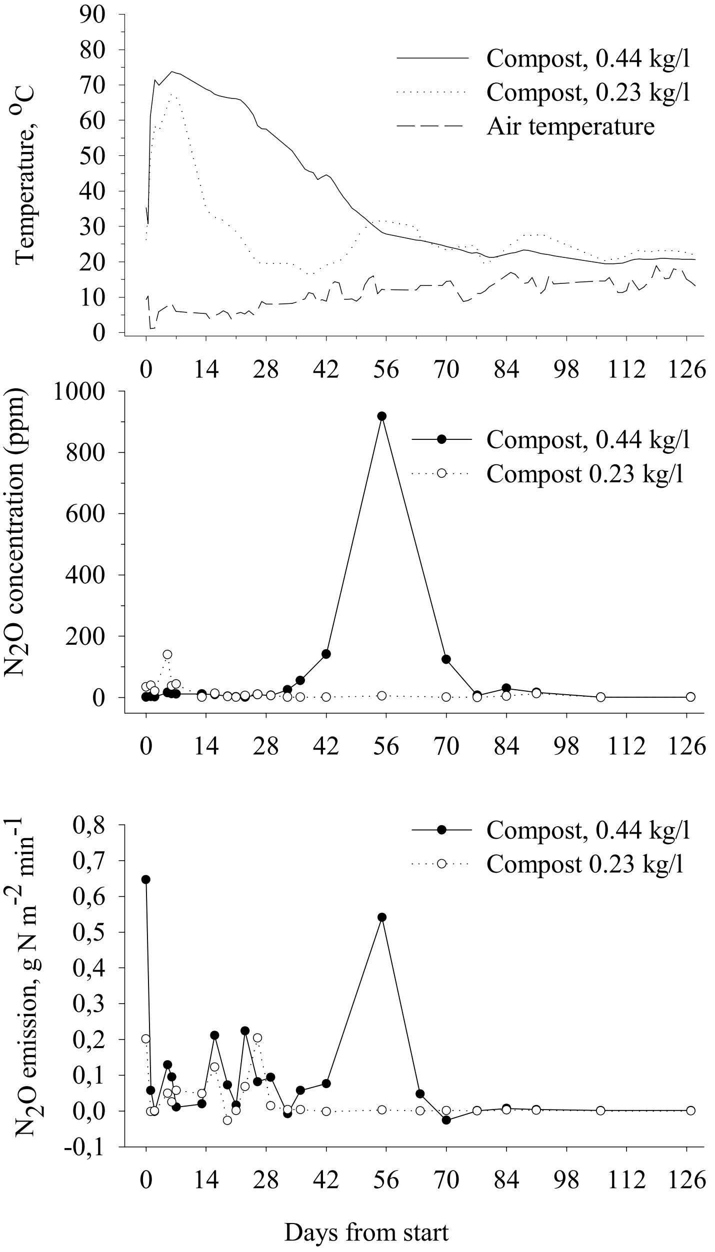

Figure 2. The manure temperature (top), N_2O concentration (middle) and N_2O emission rate from heaps with a density of 0.44 kg/l or 0.23 kg/l (adapted from Sommer & Møller, 2000).

Nitrous oxide emissions from low bulk density heaps (0.23 kg/l) were low (Sommer & Møller, 2000), possibly because NH_4^+ concentrations inside the heap were kept constantly low by the high air convection stripping NH_3 from the heap. With high density pig manure heaps (0.44 kg/l), high N_2O emissions were observed following the temperature decline (Sommer & Møller, 2000), and this was also the case in studies with compost being turned weekly or several times a week (Czepiel et al. 1996; Hellman et al. 1997), as well as in a study with undisturbed solid pig manure (Petersen et al. 1998b).

Petersen et al. (1998*b*) recorded depth profiles of N_2O which indicated that the N_2O emitted was mainly produced near the surface of the heap (Fig. 3). Also, N_2O emissions from the composting pig manure appeared to be influenced by climatic conditions. There is no direct connection between N_2O emissions and the concentrations observed inside a manure heap (Sommer, 2001; Petersen et al., 1998*b*). Nitrous oxide produced at greater depths inside the heap may be reduced during the transport towards the surface and thus not emitted, and generally emissions will be a function of production, consumption, and the air exchange rate. Czepiel et al. (1996) found that in a 9–day-old compost high N_2O concentrations were found 0-20 cm from the surface, while in a 38-day-old compost high concentrations extended to 50 cm depth.

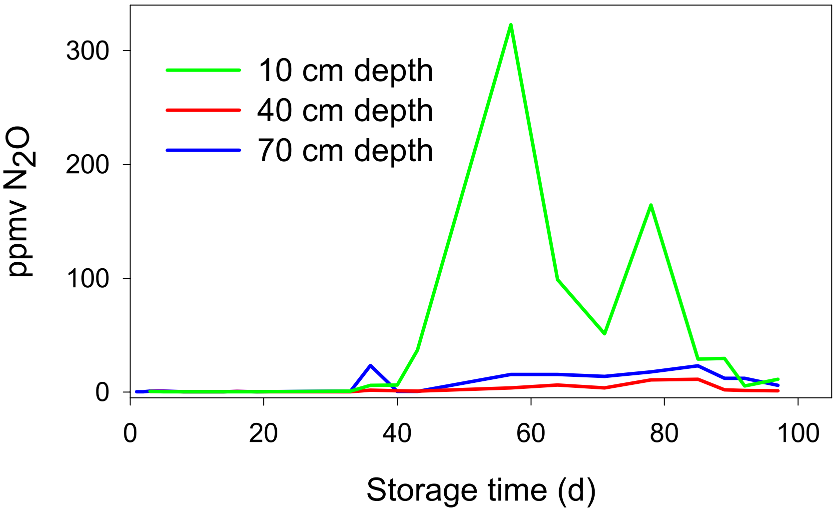

Figure 3. Concentrations of N_2O at different depths in a composting heap of solid pig manure. After 40 days of storage, the temperature at 70 cm depth had dropped to 40°C (adapted from Petersen et al. 1998b).

Studies have indicated that N_2O emissions from composting manure may be in the range 7–27 g N t^{-1} or 0.1–1% of total N (Czepiel et al., 1996; Petersen et al., 1998b, Sommer & Møller, 2000). German studies quoted by Hellebrand & Kalk (2000) suggest that N_2O emissions may account for up to 6% of total N in a compost of garden waste. Compacting compost appears to increase N_2O emissions due to poorer aeration (Sommer, 2001), although very dense heaps with no air exchange will probably not be a source of N_2O.

Algorithms for estimating N_2O emission

The IPCC methodology calculates N_2O emissions from manure on the basis of total N content and climate region only. In other words, the methodology assumes net production of N_2O throughout the entire volume of stored manure.

There are few studies of the emission of N_2O from slurry stored in animal houses, but the results we have referred to above indicate that the emissions are related to the soiled surface area. Studies have also shown that N_2O emissions from outside stores with slurry will be produced in the surface crust and thus are related to the surface area of the store, rather than the volume. Further, these emissions were related to the water balance. It thus appears that there is a need for revising the IPCC model for calculating emissions from management of liquid manure.

For stored solid manure, recent measurements have indicated that N_2O emissions from cattle houses with solid floors covered with litter are low, while the emissions from pigs on deep litter are significant. However, more studies are needed to confirm the emissions recorded. The emissions from solid manure stored in heaps will be related to the aeration and potential for composting. During composting, N_2O production will be restricted to the surface of the manure heap, while before and after this phase, N_2O may be produced throughout the heap provided the manure is well aerated. Thus, there is a need to differentiate between manure that is composting during storage and very compact manure heaps that do not compost during storage. We therefore suggest that it should be attempted to link N_2O emissions from solid manure to both the surface area of the heap and to the heap bulk density.

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A critical analysis of nitrous oxide emissions from animal manure

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Summary

Emission of nitrous oxide, N_2O*, after manure applications to agricultural soil is composed of two components. The first is the immediately increased potential for* N_2O *production due to favourable conditions in the manure-soil environment. More* N_2O *is produced and emitted when the nitrogen content of the manure is high, especially the mineral nitrogen content. The amount of carbon available for microbiological decomposition and water content regulate the oxygen availability, which is important for* N_2O *production in both nitrification and denitrification. The balance between mineralisation of organically bound nitrogen and immobilisation of mineral nitrogen by microorganisms and plants control the availability of N for* N_2O *production. The initial burst of* N_2O *to the atmosphere following manure application may last for two months, while a second component is long-term and due to nitrogen in organic matter accumulating in the soil, resulting in a small increase in background emissions over many years due to nitrogen cycling. The IPCC emission factor for* N_2O *emission due to manure addition accounts for the increased emission of* N_2O *during the first year, whereas the long-term emission is not included.*

Introduction

The largest global source of N_2O, representing 80% of the N_2O entering the atmosphere annually, is biological production by bacteria in the soil (Isermann, 1994). Two bacterial processes are responsible, both of them dependent on nitrogen (N). Nitrification is the first of these processes; it naturally utilises the ammonium released by the breakdown of organic material. The process gains energy from the oxidation of ammonium (NH_4^+) to nitrite (NO_2^-) and nitrate (NO_3^-) and uses carbon dioxide as carbon source. To gather enough energy for metabolic processes, large amounts of ammonium have to be converted. Optimal conditions for nitrification imply that oxygen is available, and pH should not be too low. Nitrous oxide is a side-product which is produced in larger quantities when conditions are suboptimal for nitrification, for example when oxygen is deficient, as in wet soils, or in situations with high biological activity consuming oxygen. Nitrification is responsible for a continuous background emission of N_2O from many soils, and it is a prerequisite for the second process: denitrification.

Denitrification is performed by bacteria with the ability to decompose organic materials both aerobically and anaerobically. When oxygen is lacking, nitrate (or nitrite) is used instead of oxygen in the respiration process. Nitrous oxide is formed as an intermediate product in a step-wise process, where nitrogen gas (N_2) is the end product. When oxygen is available at low concentrations, the process is restrained and N_2O becomes the end product. During oxygen-deficient conditions denitrification also produce the largest amounts of N_2O. High nitrate concentrations also increase the N_2O production from denitrification.

Nitrous oxide is produced in all soil ecosystems with available mineral N, but the emission varies a great deal over the year. In spring, when the temperature rises and water is available, the mineralisation of soil organic matter is one major cause for an increased emission. Occasions with increased emission have also been found during winter-time with temperatures below 0°C. Thus, N_2O emission is highly variable, and the question is: what is a natural emission level without human influence?

Most studies of emissions from different systems have been made on agricultural land after addition of mineral fertiliser, while fewer studies have been made after addition of manure. This give rise to the following questions:

- Does addition of animal manure cause a different rate of N_2O emission compared to mineral fertiliser N?
- Over what time span can the addition of N be expected to increase the N_2O emission?

Use of animal manure in Swedish agriculture

In Sweden, as in all Nordic Countries, animal manure is an important nutrient source for the agriculture. In Sweden, about 200 000 t plant available (inorganic) nitrogen are used annually in the agricultural sector, and in 1999 animal manure constituted about 15% of this amount (Statistics Sweden, 2000). Besides, animal manure contains considerable amounts of organic N, which is added to the soil as well. During 1999, about 50% of the crop area received only mineral fertiliser, 10% received only animal manure and 25% was treated with both manure and mineral fertiliser. The areas receiving both manure and mineral fertiliser were given considerably higher N doses than those receiving only mineral fertiliser, on average 130 kg plant-available N ha^{-1} yr^{-1} as compared to 90 kg N ha^{-1} yr^{-1}. Including also the organic N applied in manure, doses were even higher, about 180 kg N ha^{-1} yr^{-1}, which is approximately twice the recommended dose.

IPCC Emission factors

The emission factor for direct emissions of N_2O from fertilised agricultural soil that is presently recommended by IPCC is based on a compilation made by Bouwman (1996). This review included studies from the USA and UK conducted during a whole year, to include emissions across all seasons. This data set is shown in Fig.

1. Of the studies included, 25% used both organic and inorganic N fertiliser. A linear relationship between N_2O emission and addition of N was found, which indicated that 1.25±1.0% of the N added was emitted as N_2O during one year.

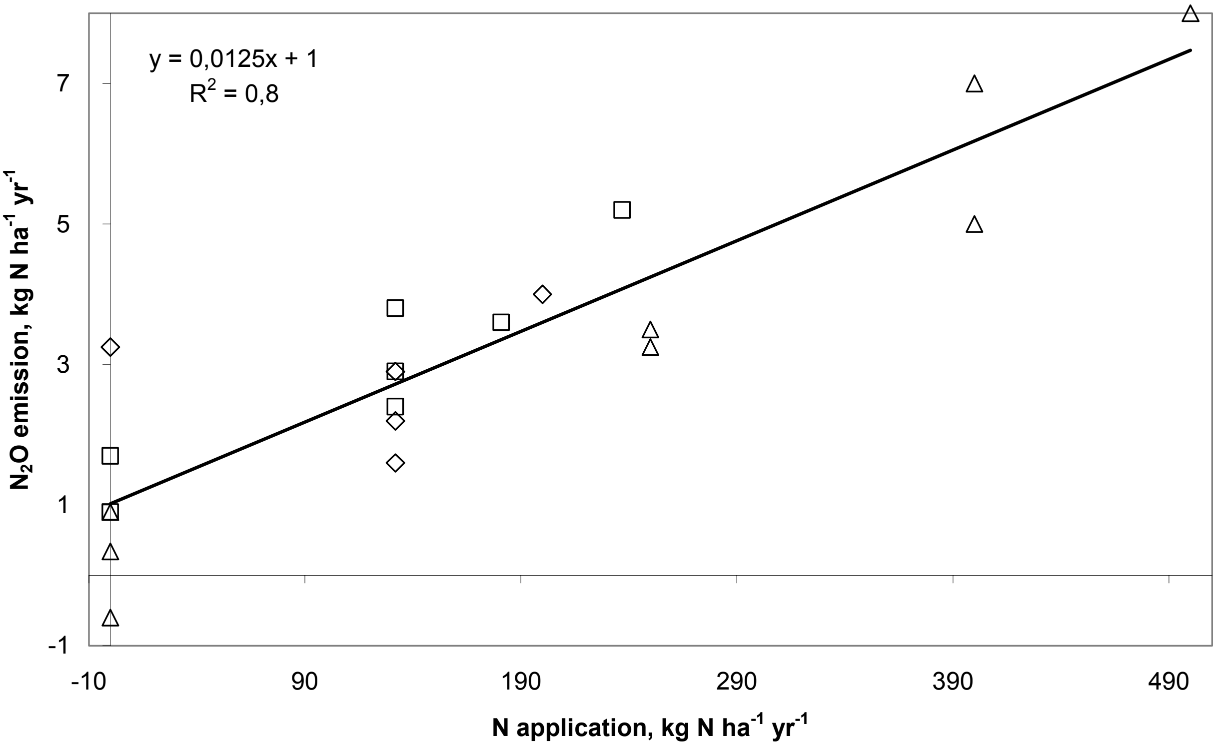

Figure 1. The data on which the IPCC emission factor for direct emissions from fertilised soil relies (Bouwman, 1996). Nitrous oxide emissions from mineral soils with different non-leguminous crops and N-additions. Squares: Both organic and inorganic N to annual crops; Rhombus: Inorganic N to annual crops; Triangles: Inorganic N to grass. Grassland data are from the UK, and the rest of the data from the USA.

From the relationship in Fig. 1 it can be inferred that soils receiving no N also emitted N_2O, around 1 kg N_2O-N ha^{-1} yr^{-1}. In the IPCC Guidelines (IPCC 1997), emissions from unfertilised fields were considered to be background emissions and not to be included among the anthropogenic sources which should be reported to the UNFCCC.

The IPCC methodology also includes indirect N_2O emissions from N lost by NH_3 volatilisation and subsequently deposited on agricultural land or in other ecosystems. The emission factor is in the same order of magnitude as for direct emissions, i.e., 1% of the N deposited. Leaching of NO_3^- will also give rise to N_2O on its route to the sea, in total 2.5% of the leached N. All countries are recommended to use the IPCC methodology for greenhouse gas inventories, unless a more appropriate and documented national methodology is available.

N_2O emission after manure addition

Fertilisation experiments on agricultural soils show that N_2O emission increase more after manure application than after addition of mineral fertilisers (Clayton et al., 1997). Many factors influence the amount of N_2O emitted, such as manure composition, weather conditions and soil content and structure (see Table 1). The composition of manure is variable, depending on storage system etc. Besides water and mineral nutrients, manure contains carbon and nitrogen in different forms (degradable and stabilized organic materials, NH_4^+ and organically bound N). Different investigations have shown different factors to regulate N_2O emissions, but generally inorganic N (NH_4^+/NO_3^-), available carbon and a high water filled pore space (WFPS) have been found to increase the potential for N_2O from nitrification and denitrification. Microbially available organic carbon seems to be important in regulating N_2O emissions shortly after manure addition in situations where denitrification contributes significantly to N_2O production (Clemens & Huschka, 2001).

Table 1. Factors influencing N_2O emissions from manure-amended soil.

	Increase of N_2O in a short-term perspective	Reference
Manure composition		
Biologically available organic carbon	X	Clemens & Huschka (2001)
Total and mineral N content	X	Clemens & Ahlgrimm (2001), Kebreab et al. (2001)
Water		
Soil conditions		
Texture	Clay	
Soil moisture	60-80% WFPS	
Carbon and nitrogen content of topsoil	Manure and crop residues increase the content.	Mogge et al. (1999)
C/N ratio	Indicates substrate availability for microbial N turnover	
pH	Positive influence on the overall activity, but negative on N_2O/N_2 ratio.	Granli & Bøckman (1994)
Inorganic N content	Increased nitrification and denitrification, NO_3^- increases the N_2O/N_2 ratio.	Granli & Bøckman (1994), Ruser et al. (2001)
Temperature	If N or other conditions are not limiting	Clayton et al. (1997)
Timing and Management		
Crop type	Vegetables	Ruser et al. (2001)
Total N applied	Above crop N uptake capacity	
Application scheme	Single application	Clayton et al. (1997)
Timing of application	Wet weather	Ferm et al. (1999)
Spreading technique	Incorporation > surface application	Ferm et al. (1999)

A typical pattern is an elevated N_2O emission level of varying duration after manure application, but mostly the emission rate is back to "normal" within 2 months. This initial "burst" of N_2O has often been interpreted as the total effect of manure on the N_2O emission. For example, Chadwick et al. (1999) based their emission factors on measurements of N_2O emission during the period from N addition until emissions for treated plots had returned to the emission level found on untreated plots. But this will only reflect the initial phase of N_2O emission, the extent of which depends on the composition of the manure, soil and weather conditions. The long-term effect is difficult to discern from the emission "noise" associated with the background emission. An important question is: What is a "background" emissions for agricultural land? A question to which we have no answer.

The soil environment is complex, and mostly mineralisation/immobilisation controls how much mineral N is available for nitrification and denitrification. Immobilisation by living organisms and plant roots reduces the potential for N_2O production. The degree of N saturation in a terrestrial system, as reflected in the accumulation of N in biomass and soil organic matter, will determine the potential for nitrification and denitrification in the long run.

It is possible to spread manure at times and with techniques that give smaller initial emissions of N_2O. But some mitigation measures may lead to increased NH_3 emissions and/or NO_3^- leaching which in turn give rise to indirect emissions of N_2O. These indirect emissions are difficult to quantify, and generally they are calculated by use of standard emission factors (Weslien et al., 1998). The size and reliability of emission factors for indirect emissions of N_2O are crucial for suggestions of management strategies to mitigate the emissions.

New emission factors based on Northern European data

Emission factors for N_2O emission from agricultural soils, including direct emissions after mineral fertilisers and manure addition to both organic and minerogenic soils, as well as indirect emissions, were scrutinised by Kasimir Klemedtsson (2001). The aim was to evaluate the IPCC methodology in a Swedish perspective.

Long-term studies of N_2O emissions are scarce, especially for organic amendments to soil, and also for organic soils. For this review, data from measurements conducted during at least eight months north of 50°N latitude in Europe were selected. We based the work on own Swedish studies, on data from the literature, and on data gathered in the reported EU concerted action "Biogenic emissions of greenhouse gases caused by arable and animal agriculture, FAIR3-CT96-1877" (Freibauer & Kaltschmitt, 2000). The results are summarized in Figs. 2-4.

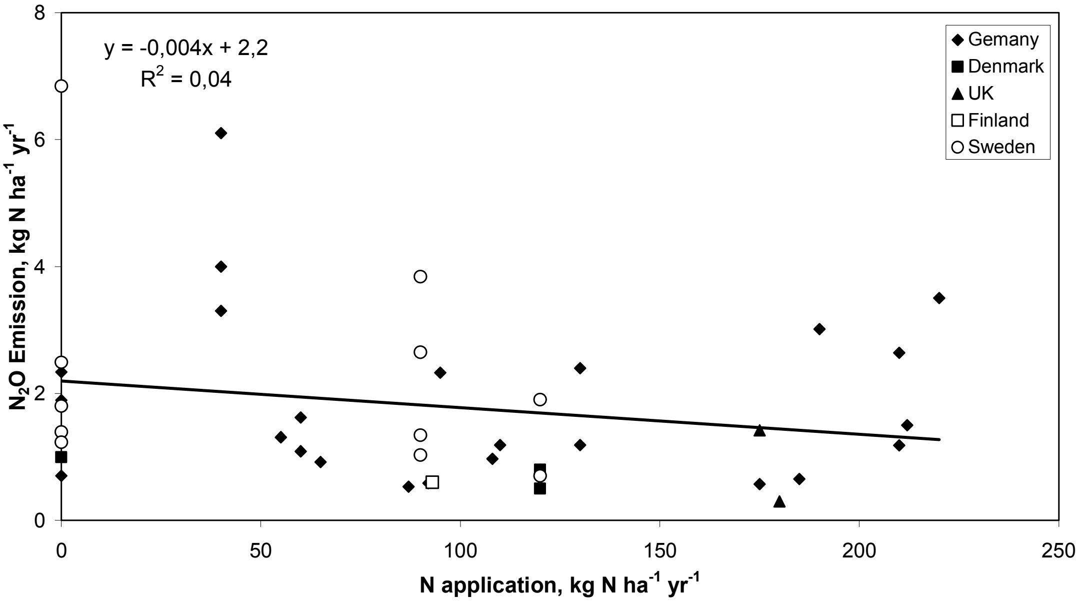

Figure 2. Nitrous oxide emissions with mineral fertiliser-N applied to cereals (n=43). Data from Kaiser et al. (1998), Kaiser & Heinemeyer (1996), Ernst (1997), Flessa et al. (1998), Jørgensen et al. (1997), Christensen (1985), Yamulki et al. (1995), Smith et al. (1998), Jaakola (1994), Kasimir Klemedtsson et al. (in prep), Röver et al. (1998).

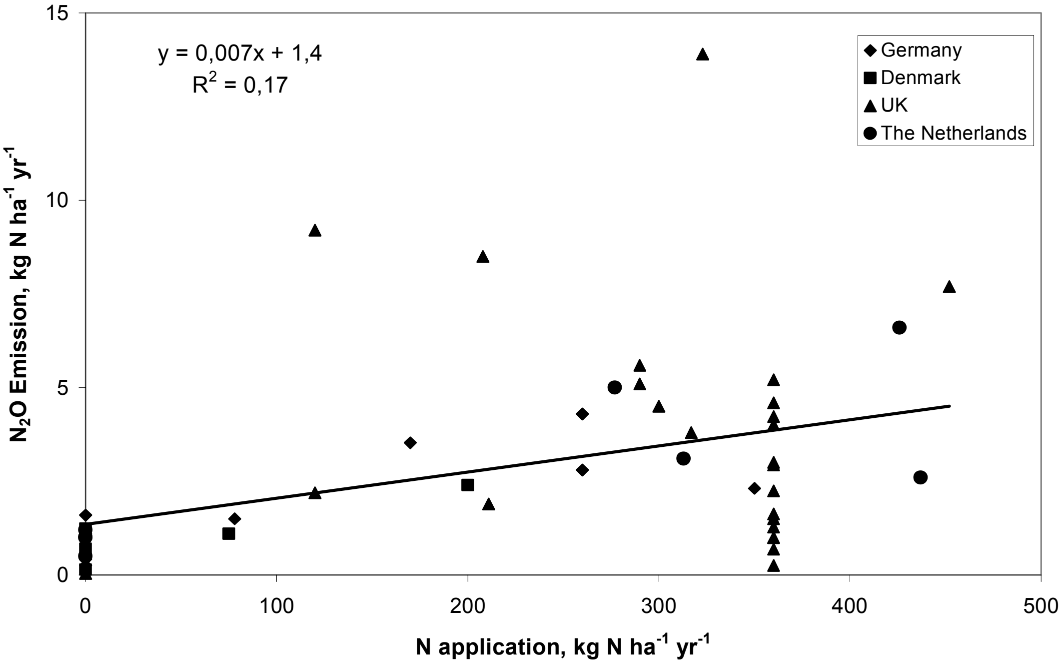

Figure 3. Nitrous oxide emissions with mineral fertilisers applied to grassland (n=55). Data from Ambus & Christensen (1995), Allen et al. (1996), Christensen (1983), Clayton et al. (1997), Dobbie et al. (1999), Duyzer (1996), Flessa et al. (1998), Heinemeyer et al. (1996), Jørgensen et al. (1997), Kaiser et al. (1998), Mogge et al. (1999), Poggeman et al. (1997), Schmädeke et al. (1997), Smith et al. (1998), Velthof et al. (1996) and Vermoesen et al. (1996).

In contrast to Bouwman's (1996) findings, no linear relationship between N_2O emission and N addition rate was found for mineral fertiliser addition to cereals (Fig. 2). Comparing data on inorganic N in Fig. 1 with the data in Fig. 2 reveals no difference in the magnitude of emissions with N addition. However, a small influence of the N application rate was found for mineral fertiliser added to grasslands, Fig. 3, which indicated that 0.7% of the N added was transformed to N_2O in one year. For addition of N to organic soils too few data were available to generate a separate emission factor. The suggested emission factor for N_2O derived from mineral fertiliser N under Swedish conditions was 0.8% of the added N, irrespective of soil and crop type (Kasimir Klemedtsson, 2001).

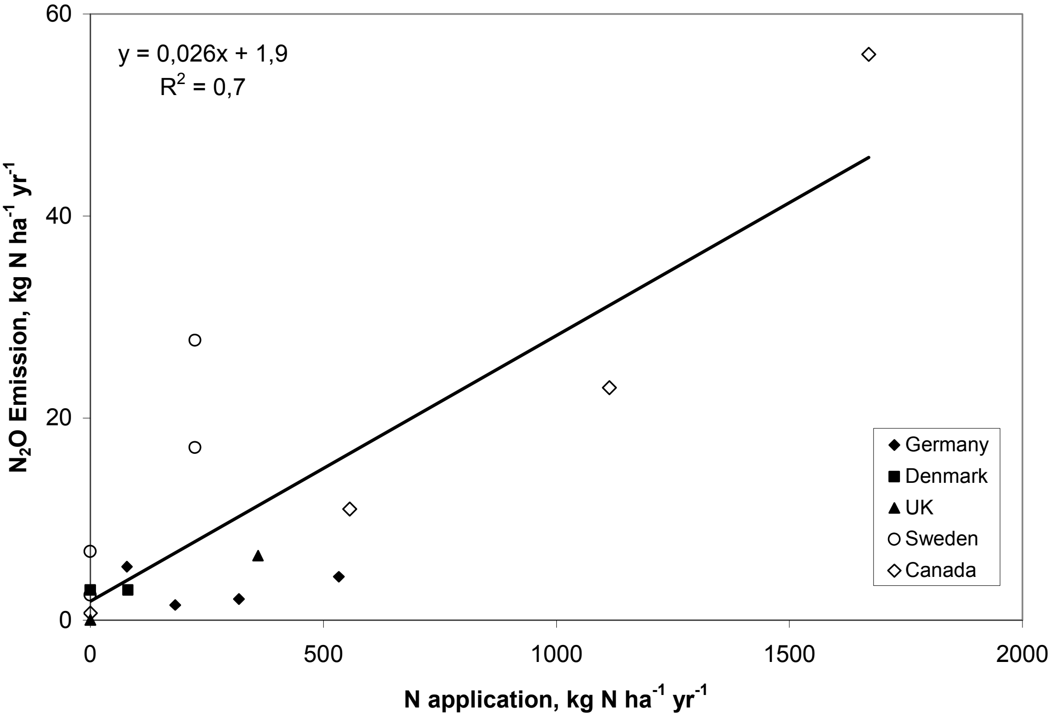

Figure 4. N_2O emissions with addition of N in organic fertilisers to cereals and grassland (studies from both Northern Europe and Canada, n=16), expressed as total N in the manure. Data from Ambus et. al. (2001) Chang et al. (1998), Clayton et al. (1997), Heinemeyer et al. (1996), Kasimir Klemedtsson et al. (in prep), Mogge et al. (1999) and Poggemann et al. (1999).

Very few studies of N_2O emission after addition of organic fertilisers have been conducted during sufficiently long time to be used for generating emission factors. The small data set (n=12) for manure addition in Northern Europe showed variable results (see Fig. 4). Therefore, data from Canada were also included, and the combined data set (n=16) gave an emission factor of 2.6%. Eventually, 2.5% was suggested as a new emission factor for organic N (manure) additions (Kasimir Klemedtsson, 2001).

It is obvious that N_2O emissions are not strictly related to additions of N, especially not inorganic N, to annual crops. However, separating data for grasslands from data for annual crops, and data for organic fertilisers from data representing inorganic N additions, resulted in stronger relationships between N added and emissions of N_2O.

Long-term effects of N additions

A linear relationship between N addition and N_2O emission is not always seen in field experiments. The reason for this may be that preceding crops and earlier soil amendments can have a large influence on the emission, sometimes larger than the effect of the most recent addition of fertiliser N (Kaiser et al., 1998). During 1995-97 we conducted measurements of N_2O emission after addition of calcium ammonium nitrate at a rate of 120 kg N ha^{-1} to agricultural land in Southwestern Sweden (Kasimir Klemedtsson et al., in prep b). Measurements were performed by use of static chambers and gas analyses by GC, and the results are shown as the three first bars for each month in Fig. 5 and summarized in Table. 2. The emission was about 2 kg N_2O-N ha^{-1} yr^{-1} in both fertilised and non-fertilised plots. Of the added fertiliser N, only 0.2% was emitted as N_2O, but this number was not significantly different from zero. Thus, the emission was not related to the N addition, and the conclusion that can be drawn is that the soil fertility and management history of the soil controlled the emissions.

Table 2. N_2O emissions measured in Southwest Sweden. Measurements were performed by use of static chambers and gas analyses by GC.

Treatment kg N in pig slurry/ kg N in fertiliser	Nitrogen application, kg N ha^{-1} yr^{-1}	N_2O emission, kg N ha^{-1} yr^{-1}, average	N_2O emission, kg N ha^{-1} yr^{-1}, median	Increase due to added N, %
No N	0	2		-
Fertiliser, broadcasted	120	2		0.2
Fertiliser, drilled	120	2		0.2
0 (pig slurry)/0 (fertiliser)	0	1.5	0.8	-
0/0, catch crop	0	3.6	1.1	-
0/90	90	1.6	0.7	0.1
0/90, catch crop	90	2.6	1.1	-1.1
90/45	135	5.1	2.5	4.0
90/45, catch crop	135	2.8	1.9	-1.0
180/45	225	10.2	1.0	4.8
180/45, catch crop	225	19.1	6.0	8.6

Other fields close to the above-mentioned experiment were given different, but continuously the same fertiliser and manure additions during 17 years (Torstensson & Aronsson, 2000). Measurements of N_2O emissions were conducted during 1998-2000. Again, a mineral fertiliser addition of 90 kg N ha^{-1} yr^{-1} did not result in higher emissions than control plots receiving no other N inputs than the atmospheric deposition (ca. 20 kg N ha^{-1}) which, of course, all sites received (Table 2 and Fig. 5). In contrast, addition of 180 kg N ha^{-1} as pig slurry together with 45 kg N ha^{-1} in mineral fertiliser resulted in a higher emission. The cause for the emission peak in May, after the pig slurry amendment, may be both the addition of N and accelerated mineralisation of soil organic matter due to relatively high soil temperatures and soil cultivation. Moreover, the crop was too small to effectively take up mineralised and/or added N, and altogether this resulted in a higher risk for nitrification and denitrification in the spring.

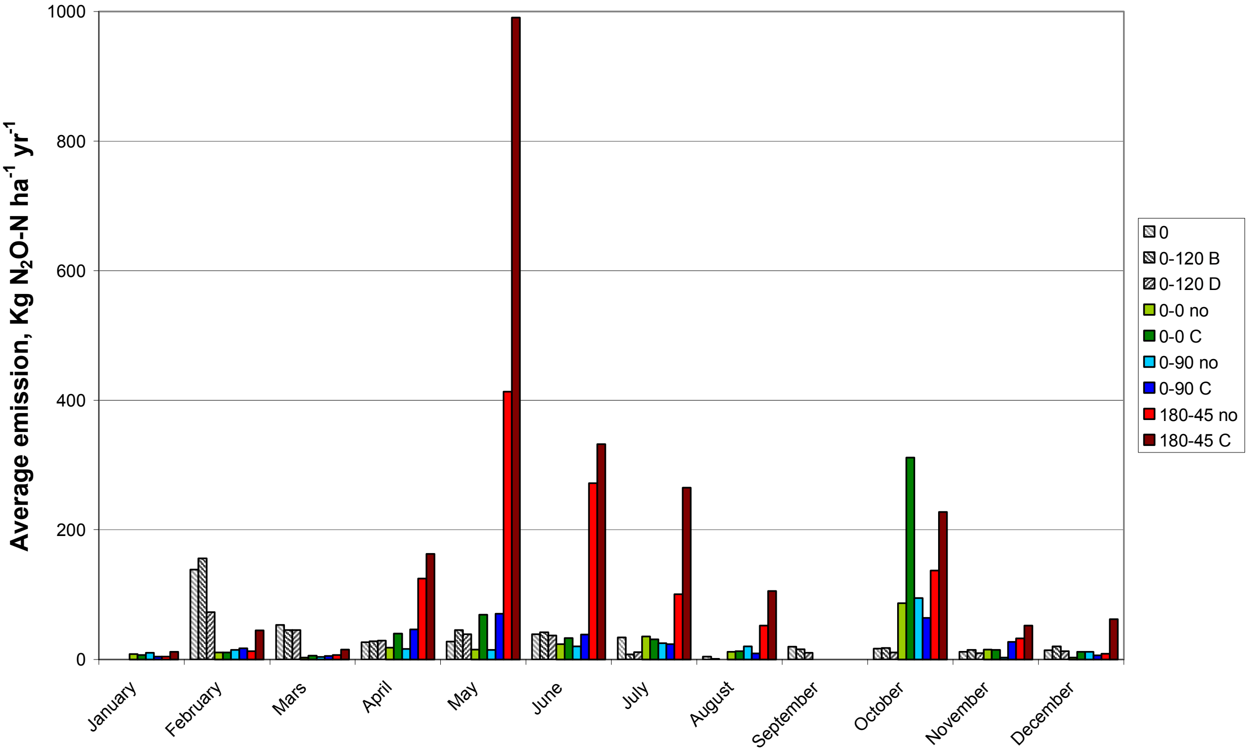

Figure 5. Average N_2O emissions for different months at Mellby in Southwest Sweden. The first three bars represent experiments from 1996-1997 with addition of calcium ammonium nitrate (120 kg N ha^{-1} yr^{-1}, either broadcasted or drilled) and no N addition, respectively. The last six bars represent a nearby site with three treatments, i.e., no N addition, 90 kg N ha^{-1} in mineral fertiliser, and both 45 kg N ha^{-1} in mineral fertiliser and 180 kg N ha^{-1} in swine slurry. The treatments were with or without perennial ryegrass, which each year was ploughed down before seeding in April.

Catch crops gathered N during the no-crop season, which decreased leaching losses. But during spring and summer, treatments with both manure and a catch crop had almost twice the mineralisation rate compared to plots with manure addition but no catch crop (Hessel Tjell et al., 1999), which can explain the higher N_2O emission from the fields with a catch crop. Generally, addition of animal manure to soils results in an increased content of carbon and nitrogen in the top soil (Peacock et al., 2001). At the Swedish site, manure addition also resulted in higher N retention in the soil, +114 to +128 kg N ha^{-1} yr^{-1}, in contrast to -16 to +15 kg N ha^{-1} yr^{-1} when mineral fertiliser was applied. The spans represent fields without and with a catch crop, respectively (Hessel Tjell et al., 1999).

The amounts of N retained were small compared to the organically bound soil N content of about 8000 kg N ha^{-1}, which probably determined the size of the background emission. But since this management had continued for 17 years, it is possible that accumulation of soil N derived from the manure had increased the "background" emission over the years. Thus, the N_2O emission peak in May was the result of N addition and mineralisation of organic matter occurring simultaneously. An emission factor for the manure addition was calculated to be 5 and 9% of the annual N addition without and with a catch crop, respectively, i.e., much higher figures than the IPCC emission factor of 1.25%. Thus we still need to separate the part of the emission that belongs to the annual N application, and to divide the background emission into a general background and a background that stems from previous fertiliser and manure applications.

Suitability of the IPPC approach for estimating agricultural emissions in the Nordic countries

Mineral fertilisers and manure have different effects on soil organic matter; only manure will cause an accumulation of N in the soil, which is important for the size of the N_2O emission in a longer perspective. Also, the above mentioned EU concerted action programme (Freibauer & Kaltschmitt, 2000), using a much larger data material than that available to Bouwman (1996), found N_2O emissions to be stronger related to the soil N content than to the annual fertiliser addition rate. The effect of organic fertiliser additions was difficult to evaluate, since too few data were available, but a tendency towards higher emissions was found for organic fertilisation compared to addition of mineral fertilisers. Also, Ruser et al. (2001) showed the importance of the soil N status in governing the N_2O emission, where soil nitrate appeared to be a more important factor in determining the emission than the fertiliser addition. All these results indicate that the total nitrogen flow has a strong influence on N_2O emissions, more important than recent N additions.

In this text, the term "N addition" represents spreading of N-containing substances in the agro-ecosystem. But on a larger scale, nitrogen can only be added to the ecosystems by nitrogen fixation - biological, industrial or during combustion. Thus, mineral fertilisers represent net additions of N, while manure addition only represents N recirculation. Since the introduction of commercial fertilisers in the 1950's, nitrogen enrichment of the ecosystems has been in progress, leading to a higher fertility of agricultural and forest soils and eutrophication of the environment. The effects of the N enrichment on N_2O emissions have been modelled by Bakken & Bleken (1998), who concluded that the IPCC emission factor of 1.25% of added N converted to N_2O-N in one year is an overestimation, but that it may estimate present-day total N_2O emissions fairly well anyway. But the factor will probably underestimate future emissions, if the industrial and biological N fixation continues at the current rate. Schmid et al. (2001) drew the same conclusion that the current IPCC method underestimates the long-term effects of fertiliser management.

In the IPPC methodology an anthropogenic "background" emission is considered for <u>organic soils</u>, and here the emission depends on mineralisation of the soil material due to drainage. Measurements on organic soil in Sweden showed the lowest N_2O emission for permanent grass, while soil cultivated for cereals and vegetable production have larger emissions (Weslien et al., in prep, Kasimir Klemedtsson et al., in prep a). But, unlike the IPCC methodology, we suggest a background emission factor to be used also for mineralisation of soil organic N in cultivated <u>mineral soils</u>. For Nordic conditions an emission of 0.5 kg N_2O-N ha^{-1} yr^{-1} is proposed, which may compensate for the lack of an emission factor for the long-term effect due to N additions to the soil in the past.

Conclusions

The potential for emissions of N_2O is higher after addition of manure to soil than after addition of mineral fertilisers. This is probably due to the organic content of the manure resulting in higher soil respiration consuming oxygen which, in combination with a moderately high content of water and mineral nitrogen, results in favourable conditions for N_2O production. The addition causes a temporarily increased emission above the background level, an increase which is the basis for the emission factors proposed by IPCC. The IPCC concept is thus based on short-term effects of annual N additions resulting in N_2O emission over less than one year. New investigations and inventories have shown the soil N content to be of greater importance than the recent N addition in governing the N_2O emission. Thus the land use history gives a "background" emission, which can persist for many years. It is probably difficult to determine this increased background emission by field measurements, but modelling the long-term effect is a possible way out.

We can conclude that there are still many questions to be answered about the effect of manure applications on N_2O emissions, and what the best mitigation options are. Is it possible to add manure to soil in a way that avoids a stimulation of N_2O emission? Or is it more fruitful to modify the feed intake of the animals or treat the manure in order to decrease the potential for N_2O emissions after spreading on agricultural land? Or should the abatement focus on the total N-use of the society?

To answer all these questions, long term field studies, site modelling, and modelling including the society level is needed. Furthermore, there is a need for improvement of the IPCC methodology in these respects.

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Nitrous oxide emissions from field-applied fertilizers

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Summary

Nitrous oxide emissions originating from the use of fertilizers, as from other nitrogen sources in agriculture, result from complex biological processes in the soil. Their quantification is subject to high uncertainties, the emission rates are very variable and the responsible processes are difficult, though not impossible, to influence.

The current IPCC methodology for calculating N_2O *emissions from fertilized agricultural land use an emission factor of 1.25% ± 1% of applied fertilizer as the only controlling variable. This approach has the benefit of administrative simplicity, but it also gives the impression that the only way to reduce* N_2O *emissions is to reduce the N input. The current factor was based upon 20 annual emission estimates available by 1994 (Bouwman, 1996). Today, more data are available, making it possible to reevaluate the existing factor as well as the usefulness of such factors.*

Our analysis, as well as other similar studies, indicated that a global emission factor of 1.25% is too large; including more data lead to a lower factor which varied with type and amount of data included in the evaluation, but centered around 0.8%. The geographical distribution of these studies was very uneven, hence any global emission equation should be regarded as provisional until this deficiency has been corrected. Further, including more data resulted in a wide scattering of emissions, where nitrogen application rates explained less than 15% of the variation, compared with more than 50% in the smaller original data set upon which the IPCC default value is based. Such wide scattering is expected because emissions from a field result from biological processes, which again depend on numerous factors such as soil temperature, water content, degradable organic matter, soil texture etc. in addition to nitrogen availability. This points to the potential for reducing emissions through improved management of agricultural operations, rather than merely reducing nitrogen inputs and thus yields.

A more realistic model that includes the main drivers should be developed and validated. Such a model would be a useful tool for identification of farming practices that may reduce N_2O *emissions. Many studies on* N_2O *emissions exist, but they vary with regard to crop, fertilizer type, and management, they often cover only part of the year, and information on important drivers is commonly lacking. This reduces their usefulness for statistical treatment and modeling, and more closely supervised long-term field trials are thus recommended.*

Introduction

In contrast to N_2O emissions from the use of fertilizers, emissions from fertilizer production can be estimated accurately, and technologies are now under development that may reduce emissions from production facilities to a small fraction of the present level. The dominant sources in the future will thus be agricultural operations, hence it is important to get accurate estimates of emissions also from these sources, and to identify measures that may reduce emissions.

The current guidelines (IPCC, 1997) for calculating national direct N_2O emissions from the application of N fertilizers and animal manure include a default global emission factor of 1.25% (0.25-2.25%) for N fertilizer-induced emissions. An emission factor for N fixed by legumes is also included, as well as a method to calculate the indirect emissions from N lost from the soil-plant system by leaching and ammonia volatilisation. The latest IPCC guidelines (IPCC, 2000) allow countries to use national emission factors based on local measurement data. However, national emission factors require documentation and are subject to review by UNFCCC. Further, emission factors are also introduced to account for the N contribution from crop residues and green manure crops.

The use of such emission factors has the advantage of being administratively simple, but this approach also implies that the only way to reduce N_2O emissions from agriculture is by reducing the N input. Fertilizer use is expected to increase along with projections for population growth. Hence, it is important to identify management practices that may reduce the N_2O emissions per unit of food produced. A large number of publications on the effect of farm management practices on N_2O emissions already exist, which stem from locations with varying conditions with respect to soil, climate and management, different crops and fertilizer types etc. In addition, annual variation in weather conditions leads to large variation in emission rates from the same locality. Unfortunately, details about important drivers are often lacking. This makes published N_2O emission data difficult to interpret. Statistical analysis and modeling are therefore the most appropriate tools. The objective of this work was to evaluate the usefulness of statistical tools for identification of main drivers for N_2O emissions, and how these are influenced by management practices. The long-term objective of the research is to define those management practices that reduce N_2O emissions to a minimum.

Materials and methods

Data selection

We have used three datasets that differ in numbers of observations included, where observation here refers to a series of measurements covering one set of experimental conditions which is carried out over a given period of time. Dataset 1, compiled by the authors, consisted of about 300 observations collected from published studies covering the period 1992-1999. Among these, 114 observations came from measurements that lasted for one year or more, but no measurement periods were shorter than two months. Dataset 2, compiled by IFA/FAO (2001) and Bouwman et al. (2002), comprised more than 900 observations published since 1980 covering a wide range of measurement periods, of which about 240 last for one year or more. Dataset 3 was taken from Kaiser & Ruser (2000) and covered about 100 observations from Germany only. All observations in Dataset 3 were annual emissions covering the period 1992–1996. Most, but not all, of the observations in Datasets 1 and 3 were included in the larger Dataset 2.

Organic soils, legumes and grazed grassland, as well as fertilizer rates above 500 kg N ha^{-1} y^{-1} were excluded in accordance with Bouwman's earlier selection criteria (Bouwman, 1996). Both of the Datasets 1 and 2 – but particularly the larger Dataset 2 - were inhomogeneous and unbalanced, e.g.:

- the observations cover a span of many years, but the individual observation periods last from only a few days to a few years;
- the geographical distribution and distribution over climates is uneven: 75% of long-term observations (one year duration) are from Europe. Further, experiments on bare soil and maize have mainly been done in North America, rice experiments dominate the tropics, while experiments on grassland, wheat, barley and vegetables have mainly been done in Europe;
- there is an unbalanced distribution of fertilizer types and amounts within and between the various crop types;
- information on management practices apart from fertilizer application rate is sparse;
- information on climatic conditions during the measurement period is variable.

For Dataset 2, observations shorter than 50 days were excluded from further analysis because they deviated from the others, with considerably higher emission rates and also a higher range of variation (Lægreid & Aastveit, 2002). The reduced Dataset 2 contained about 580 observations, of which 224 lasted for one year or more.

Statistical analyses

Data included in the statistical analyses were:

- duration of observation: ≥50, ≥150, ≥300 days, ≥1 year, or 50-150, 151-300, >300 days;
- soil factors (categories): texture (sand, silt, loam, clay), drainage properties (good, poor), pH (<6; 6-7.5; >7.5), organic carbon (<1.5; 1.5-3; >3%);
- crop type: bare soil, grass with or without legumes, cereals with or without further sub-divisions, vegetables consisting of several different crops including also crops such as oilseed rape and sunflower;

- fertilizer type: pure organic and mixed organic/mineral fertilizers were treated as separate groups or combined, mineral fertilizers were taken as one group or divided into NH_3-based, NH_4NO_3-based, NO_3-based, or further detailed into various types (e.g. urea, AN, CAN);
- management practices: fertilizer application method, irrigation, ploughing, no-till, crop residues incorporated;
- location: continents (North America, Europe, Australia/New Zealand, Tropics) or further subdivided into countries for Europe and six regions for the US;
- climate: annual precipitation.

Missing information was termed "unknown". Actual weather conditions and other soil conditions (e.g. soil water content and temperature) are known to be important modifiers of N_2O emissions, so for Dataset 1 attempts were made to include more relevant factors. However, in many instances information was absent or presented in a form, which was difficult to extract and incorporate into a worksheet.

The data were treated by use of linear models with the statistical programme SAS – for details, see Lægreid & Aastveit (2002). In addition, principal component analyses with cross validation of the model estimates (Wold, 1979) were run on Datasets 1 and 2.

There was more information available in dataset 3 than in the other two datasets, making a more detailed analysis possible:

- information on crop yield and N in/out balances was given;
- fertilizer application rate was related to estimated rates for optimal yield: 0, 0.5 and 1 times the optimum level was applied;
- N inputs from other sources than fertilizers were included, i.e., atmospheric deposition and mineralization of soil organic matter;
- the emissions during winter were also included.

Results and discussion

Global emission factors – N fertilizer induced emissions

No clear correlation pattern was found between any of the input factors and N_2O emissions when using principal component analysis. Linear regression analysis of dataset 2 showed a significant correlation between N input and N_2O emissions. However, N applied could explain less than 15 % of the variability in the N_2O emissions, indicating that other important drivers for N_2O emission need to be identified. Log-transformation of the data, which reduces the influence of extreme values, did not affect the importance of N rate. A plot of N_2O emission versus N input for observations lasting for one year or more gave a scattered picture (Fig. 1A). The trend line indicated that on average 0.9% of N applied was emitted as N_2O based on linear regression, while the corresponding value for a regression of log-transformed values was approximately 0.7%. IFA/FAO (2001) and Bouwman et al. (2002) reported an emission factor of 0.8% for the same data.

Plotting the average emissions of each 10 observations at their average N application rate reduced the large variation and improved the explanation without changing the slope (Fig. 1B). Mineral fertilizers dominated as N source, so including only mineral fertilizers gave the same result.

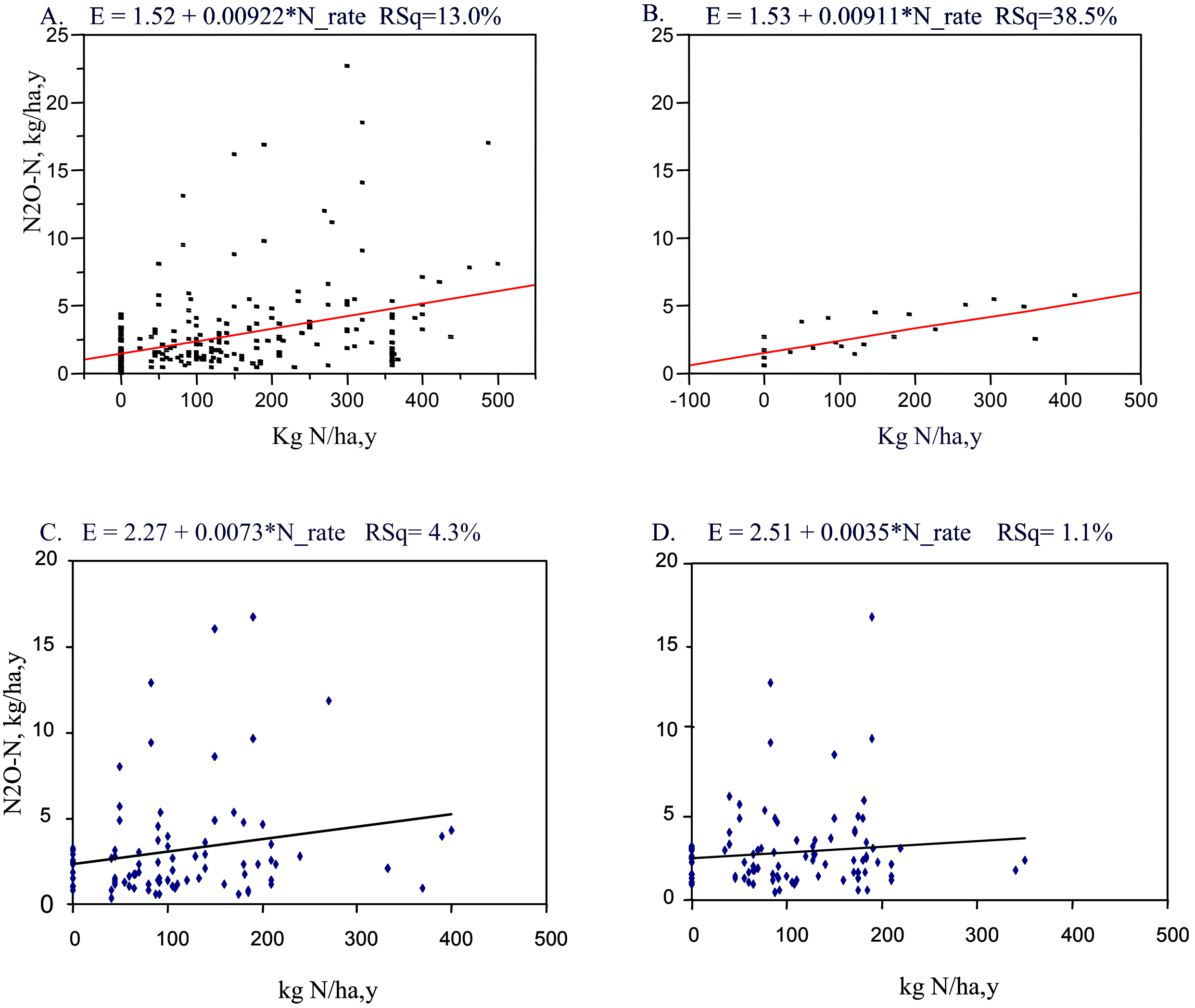

Figure 1. Plots of N_2O emission as a function of N applied for measurements lasting for one year or more. A: Linear plot from dataset 2; B: Like A, but taking averages of 10 emissions at their average N rate; C: linear plot of only German data from dataset 2; D: linear plot of data from dataset 3.

A linear regression plot of data set 1 gave an N-induced emission factor of 0.7%, with the same low degree of explanation.

Among the annual observations in dataset 2, 75% were from Europe, and half of these again were from Germany. Using a subgroup of dataset 2 including only observations from Germany (approx. 90 observations) gave an equally scattered picture, with an N-induced emission factor of 0.7% (Fig. 1C). Treating dataset 3 in the same way – excluding all legumes - resulted in an emission factor of 0.35% (Fig. 1D). The number of observations included was approximately the same for the two different German data sets. Most of the observations were also the same, but there were some deviations; dataset 3 contained slightly more observations on wheat and barley, while dataset 2 contained some more data on grass and other crops. Hence, the emission factor obtained depends on the data included. Similar scattered plots of N_2O emission versus N applied were found by others (Kaiser et al., 1996; Kaiser and Ruser, 2000; Kasimir-Klemedtsson, 2001).

The total emissions from agricultural land are higher than from pristine unfertilized fields. Some of the difference is due to increased N inputs (N fertilizer-induced emissions), but other factors (e.g., crop type, management practice and weather conditions) may also enhance N_2O fluxes from fields.

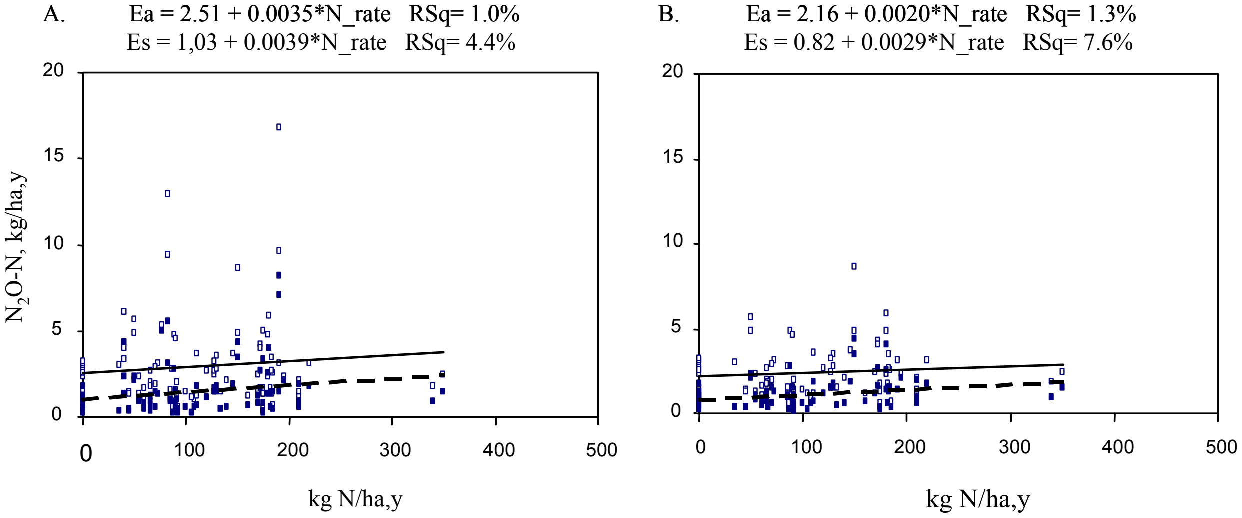

Figure 2. Annual (open symbols, fully drawn trend line) and summer (closed symbols, dashed trend line) emissions based on dataset 3. A: all data minus legumes; B: only mineral N.

Kaiser & Ruser (2000) included information on the fraction of the annual emissions which was being released during winter. The winter emission ranged from 7 to 89% of the annual emission, driven mainly by freeze/thaw cycles. The additional information made it possible to distinguish between annual and summer emissions, as shown in Fig. 2 for dataset 3 excluding legumes (Fig. 2A), and for mineral N only (Fig. 2B) where also green manure inputs are excluded. Including only mineral fertilizers reduced the emission factor further compared to when other N sources were included. However, in both cases the annual and the summer N fertilizer-induced emission factors were almost identical, but there is an upward parallel displacement of the trend line for the annual compared to the summer emission due to winter emissions. Winter emissions seemed to be independent of N fertilizer input.

The variability of N_2O emission factors depended on which data sets were included in the analysis. This is illustrated further in Table 1, where dataset 2 has been subdivided into regions/countries and observation periods.

*Table 1: Dataset 2; numbers of observations (n) and emission equations derived for continents and countries/regions in Europe, for various observation periods. First line: linear regression (E= a + b*N_rate); second line in italics: logtransformed regression ($E = 10^{a} * 10^{b*N\_rate}$; reduces influence from extremes).*

Location	n	Emission >50 d	n	Emission >150 d	n	Emission >300d
Europe	282	1.87 + 0.0119*N	227	2.1 + 0.009*N	170	1.64 + 0.0086*N
		_0.40 + 0.00090*N_		_0.41 + 0.00081*N_		_0.37 + 0.00084*N_
Germany	113	2.19 + 0.0169*N	100	2.39 + 0.0138*N	88	2.28 + 0.0072*N
		_0.24 + 0.0012*N_		_0.29 + 0.0011*N_		_0.255 + 0.00077*N_
UK	101	2.23 + 0.0074*N	81	1.79 + 0.0082*N	53	1.13 + 0.0097*N
		_0.18 + 0.00088*N_		_0.12 + 0.0010*N_		_0.0095 + 0.0013*N_
Belgium/ Netherlands	18	7.85 + 0.0018*N	16	1.97 + 0.012*N	9	1.43 + 0.011*N
		_0.028 + 0.0015*N_		_0.032 + 0.0020*N_		0_.0013 + 0.0019*N_
Scandinavia	31	-0.6 + 0.057*N	19	1.63 – 0.0024*N	12	1.34 – 0.0004*N
		_-0.001 + 0.0026*N_		_0.19 – 0.0010*N_		_0.12 – 0.00043*N_
France/Spain /Italy	25	0.6 + 0.011*N	17	0.22 + 0.011*N	12	0.088 + 0.0103*N
		_-0.32 + 0.0031*N_		_-0.63 + 0.00375*N_		_-0.80 + 0.0043*N_
N. America	188	1.43 + 0.0285*N	95	0.19 + 0.028*N	35	1.19 + 0.0072*N
		_0.38 + 0.00164*N_		_0.32 + 0.00160*N_		_0.29 + 0.0012*N_
Tropics	90	0.8 + 0.014*N	31	1.35 + 0.015*N	17	1.52 + 0.015*N
		_0.31 + 0.00127*N_		_0.34 + 0.0013*N_		_0.37 + 0.00115*N_
Australia/ New Zealand	14	-8.3 + 0.081*N	6	3.25 - 0.0022 *N	6	3.25 - 0.0022 *N
		_0.25 + 0.0022*N_		_0.48 + 0.00016*N_		_0.48 + 0.00016*N_

For most countries and regions, the number of observations lasting for >300 days were too few to make a separate analysis meaningful, with Germany and the UK as exceptions. When including observations lasting for >50 and >150 days, emissions for the larger regions increased in the order Europe < The Tropics < North America, but when turning to observations of >300 days, the sequence changed for the linear regression of emission data, but not for the log-transformed regression. When looking at only Germany and the UK, the sequence changed for both the linear and log-transformed regressions, i.e., the sequence when including observations >50 and > 150 days was UK < Germany, but when switching to observations >300 days the sequence became Germany < UK.

Thus, both global and country-specific emission factors changed depending on which data were included in the analysis. However, in nearly all instances the emission factors were lower than the currently used IPCC factor, and centered around 0.8%. Without exception, the degree of explanation was low. Due to climatic variations, regional emission models should be developed, but with the current low and varying amount of data available and with emission estimates based only upon N inputs, the justification for introducing country-specific emission factors is questionable. Furthermore, due to the dominance of observations from Northern Europe, the global emission factor mainly reflects this region with its particular climatic conditions, e.g., cold winters. Including more data from regions such as Asia might change the global emission factor.

Use of cover crops and crop residue incorporation are common practices in Europe, and emissions from these N sources will be reflected in the total annual emission. The new IPCC guidelines (IPCC, 2000) include emission factors also for incorporated crop residues and green manures. This is commendable, but requires adjustment of the emission factors to fit realities in the field. Unfortunately, in most of the available studies it is difficult to distinguish between the emissions from the various N sources. Including emission factors for these N sources based on the present emission factors may thus result in double accounting, since these are already included in the overall annual emission factor for fertilizers.

Other factors of importance for N_2O emission

Visual inspection of the points above the trend line of the various data sets indicated that the following operations tend to be associated with enhanced emissions of N_2O:

- injection of anhydrous ammonia;
- application of organic or mixed organic/mineral fertilizers;
- growing of vegetables - mainly potatoes - irrespective of fertilizer type;
- ploughing-in of crop residues and green manure;
- fertilizers applied to poorly drained grassland in Scotland.

However, these trends were not distinct. When analyzing the entire dataset 2 in more detail (Lægreid & Aastveit, 2002), the following factors were found to have a statistically significant influence on N_2O emissions in addition to N application rate:

- fertilizer type: organic plus mixed organic/mineral fertilizers emitted more N_2O than pure mineral fertilizers, but there were no significant difference between the various mineral fertilizer types;
- crop type: generally the overall emission increased in the order grass < cereals < vegetables, but the N fertilizer induced emissions for these various crops was not significantly different;
- soil organic carbon: there is a trend of increasing N_2O emission with increasing soil organic carbon;
- drainage: poorly drained soils emit more N_2O than well drained soils;
- pH: there is a trend of increasing emissions with increasing pH.

Only soil organic carbon was found to interact with N application rate, i.e., the fertilizer-induced emission was higher from soils rich in carbon than from soils with a low carbon content.

Further breakdown of the data into sub-groups based on country or region, duration of observation, crop or mineral fertilizer type etc. gave no improved explanation.

Important drivers for N_2O emissions are high soil moisture contents, freezing/thawing and drying/wetting cycles, and substrate availability (e.g., organic matter and nitrogen), together defining soil and climatic conditions (Flessa et al., 1995; Clayton et al., 1997; Dobbie et al., 1999). The few measurements spanning over two years or more showed that differences in emissions between management or fertilization practices within one year were smaller then the differences between identical treatments from one year to the next (Clayton et al., 1997; Kaiser et al., 1998a,b), as illustrated by Table 2.

Table 2: N_2O emission from grassland in Scotland at various fertilizer types and years (Clayton et al, 1997).

Year	N rate		AS	Urea	CN	AN	Slurry	Control	
1992	360	Emission:	0.7	3.0	1.6	1.5	(0.5)*	0.04	kg N/ha,y
1993	360	Emission:	1.3	5.2	4.0	4.2	6.4	0.3	kg N/ha,y

*no measurements immediately after first application;

AS = ammonium sulphate; CN = calcium nitrate; AN = ammonium nitrate

In the example above, winter emissions were low. Differences in weather conditions in the periods after fertilizer applications probably caused the interannual variability illustrated in Table 2. Further, as shown in several German studies, N_2O

emissions during winter can range from 7 to 89% of annual emissions (Flessa et al., 1995; Röver et al., 1998; Kaiser et al., 1998b; Kaiser & Ruser, 2000). We thus hypothesize that differences in emissions between management practices can be overshadowed by differences in emissions caused by climatic variations.

Crop type and yield, rotation and previous farming practices also influence N_2O emissions. Kaiser & Ruser (2000) found better correlation between average N_2O emissions during a crop rotation and average soil N input/output balances during the same rotation than with N application rate when comparing six sites in Germany. However, there are few data covering an entire rotation, and no such correlation was found when looking at the larger set of individual annual emission rates and N input/output balances.

The influence of previous cropping history was illustrated by Mogge et al. (1999), who found that a field kept for 30 years under a crop rotation with manure application emitted more than twice as much N_2O as a field farmed to maize monoculture for 20 years. Both fields had the same texture and organic matter content, they were both planted to maize during the observation period, and yields were similar. Differences in microbial biomass seemed to be the main reason.

Ploughing in of cover crops and crop residues increases N_2O emissions in inverse relation to the C/N ratio of the plant material (Baggs et al., 2000; Larsson et al., 1998).

Hence, climatic conditions, crop types and rotation practice as well as previous cropping history are all factors that need to be considered when evaluating N_2O emissions from agricultural land.

N_2O emission and food production

Today, N_2O emissions are reported per unit of land area, and the only way to reduce N_2O emission with reference to the IPCC guidelines is to reduce N input. However, fertilizer input is a prerequisite for food production. Projections of population growth indicate the need for an increased input of fertilizers. Nitrous oxide emissions cannot be avoided, hence an optimal use of fertilizers to provide the necessary food for the world population with a minimum loss of N to the environment should be the ultimate goal. Information on N_2O emission per unit yield is therefore of more interest than the present information related to land area. In most publications, yield information is lacking, but it was included by Kaiser & Ruser (2000). Using their data, it is clear that the N_2O emissions per unit yield did not differ much between N applied at optimum, at half-optimum, or at zero N input where atmospheric deposition and mineralization of soil organic matter were the only sources of N (Fig. 3). The two high emission data points in square represent oilseed rape from 1996 when yields were very low; yields for the zero N plot of the same data series were nearly absent and thus not included in this comparison.

Henault et al. (1998) found that over-fertilization increased N_2O emissions compared to fertilization at optimum N rate. Hence, fertilization at optimum N rates with regard to crop yield seems to be the best practice for food production with least N_2O emissions. Overfertilization may increase N_2O emissions, while fertilization below yield optimum did not seem to reduce N_2O emissions per unit of food produced.

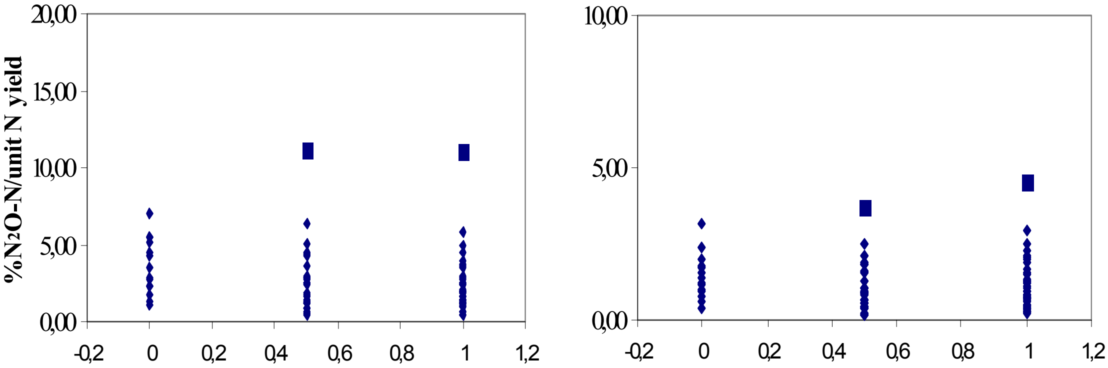

Figure 3. Percentage of N emitted as N_2O per unit N yield for annual emissions (left) and summer emissions (right). Source: Kaiser & Ruser (2000). The two points with notably high emissions (marked in square) represent oilseed rape from 1996, the year with highest overall emissions.

Different regions of the world, such as hot dry and cold temperate regions, and different farming systems, e.g., irrigated versus non-irrigated land, and extensive versus intensive farming, have different productivity. Their overall N_2O emission rates will most likely also vary. How the land is managed will probably also influence N_2O emissions, but this is difficult to evaluate based on the data available today, particularly with regard to emissions per unit of product output.

Conclusion

Fertilizer N input explains only a fraction of the variation in N_2O emitted from agricultural fields. Hence, a better model to estimate N_2O emissions and improve the IPCC model, as well as to identify good management practices that improve crop N utilization and reduce all N losses including N_2O emission, is desirable.

Statistical tools and modelling are becoming increasingly powerful, but this cannot substitute missing data. Many of the published studies available lacked relevant information on factors that might influence N_2O emissions, or the information was given in a form which was difficult to extract and make use of in worksheet form. We therefore recommend that the following data should be reported in future studies:

- split observations between high/low emission periods, and reporting of all relevant information for each period in an extractable form;
- soil parameters: include data for water filled pore space, the mineral N content in the soil, previous field use;
- crop: species and observed yields;
- climate: precipitation and temperature;
- management: type and method of fertilizer application, number and timing of splits, calculated optimal application and yield, as well as soil preparation, irrigation schedules, use of cover crops, crop residue incorporation.

Preferably observations should last for at least two years; even better data would be obtained if they lasted throughout a whole rotation. Additional improvements could be achieved if experiments were coordinated across field and country borders. EU programmes such as the MIDAIR (EVK2-2000-22045) on biogenic greenhouse gas emissions from dairy farming, which includes N_2O monitoring from arable soils in several countries using identical methodology, could be the way forward. This would make it easier to identify the most important drivers and give recommendations for farming practices that may contribute to reduce N_2O emissions. Vegetables might be hot-spots for N_2O emissions, but on a global basis grassland and cereals are the dominant crops. To reduce the number of factors involved, we therefore recommend to concentrate on grass and cereals.

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Nitrous oxide emissions at low temperatures

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Summary

Microbial processes in soil are generally stimulated by temperature, but at low temperatures there are anomalies in the response of microbial activities. Soil physical-chemical characteristics allow existence of unfrozen water in soil also at temperatures below zero. Therefore, some microbial activities, including those responsible for nitrous oxide (N_2O) production, can take place even in "frozen" soil. Nitrous oxide emissions during winter are important even in boreal regions where they can account for more than 50% of the annual emissions. Snowpack therefore has great importance for N_2O emissions, as it insulates soil from the air allowing higher temperatures in soil than in air, and possible changes in snow cover as a result of global warming would thus affect the N_2O emission from northern soils. Freezing-thawing cycles highly enhance N_2O emissions from soil, probably because microbial nutrients, released from disturbed soil aggregates and lysed microbial cells, support microbial N_2O production. However, the overall interactions between soil physics, chemistry, microbiology and N_2O production at low temperatures, including effects of freezing-thawing cycles, are still poorly known.

Introduction

Microbial processes in soil are controlled by a network of physical, chemical and biological factors. Temperature is the key parameter driving microbial activity in soil. Among microbes there is a high diversity in their reaction to temperature. However, most microbial activities in soil decrease with decreasing temperature. The effect of temperature on soil and its biota is more complex at low temperature, especially when the temperature fluctuates between minus and plus degrees.

There is an urgent need to reduce emissions of greenhouse gases associated with various anthropogenic activities. Soils, especially the agricultural ones, are globally the main source of nitrous oxide (N_2O), an efficient greenhouse gas which also participates in reactions destroying ozone in the stratosphere (Kroeze et al., 1999). To control N_2O emissions, knowledge about soil processes as affected by land-use practices is required. Recently, the high capacity of soils to produce N_2O even at low temperatures has received attention (e.g. Flessa et al., 1998; Groffman et al., 2001; Kaiser et al., 1998a,b; Röver et al., 1998; Teepe et al., 2001; Papen & Buterbach-Bahl, 1999; Prieme & Christensen, 2001). Feedback mechanisms caused by global warming could also highly affect N_2O emissions from soils in the future. In northern regions, the soil temperature remains at a low level for several months and for N_2O emissions the change in the temperature during the "cold" seasons may therefore be even more important than the change in mean annual temperature.

The N_2O production at low temperatures cannot be understood by considering only the physiological capacity of soil microbes to be active at various temperatures. At low temperatures, including sub-zero ones, soil physics and chemistry highly affect soil microbiology. This paper shortly discusses the physical, chemical and biological aspects of N_2O production at low temperatures.

Biology behind the N_2O production

Nitrification and denitrification are the main microbial processes involved in N_2O production. In chemolithotrophic nitrification, ammonium (NH_4^+) is oxidized first to nitrite (NO_2^-) and further to nitrate (NO_3^-) by bacteria, which gain energy from the oxidation of these inorganic compounds and utilize carbon from carbon dioxide for cell growth. The bacterial species in chemolithotrophic nitrification oxidize either ammonium or nitrite – the same organism cannot oxidize both. Some N_2O is produced in connection with ammonium oxidation, especially at low oxygen concentrations (Poth & Focht, 1985). In denitrification, denitrifying bacteria in their respiration replace oxygen with nitrate, if oxygen is depleted. Most of them need organic substrates as energy source and for the synthesis of cell constituents. Denitrification is a stepwise process, where nitrate is reduced via nitrite, nitric oxide (NO) and nitrous oxide (N_2O) to nitrogen gas (N_2). Some denitrifiers are capable of starting the reduction with nitrite. Also, the gaseous species N_2O and NO can be taken up from the environment and then be further reduced. Denitrification is closely linked to nitrification via nitrate/nitrite production. Nitrous oxide production can thus be favoured by increasing availability of ammonium and/or nitrate.

In soils, ammonium can originate from external sources (fertilization, deposition) or from mineralization of nitrogenous organic matter. Without external nitrate, some oxygen is always needed to generate nitrate/nitrite. In addition to nitrate, organic substrates are essential for denitrification. Therefore, also vegetation has a connection to denitrification, because plants take up inorganic nitrogen and release organic substrates in their above- and below-ground litter production. Also, root exudates are an important source of easily decomposable organic substances to the soil. Denitrification in soil is partly regulated by the general microbial activity which regulates the oxygen status of the soil via oxygen consumption (Fig. 1).

There are factors which affect not only the overall denitrification rate, but also the relative activity of the key enzymes in denitrification (Fig. 1). The activity of nitrous oxide reductase, the enzyme reducing N_2O to N_2, decreases under some conditions, thereby increasing the ratio of N_2O to N_2 in the gaseous end products. Therefore, the emission of N_2O can increase even without any increase in total denitrification (the sum of N_2O and N_2). An increase in oxygen availability is a basic factor increasing the N_2O:N_2 ratio (Firestone & Davidson, 1989). Similarly, high concentrations of nitrate in soil cause an imbalance in the activity of nitrous oxide reductase and the preceding reductases which increases the ratio of N_2O to N_2 from denitrification (Blackmer & Bremner, 1978; Cho & Sakdinan, 1978; Nömmik et al., 1984). Low pH and low temperature, typical characteristics of boreal soils, are among the factors retarding the activity of nitrous oxide reductase (Focht & Verstraete, 1977; Firestone & Davidson, 1989). In accordance with this, N_2O is often the main end product of denitrification in boreal acidic soils (Regina et al., 1996, Maljanen et al., unpublished).

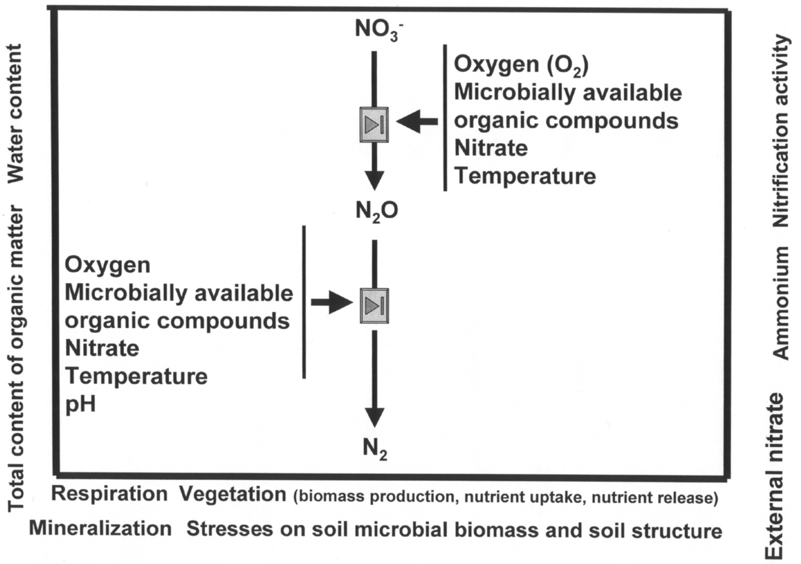

Figure 1. Relationships between soil physical, chemical and biological characteristics N_2O production via denitrification.

Soil as an environment allowing activity at low temperature

When considering N_2O production in northern soils, periods with a temperature below zero cannot be neglected. Biological activity requires the presence of water, but the soil is an environment that allows the presence of unfrozen water below 0°C, During freezing, dissolved inorganic and organic compounds will concentrate in an unfrozen film around the soil inorganic/organic matrix. The proportion of unfrozen water drops rapidly as the soil temperature decreases from 0°C to -1 or -2°C. However, after this initial sharp decrease the content of unfrozen water remains almost constant with further temperature decrease (Patterson & Smith, 1981). As stated above, unfrozen water exists as a film around the soil matrix (Fig. 2), and clay soils with a high specific surface therefore have a greater capacity to retain unfrozen water than more coarse-textured soils. The nutritional conditions for microbes are good in the unfrozen film, because inorganic and organic solutes are excluded from the freezing water. Microbial oxygen consumption in an unfrozen film surrounded by ice can be expected to create oxygen deficiency and thereby stimulate denitrification (Fig. 2).

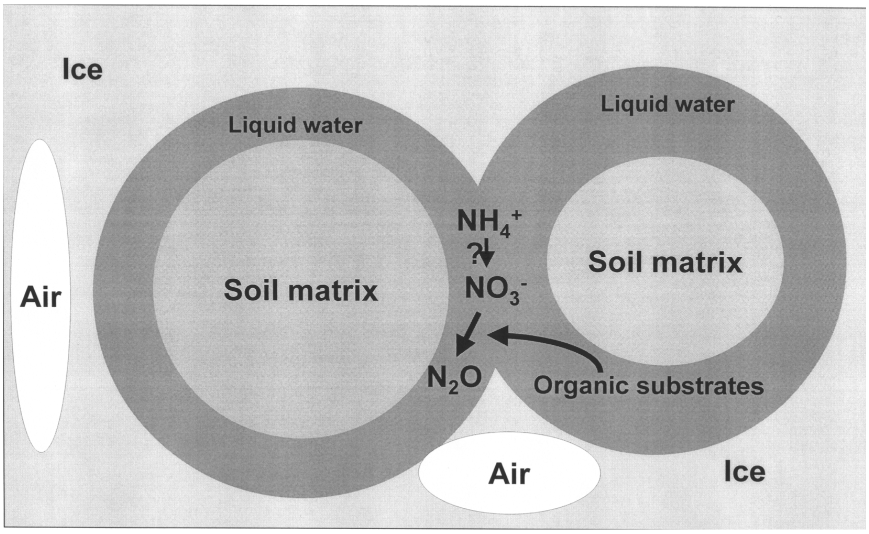

Figure 2. A conceptual model for N_2O production in soils at temperatures below 0°C. When soil temperature falls below zero there is a liquid water film around the soil matrix. Inorganic and organic microbial nutrients are transported from the freezing water into the liquid film. The high nutrient concentrations in the film would favour microbial processes like nitrification and denitrification. Low oxygen availability in water films surrounded by ice would enhance denitrification and N_2O production (modified from Teepe et al., 2001).

Microbial enzymatic activities in the "physiological" temperature range theoretically follow the temperature dependency of chemical reactions described by the Arrhenius equation. A temperature increase of 10°C, Q_{10}, is known to increase, e.g., respiration on average 2.4 fold (Raich & Schlesinger, 1992). Q_{10} values for denitrification have ranged from 1.5 to 3.0 at temperatures between 10 and 35°C (Knowles, 1982). However, at low soil temperatures (that is, a few degrees below and above zero) much higher Q_{10} values for microbial respiration have been reported, even higher than 10 (Clein & Schimel, 1995). It is probable that these high Q_{10} values do not describe solely the effect of temperature on microbial respiration, but reflect also an effect on the availability of substrates at low temperatures (Clein & Schimel, 1995). Relative microbial activities thus react much stronger to small temperature changes at low temperatures than at higher temperatures. This would have importance also for denitrification at temperatures close to zero (see below).

Freezing and thawing of soil

It has been clearly shown that microbial activities (Rivkina et al., 2000), including denitrification (Teepe et al., 2001), take place at soil temperatures below zero. There is evidence that N_2O can be produced in boreal agricultural soils at least down to –6°C (Koponen et al., unpublished). However, the highest N_2O emissions generally occur during periods when the soil temperature fluctuates between minus and plus degrees (Prieme & Christensen, 2001; Teepe et al., 2000; 2001). Several reasons for the high microbial activity in thawing soil can be given. It has been known for decades that freezing-thawing or drying-wetting disturb soil aggregate structure, which increases the availability of inorganic and organic substances supporting activity of soil microbes (Soulides & Allison, 1961; Denel et al., 2001; Prieme & Christensen, 2001). In denitrification, the availability of energy is among the key factors regulating the process (Fig. 1). Therefore, it is not surprising that soil freezing-thawing as well as drying-wetting increases the N_2O evolution from soil (Prieme & Christensen, 2001; Teepe et al., 2001). Both these "stress factors" can evidently also destroy microbial cells in the soil, and part of the substrate available probably originates from lysed cells. Also, the microbes contributing to the increase in the N_2O production after soil thawing or wetting are able to respond rapidly to the elevated substrate availability. A portion of the N_2O emitted during thawing can originate from the release of N_2O produced in unfrozen water films, but trapped in the soil by the surrounding ice (Teepe et al., 2001; Koponen et al., unpublished).

Chemodenitrification would also participate in the N_2O production in frozen soil. Nitrite is known to react with humic substances producing N_2O (Stevenson, 1982). Nitrite from nitrification or nitrate reduction would concentrate in the unfrozen water film together with the other solutes (see above) and may react there with organic matter.

Nitrous oxide emissions during winter

Winter emissions of N_2O can contribute significantly to annual emissions, also in regions with sub-zero temperatures during winter (Table 1). The previous sections have shown that there is a potential for N_2O production in unfrozen films around soil particles at sub-zero soil temperatures. Also, the soil temperature during winter can be considerably higher than the air temperature if there is a thick snowpack insulating the soil from the air (Papen & Butterbach-Bahl, 1999). There are results showing that N_2O emissions during winter are highly regulated by the thickness and timing of snowpack, and by the length of the period with snow cover (Brooks & Schmidt, 1997).

Table 1. Emissions of N_2O during cold seasons as a percentage of total annual emissions from some soils located in boreal/temperate regions.

Site location	Site type	Cold season emissions (% of the annual)	Reference
Central Germany	Agricultural	58	Teepe et al. (2000)
	Fallow	45	
	Forest	50	
Lower Saxony, Gemany	Agricultural (silty loam)	70	Röver et al. (1998)
Lower Saxony, Germany	Agricultural (luvisol)	50	Kaiser et al. (1998a)
Lower Saxony, Germany	Agricultural (sandy loam)	47	Kaiser et al. (1998b)
USA, Rocky Mountains	Alpine tundra	> 50	Brooks & Schmidt (1997)
Eastern Finland	Organic agricultural	30-60	Maljanen et al. (unpubl.)
	Forested organic soil	60	
Eastern Finland	Organic agricultural	38	Alm et al. (1999)
	Forested fen	28	
Eastern Finland	Forested fen	37-52	Regina et al. (1998)

Global warming and the N_2O emissions

Global warming is likely to increase N_2O emissions by enhancing both organic matter mineralization, nitrification and denitrification. However, if the extent of snowpack covering the soil is also reduced as a result of warm winters, the change in N_2O emissions during winter is difficult to predict. If the insulating effect of the snowpack is reduced, the soil temperature would be lower during winter, decreasing microbial activities and N_2O production. On the other hand, the possibly more frequent freezing-thawing cycles during winter would increase N_2O emissions. In a changing climate, mild freeze events will probably be common. The effects of mild freeze on soil nutrient dynamics and denitrification seem to vary, at least in forest ecosystems (Groffman et al., 2001).

Low-temperature N_2O emissions and IPCC quidelines

The N_2O emissions at low temperatures have to be known for the accurate regional N_2O inventory. The short-term periods with high N_2O emissions associated with the sub-zero temperature fluctuations are a special challenge to the inventory methodology. There can be a great annual variation in the cold-season emissions depending, e.g., on the numbers of freezing-thawing cycles. As mentioned above the freezing-thawing cycles and their high N_2O emissions would increase with global warming which obviously affects the N_2O inventory.

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Nitrous oxide emissions derived from N leaching[5]

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Summary

Much of the fertilizer and manure nitrogen (N) that is applied to crop fields leaves the field in runoff and leaching to groundwater. This N is transformed as it moves across the landscape through riparian zones, rivers and estuaries, and nitrous oxide (N_2O) is produced along the way. In this paper, we 1) discuss the mechanisms that lead to these "indirect" N_2O emissions, 2) describe the Intergovernmental Program on Climate Change (IPCC) methodology for assessing these emissions, 3) review the data in support of the methodology, 4) discuss implications for mitigation and 5) summarize with a case study for the nation of Denmark.

Introduction

Nitrous oxide (N_2O) is a "greenhouse" gas that influences the radiative budget of the earth and contributes to stratospheric ozone destruction (Mooney et al., 1987; Prather et al., 1995). The concentration of N_2O in the atmosphere is increasing at a rate of 0.2 - 0.3% a year and is responsible for approximately 5% of the global enhanced greenhouse effect (Prather et al., 1995).

Under the terms of the United Nations Framework Convention on Climate Change (UNFCC) and the Kyoto protocol, each nation is required to compile national emission inventories for radiatively active trace gases (CO_2, CH_4, N_2O). The Intergovernmental Program on Climate Change (IPCC) has developed protocols for quantifying N_2O emissions from industry, agriculture and natural ecosystems (IPCC, 1997). The protocols for N_2O emissions from agriculture consider "direct" emissions from fertilized/manured crop fields as well as "indirect emissions." The indirect emission calculations attempt to account for N_2O production associated with transformations of the significant amount of the fertilizer nitrogen (N) that leaves crop fields in harvest, leaching and runoff or is transferred to the atmosphere (Mosier et al., 1998).

In this paper, we discuss the mechanisms and rates of N_2O emission associated with the N that leaves crop fields in leaching and runoff. The IPCC methodology for calculating indirect emissions is based on the idea that small amounts of N_2O are produced as agriculturally derived N moves through the landscape from fields

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[5] Much of the text in this paper is taken from Groffman et al. (2002).

to groundwater to streams to estuaries to the ocean. The validity of the methodology is hindered by the fact that the amounts of N moving across the landscape, and the emissions along the way, are poorly quantified. There is great interest in evaluating and improving the methodology because indirect emissions represent a significant fraction of the agricultural N_2O source. Moreover, there is potential for mitigation of indirect emissions because there are active efforts to control the movement of N across the landscape for water quality protection. Methods for controlling N movement could possibly be adapted to reduce N_2O emissions. In the sections that follow, we 1) discuss the mechanisms that lead to indirect N_2O emissions, 2) describe the IPCC methodology for assessing these emissions, 3) review the data in support of the methodology, 4) discuss implications for mitigation and 5) summarize with a case study for the nation of Denmark.

Mechanisms

It has long been known that a significant portion of the fertilizer and manure N that is applied to crop fields leaves the field in leaching and runoff. A general assumption is that roughly 50% of N applied is removed in harvest (Keeney & Follett, 1991). Given that most agricultural soils are not accumulating organic matter (Paul & Clark, 1996), the other 50% of applied N leaves by either hydrologic or gaseous pathways. In regions where precipitation exceeds evaporation, especially during the non-growing season, the dominant vector of N loss is leaching of nitrate (NO_3^-). Nitrate is a drinking water pollutant and a prime cause of eutrophication in marine waters (Keeney, 1986; Diaz, 2001).

Once NO_3^- leaves crop fields it passes through the vadose (unsaturated) zone of the soil profile and into groundwater. While the potential for biological processing of NO_3^- in the subsurface is often thought to be low, many studies have found biological activity, including N_2O production in groundwater (Groffman et al., 1998). Groundwater-borne NO_3^- moves towards streams and is subject to processing in the near-stream (riparian) zone (Fig. 1). In riparian zones, groundwater often approaches the soil surface where the potential for biological activity is much higher than in deeper aquifers. Once NO_3^- moves into streams, lakes, estuaries and oceans, there is potential for biological processing in both the water column and sediments of these aquatic ecosystems.

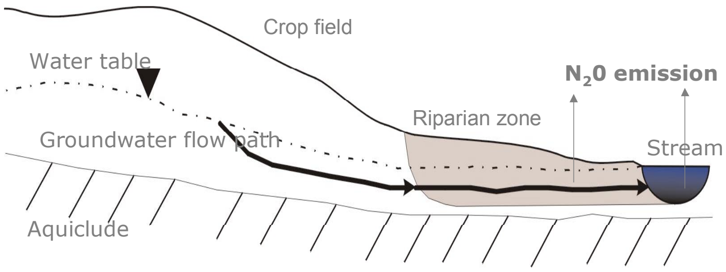

Figure 1. Nitrogen flows through the landscape lead to indirect N_2O emissions from agriculture.

The dominant biological processes leading to N_2O production are denitrification and nitrification. Denitrification refers to the primarily anaerobic reduction of NO_3^- to nitrite (NO_2^-) and the N gases nitric oxide (NO), N_2O and dinitrogen (N_2). The yield of different gases is highly variable and is controlled by several environmental factors (e.g., oxygen, pH). Most of the denitrifying bacteria that have been studied are heterotrophic (use carbon as a source of energy), however there are some denitrifiers capable of deriving energy from the oxidation of inorganic compounds, e.g. pyrite (Hiscock et al., 1991). Denitrification is expected to be vigorous in wet, high C wetland soils and in the anaerobic layers of aquatic sediments.

Nitrification refers to the oxidation of NH_4^+ to NO_2^- and NO_3^- by a specialized group of chemoautotrophic bacteria that derive energy from these oxidations. N_2O is produced as a by-product of the oxidation of NH_4^+ (Davidson et al., 2001). The process is considered to be aerobic, but has been observed to occur under microaerophilic (low oxygen) and anaerobic conditions (Firestone & Davidson, 1989). Nitrification is vigorous in upland soils, the water column of lakes, streams and estuaries and in aerobic layers of aquatic sediments.

IPCC methodology for calculating indirect N_2O emissions from agriculture

The IPCC methodology for calculation of national emission inventories for agriculturally derived N_2O includes both direct and indirect emissions. Direct emissions from fertilized fields are assumed to be 1.25% of fertilizer and manure N applied to the field. The calculation for indirect N_2O emissions is:

N_2O(Indirect) = N_2O(G) + N_2O(L) + N_2O(S) where:

N_2O(G) = emissions associated with atmospheric deposition of agricultural N that has been transferred to the atmosphere.

N_2O(L) = emissions associated with the N that leaves crop fields in leaching and runoff.

N_2O(S) = emissions associated with human sewage.

N_2O(L) = NLEACH * EF5 where:

NLEACH = the amount of N that leaves crop fields in leaching and runoff. This is assumed to be 30% of the fertilizer and manure N that is applied to crop fields.

EF5 = N_2O emission factor for N that leaves crop fields in leaching and runoff and is processed as it moves ultimately to the world ocean. This factor is assumed to be 2.5% and is partitioned as EF5-g (groundwater, 1.5%), EF5-r (rivers, 0.75%) and EF5-e (estuaries, 0.25%).

The indirect emissions represent 1/3 of total agricultural emissions and are dominated (75%) by those associated with leaching and runoff, which in turn is dominated (60%) by the emissions from groundwater. It is interesting to note that EF5 is the highest emission factor in the inventory methodology, higher even than the emission factor for direct emissions from fertilized fields, suggesting that leached N is even more likely to lead to N_2O emissions than fertilized N applied directly to surface soil. The uncertainty associated with indirect emissions is large and the dataset supporting EF5 is small (Nevison, 2000).

The original formulation of EF5-g was based on the idea that some of the N_2O produced in surface soils is transported to groundwater with leaching water and eventually degasses to the atmosphere. The value of EF5 was derived from a small number of studies that reported $N_2O:NO_3^-$ ratios in agricultural drainage water. This formulation is problematic because it assumes that there is no biological processing of N and N_2O production between surface soils and streams (Groffman et al., 2000). Numerous studies of the vadose zone, groundwater and riparian zones have found active N processing and significant N_2O production along the pathway from fields to streams (Hill, 1996; Lowrance, 1998; Groffman et al., 1998; Groffman et al., 2000). Degassing of surface-produced N_2O is likely important in areas with artificial drainage that greatly increases the speed, and reduces the biological processing, of water and N movement from fields to streams (Hack & Kaupenjohann, 2002). Nevison (2000) reviewed the literature on groundwater degassing and suggested that the original value for EF5-g (0.015) was overestimated and should be reduced, possibly to as low as 0.001.

Values for EF5-r and EF5-e were based on the idea that N processing and N_2O production in rivers and estuaries are a function of NO_3^- inputs to these water bodies (Seitzinger & Kroeze, 1998). The database in support of these factors is small. Moreover, factors such as water depth and residence time may be more important controllers of NO_3^- processing and N_2O production in rivers than NO_3^- inputs (Cole & Caraco, 2001).

Data in support of the IPCC methodology

There are very few data available to validate the IPCC emission factors for N_2O emissions associated with leaching and runoff. This lack of data is in marked contrast to the emission factor for direct emissions, which is based on several hundred field studies (Bouwman, 1996; Lægreid, 2002). True validation of indirect emissions requires a combination of data on N flows across the landscape with measurement of N_2O concentrations and fluxes. While hydrologic-based analyses of N flows in agricultural watersheds are relatively common, N_2O data are seldom collected in these studies. Evaluation of the methodology for indirect emissions is also complicated by the fact that these emissions are a spatially explicit phenomenon, involving the interaction of specific parcels of water with specific landscape features with different potential for N processing and N_2O production. It is difficult to incorporate spatially explicit phenomena into a methodology that is driven solely by the amount of N added.

Weller et al. (1994) presented data on N flows and N_2O emission from a small watershed in Maryland, USA with maize cropping in the upland and riparian forests at the interface between the fields and stream. N_2O production in the riparian forest was equal to 0.0065 kg N_2O per kg of NO_3^- input into the riparian forest, which is less than the IPCC value for EF5-g of 0.015.

Gold et al. (2002) produced estimates of riparian N_2O emission for an 850-km^2 watershed in Rhode Island that attempted to account for the spatially explicit nature of indirect emissions. Their analysis considered variation in the amount of N that different riparian zones process by denitrification as well as variation in N_2O production during this denitrification (the $N_2O:N_2$ ratio). They accounted for variation in riparian denitrification using field data on riparian characteristics known to influence the ability of these areas to intercept and denitrify upland-derived NO_3^- (Rosenblatt et al., 2001). These characteristics were linked to soil characteristics (parent material, drainage class) that are included in new soils databases (SSURGO) available for many states in the U.S. (Soil Survey Staff, 1997). Variation in $N_2O:N_2$ ratios was assessed with ^{15}N-based field measurements at four riparian sites in the watershed. For the entire watershed, 0.014 kg N_2O were produced per kg of NO_3^- leached, a value very close to the IPCC value for EF5-g of 0.015.

Cole & Caraco (2001) assembled data on N_2O emissions from rivers and evaluated the assumption that these emissions are driven by NO_3^- inputs. They were able to assemble data from seven rivers and compared measured emissions with those derived from an emission-factor type model driven by NO_3^- input (Seitzinger & Kroeze, 1998). The model tended to over-predict N_2O emissions, suggesting that river physical and/or biological characteristics may also need to be considered, along with NO_3^- inputs, as a driver of emissions.

While there have been no systematic evaluations of indirect N_2O emissions in the Nordic countries, several studies have quantified landscape N flows and associated N_2O emissions. Ambus & Christensen (1995) measured N_2O emissions from riparian zones but found no clear relationships between emissions and N inputs, and high variability that could not readily be explained by environmental factors. Paludan & Blicher-Mathiesen (1996), and Blicher-Mathiesen & Hoffmann (1999) quantified NO_3^- absorption and N_2O dynamics in a riparian fen in Denmark and found that the N_2O yield varied with hydrologic flowpath, i.e., if there is a long anaerobic flowpath through the wetland, denitrification acts as a strong sink for N_2O. Nitrogen retention by Nordic lakes and rivers has also been shown to be dependent on hydrologic conditions, with retention increasing along with residence time in both lakes (Windolf et al., 1996) and rivers (Svendsen & Kronvang, 1993).

There is a clear need for more data to evaluate the components of EF5. While the two riparian studies described above suggest that the values for EF5-g may be reasonable, the degassing-based formulation of this factor needs to be revised to include microbial processing of leached N, and many more spatially explicit evaluations are needed to increase confidence in its validity. The analysis of N_2O emissions from rivers suggests that EF5-r may be an overestimate, but again, many more measurements are needed. More fundamentally, we need to consider if we need to make the methodology for computing indirect emissions spatially explicit to at least some degree. It is clear that there is great variation in the ability of groundwater, riparian zones, wetlands, rivers and estuaries to process agriculturally derived NO_3^- and produce N_2O. Incorporating this variation into the methodology may be critical for reducing the uncertainty associated with indirect emissions.

One efficient route to improving the database underlying the indirect emission calculations is to add N_2O measurements to existing hydrology-based studies of N flows in landscapes. For example, Steinheimer et al. (1998) presented data from

23 years of intensive monitoring of a 40 ha maize-dominated watershed in Iowa, USA. They determined that approximately 50% of the N applied in fertilizer left the field in harvest, 17% left in leaching and runoff and roughly 30% was unaccounted for. If we assume that most of this unaccounted for N was lost as gas (assuming that the soils are not accumulating N), this provides an upper limit on the amount of N_2O that could be produced. However, our ability to assess the actual amounts of N_2O emitted is limited by lack of knowledge of the N_2O:N_2 ratio during gaseous loss. Given that this ratio can range from 1:50 to 99:1, estimates of N_2O emission from this watershed could range from 0.6 to 30% of the N applied to the field (Groffman et al., 2000). The IPCC methodology predicts that 1.7% of the N applied would be emitted as N_2O in direct (1.25%) and indirect emissions (0.45%) from groundwater (1.5% of leached N, which is 30% of the fertilizer applied, i.e. 0.3.*1.5 = 0.45). If N_2O concentrations and fluxes had been measured along with the N flows in the Steinheimer et al. (1998) study, we would have a powerful evaluation of the IPCC methodology. Given that there are numerous national and regional programs to monitor N flows, at a wide range of scales, adding analysis of N_2O to these programs would be an efficient mechanism for improving the scientific basis for EF5.

Implications and opportunities for mitigation

Consideration of indirect emissions of N_2O from agriculture readily leads to consideration of mitigation. There is great interest in managing N flows in the environment for water quality reasons. There is particular concern about NO_3^-, which is a drinking water pollutant and an agent of eutrophication in coastal waters (Keeney, 1986; Diaz, 2001). Coastal eutrophication is a truly global problem and is most frequently linked to agricultural N use (Diaz, 2001). Approaches to controlling eutrophication include reducing fertilizer use, increasing the efficiency of fertilizer use, i.e. reducing leaching loss, and establishing "sinks" for N in watersheds by managing riparian areas, wetlands and streams (Mitsch et al., 2001). Each of these approaches has implications for direct and/or indirect emissions of N_2O.

While reductions in fertilizer use to decrease NO_3^- leaching would clearly reduce indirect N_2O emissions (indeed, this is the only route to reduce emissions under the IPCC methodology), other efforts to manage N flows in the landscape have more complex, and possibly contradictory effects on emissions. For example, if NO_3^- leaching is reduced in a field, the current IPCC methodology would produce a reduction in indirect losses because the N_2O production in groundwater, rivers and estuaries is driven by the amount of N leaching (currently fixed at 30%). However, if N stays in the field, e.g., in winter cover "catch crops" instead of leaches, this could lead to an increase in direct emissions (currently fixed at 1.25% of fertilizer and manure input). The IPCC methodology would partially account for this possible increase because it includes an emission from crop residues, and the addition of N in the cover crop residues would be included in this calculation. Still, there is a clear need to evaluate the effect of reducing N leaching on both direct and indirect emissions. In many cases, leaching is less than 30% of the N applied, and there are active efforts to reduce leaching in many areas. The effect of these reductions on N_2O emissions could be significant (Brown et al., 2001; Silgram et al., 2001*)* but needs to be verified with data.

Widespread management of landscape N flows by creating "sinks" could have significant effects on indirect N_2O emissions. This management essentially changes the location of N_2O production, e.g., moving it from estuaries to riparian zones. If the yield of N_2O during denitrification in riparian zones is less than the yield in estuaries, this management would reduce indirect emissions of N_2O. In contrast, managing N flows in the landscape for water quality purposes could increase indirect emissions of N_2O, e.g., if riparian zones emit more N_2O during denitrification than estuaries. The central question is our ability to control N_2O emissions in these systems.

The effect of managing landscape N flows on indirect N_2O emissions depends fundamentally on variation in N_2O:N_2 ratios in different landscape features. We have been investigating environmental controls on this ratio in riparian forests, with an eye towards developing protocols for management of these forests to reduce indirect emissions of N_2O. Unfortunately, this ratio does not exhibit coherent patterns with environmental variables amenable to management in riparian zones. For example, we hypothesized that soil pH, dissolved oxygen and denitrification rate would all be strong controllers of N_2O:N_2 ratio. However, none of these variables were significant predictors of the ratio in our field studies. Several studies have found strong control of this ratio by these variables in the laboratory (Firestone & Davidson, 1989). However, these controls are not readily expressed in the field due to multiple factor interactions and the effect of physical factors, e.g. diffusion, hydrologic flow path, that are important in the field, but not in the laboratory (Blicher-Mathiesen & Hoffmann, 1999). If we cannot control the ratio in landscape features that we are managing as NO_3^- sinks, we will not mitigate indirect N_2O emissions.

Indirect emission scenarios for Denmark

Data from the nation of Denmark are useful for illustrating the nature of indirect emissions and the potential for, and complexities of, options for mitigation of these emissions. An emission inventory for 1998 showed N input to crop fields (fertilizer plus manure) of 653 kt, which produces 196 kt of N leaching using the IPCC default leaching fraction of 30%. This application yields 8.2 kt of direct and 4.9 kt of indirect emissions of N_2O using the IPCC default factors of 1.25% and 2.5% for direct and indirect emissions respectively (Table 1).

These data allow us to explore the effect of different N management schemes on N_2O emissions. The most straightforward approach to N management is to reduce fertilizer input. A 30% reduction in input produces a 30% reduction in leaching losses and both direct and indirect emissions (Table 1, line 2). Reducing input is the only way to mitigate emissions in the current IPCC methodology. However, in some areas, reducing fertilizer input could result in unacceptable reductions in crop production. Moreover, in many areas, leaching is currently less than 30% and/or there are active efforts to reduce leaching to less than 30%. By the current methodology, a 50% reduction in leaching would yield a 50% reduction in indirect emissions and a significant reduction in total emissions (Table 1, line 3). However, the fate of N preserved by reducing leaching is unclear. If this N remains in the crop field, it can be cycled by plants and microorganisms with emission of N_2O. We must accept the possibility that reducing leaching could increase direct emissions from the field, negating some of the benefit from reducing the indirect emissions (Table 1, line 4). Research to determine the effect of leaching reductions on direct and indirect N_2O emissions should be a priority and should be evaluated before the IPCC methodology is altered to allow for reduction of the 30% N leaching default value.

Table 1. Nitrogen input, management and N_2O emission scenarios for Denmark using national input and leaching data for 1998 from Kyllingsbæk et al. (2000).

Scenario	N Input -- kt --	N leached --- kt ---	Direct emissions --- kt -----	Indirect emissions ---- kt ----	Total emissions ---- kt ----
Current	653	196	8.2	4.9	13.1
Reduce fertilizer input by 30%	457	137	5.7	3.4	9.1
Reduce leaching by 50%	653	98	8.2	2.4	10.6
Reduce leaching by 50%, but increase direct emissions by 15%.	653	98	9.4	2.4	11.8
Reduce indirect emissions by 50% by managing riparian zones.	653	196	8.2	2.4	10.6
Increase indirect emissions by 50% by managing riparian zones	653	196	8.2	7.4	15.6

The data from Denmark also allow us to evaluate the possible effects of riparian zone management on indirect N_2O emissions. If we can develop clever ways to manage the N_2O:N_2 ratio during denitrification in these zones, there could be significant reductions in indirect emissions (Table 1, line 5). However, given that we currently do not have a clever way to manage this ratio, it is quite possible that aggressive use of riparian zones to prevent the movement of NO_3^- to coastal waters will increase indirect and total N_2O emissions (Table 1, line 6).

Conclusions

- There is a clear need for more data to evaluate the magnitude of indirect N_2O emissions and to assess the validity of the IPCC methodology for quantifying these emissions. Data on both N flows across the landscape and the N_2O emissions associated with these flows are required. There are currently active programs to assess the movement of water and N across the landscape in many nations. Adding N_2O measurements to these programs would be an efficient way to rapidly reduce the uncertainty in the IPCC methodology for assessing indirect emissions.
- It may be necessary to make the IPCC methodology for assessing indirect N_2O emissions at least partially spatially explicit. These emissions depend fundamentally on the interaction of specific parcels of NO_3^- -laden water interacting with specific landscape features with variable capacity for transforming N and producing N_2O. In areas with well-developed databases on groundwater, geology, soils and hydrology (e.g. SSURGO in the US), it should be possible to markedly improve assessment of indirect N_2O emissions.
- The potential for mitigating indirect emissions is highly uncertain and somewhat controversial. While there is great interest in managing N flows across the landscape for water quality purposes, the effects of this management on N_2O emissions is not clear. There is a strong need to evaluate the effects of water quality-based N management schemes on N_2O emissions (both direct and indirect). There is potential for these schemes to reduce emissions, but only if we can develop techniques to manage the N_2O:N_2 ratio during denitrification.

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