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Biogenic Greenhouse Gas Emissions from Agriculture in Europe – Quantification and Mitigation

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Abbreviations

ARD	Afforestation, reforestation and deforestation
CAP	Common Agricultural Policy
CHP	Combined heat and power plant
CoP	Conference of the Parties of the UNFCCC
EC	European Communities
EEC	European Economic Community
EU	European Union
GATT	General Agreement on Tariffs and Trade
GHG	Greenhouse gas
GPP	Gross primary production
HFC	Halogenated Fluorocarbon
IPCC	Intergovernmental Panel on Climate Change
LCA	Life cycle assessment
NBP	Net biome production
NEP	Net ecosystem production
NPP	Net primary production
NSC	Non-structural carbohydrate
PFC	Perfluorocarbon
RME	Rape methyl ester
UNFCCC	United Nations Framework Convention on Climate Change
WHO	World Health Organisation
WTO	World Trade Organisation

Conversion Table

1 kg = 10^3 g
1 Mg = 10^6 g = 1 t
1 Gg = 10^9 g = 1 kt
1 Tg = 10^{12} g = 1 Mt
1 Pg = 10^{15} g = 1 Gt
1 kg N_2O = 44/28 kg N_2O-N ≅ 1.57 kg N_2O-N
1 kg N_2O-N = 28/44 kg N_2O ≅ 0.636 kg N_2O
1 kg CH_4 = 16/12 kg CH_4-C ≅ 1.33 kg CH_4-C
1 kg CH_4-C= 12/16 kg CH_4 ≅ 0.75 kg CH_4
1 kg CO_2 = 44/12 kg CO_2-C ≅ 3.67 kg CO_2-C
1 kg CO_2-C= 12/44 kg CO_2 ≅ 0.273 kg CO_2

Global warming potential (time horizon: 100 years)
1 kg N_2O = 310 kg CO_2-equivalents = 84.5 kg C-equivalents
1 kg CH_4 = 21 kg CO_2-equivalents = 5.73 kg C-equivalents
1 kg CO_2 = 1 kg CO_2-equivalents = 0.273 kg C-equivalents

1 Introduction

The average global surface temperature has increased by about 0.6 °C during the 20th century and extreme weather events have become more frequent in several parts of the world /IPCC 2001/. The atmospheric concentrations of the greenhouse gases (GHGs) carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) have increased by 31 %, 151 % and 17 % respectively /IPCC 2001, Bouwman et al. 2000/, within less than 150 years mainly as a result of human activities /IPCC 2001/. Stabilization of CH_4 and N_2O concentrations at today's levels would require reductions in their anthropogenic emissions of between 8 % (CH_4) to more than 50 % (N_2O) /IPCC 1996/. Since the late 1980s, increasing scientific evidence and public awareness of global climate change have brought the protection of the global climate system on the political agendas.

Agriculture and land use change contribute significantly to the anthropogenic emissions of GHGs. As demonstrated in Table 1, agriculture represents one of the major sources of non-CO_2 GHGs in Europe as well as in Germany and is a fundamental component of mitigation strategies that involve GHG reduction, carbon substitution and carbon sequestration /IPCC 1996/. Land use has changed in minor terms in Europe during the last decades /Rabbinge and van Diepen 2000/.

Table 1 Contribution of agriculture and land use change to anthropogenic emissions /Ritter 1999; Burniaux 2000/

Region	**Contribution to CO_2 [%]**	**Contribution to CH_4 [%]**	**Contribution to N_2O [%]**
World	25	55	>66
Europe (EU-15)	3	42	>50
Germany	1	40	>40

1.1 Political background

For more than a decade, policy has recognized climate change as one of the most critical dangers of a global dimension for human welfare and development. Therefore, policies for the mitigation of and adaptation to climate change have been discussed and formulated at all scales – global, European, national, and local. Political measures must be based on sound scientific grounds in order to reveal adequate, efficient and verifiable response. Science can and must contribute relevant information and methodological instruments for the formulation and implementation of strategies to abate climate change. In order to set the basis for assessing the mitigation potential for GHG emissions in agriculture, the political framework, within which suitable strategies shall be developed, has to be analysed. Both climate and agricultural policy have to be considered in order to address the needs of policy and the frame conditions in which actions will take place.

1.1.1 Climate policy

The drivers of climate change act at the global scale, so accordingly, climate policy has to be coordinated at the same level. The implementation of the global political targets requires an adequate response at the European and national level.

Global climate policy

In 1992, the United Nations Framework Convention (UNFCCC) was negotiated /UNFCCC 1992/. The convention took effect on 21 March 1994. By September 2000, 186 parties – among which all EU member states and the EU as a whole – have ratified the UNFCCC making it legally binding for them /UNFCCC 2001a/. The UNFCCC sets “an ultimate objective” of stabilising greenhouse gas concentrations in the atmosphere “at a level that would prevent dangerous anthropogenic (human-induced) interference with the climate system.” /UNFCCC 1992/. Among other obligations, parties to the convention have committed themselves to “develop, periodically update, publish and make available to the Conference of the Parties (CoP), national inventories of anthropogenic emissions by sources and removals by sinks of all greenhouse gases not controlled by the Montreal Protocol”, using comparable methodologies (Article 4.1(a)). A standard method to be applied was developed, regularly extended and updated by the Intergovernmental Panel on Climate Change (IPCC) aiming at a harmonised, simplified spreadsheet model with global applicability /IPCC 1995, 1997/. Developed countries included in Annex I of the UNFCCC also shall “adopt national policies and take corresponding measures on the mitigation of climate change, by limiting its anthropogenic emissions of greenhouse gases and protecting and enhancing its greenhouse gas sinks and reservoirs” (Article 4.2(a)). The commitments of the Annex I countries were specified in the Kyoto Protocol /UNFCCC 1997/. Anthropogenic GHG emissions have been increasing in most Annex I countries since 1990 /UNFCCC 2001b/, making the actual reduction targets more ambitious.

Greenhouse gases in the context of the UNFCCC are restricted to trace gases directly increasing the radiative forcing in the atmosphere and having an anthropogenic origin. They form a „Basket of GHGs“: CO_2, CH_4, N_2O, HFC, PFC, and SF_6 /UNFCCC 1992/.

The CoP6 conference at The Hague and its resumed session in Bonn aimed at the implementation of the Kyoto Protocol by setting up rules for financial mechanisms, compliance, methods and accountable measures. One of the major obstacles arose from disagreement upon whether or not the terrestrial carbon sink shall be accountable and if yes, how and to what extent /UNFCCC 2000/. So agriculture and forestry has turned out as the key sector for a future success in the negotiations. The Bonn agreement sets maximum accountable amounts for carbon sequestration in a special annex table and operates with complicated caps and discounts for land management measures and non-domestic actions.

EU Climate Policy

For the European Union (EU), /Ritter 1999/ reports relative contributions of the major GHG species to global warming at 80 % CO_2 (CO_2 emissions minus CO_2 removals), 11 % CH_4 and 9 % N_2O.

Not only the Member States, but also the EU as a whole are Parties of the UNFCCC with its own obligations for reporting, monitoring and the reduction of GHG emissions by –8 % below 1990 level. The European Union decided to share the burden of a reduction of GHG emissions by –8 % below 1990 levels unequally among the member states, giving Germany a major target of –21 % below 1990 level (”Burden Sharing Agreement” reached by EU environment ministers in June 1998). The EU Member States have made clear that additional action at Community level is vital for them not only to meet their Kyoto commitments but also to respect the EU burden-sharing agreement of June 1998 /EC 1999a/. A number of EU guidelines and official documents have been prepared to ensure that the EU will meet its Kyoto targets. An important step was the creation of a “Monitoring mechanism of Community CO_2 and other greenhouse gas emissions“ /EC 1999b/ to ensure timely and harmonised reporting. The EU has outlined a strategy for GHG reduction /EC 1999a/ and is working on its implementation /EC 2000a/ in all economic sectors, including agriculture.

German Climate Policy

For Germany, the relative contributions of the major GHG species to global warming are 87 % CO_2 (CO_2 emissions minus CO_2 removals), 8 % CH_4 and 5 % N_2O. HFCs, PFCs and SF6 together amount for less than 0.5 % /Ritter 1999/.

Germany has two different commitments to fulfill. The Kyoto commitment requires a GHG reduction by –21 % below 1990 level until 2008 to 2012 according to the European burden sharing agreement. In addition, Germany has set an even more ambitious goal to reduce its CO_2 emissions by –25 % until 2005 /BMU 2000/ to which the German government committed itself at the Berlin Climate Summit in 1995. The German National Climate Protection Programme /BMU 2000/ sets detailed sectoral targets, most of which shall be realised by the promotion of renewable energies and increased energy use efficiency, energy savings, and volontary measures by industry. So far, agriculture was mentioned in the area of bioenergy and biogas production only.

In order at least to double the share of renewable energy sources in total energy consumption by the year 2010 /BMU 2001/, the German government created financial incentives for investment in infrastructure and a legal framework to promote the use of renewable energies in power plants. In the context of the Ecological Tax Reform, a programme to promote the use of renewable energy sources was launched in 1999 (Förderprogramm für "Maßnahmen zur Nutzung erneuerbarer Energien" of the „Marktanreizprogramm für erneuerbare Energien“). Among other goals, this programme financially supports the construction of certain heat or combined heat and power production facilities fired with solid biomass. Cheap credits and partial funding are offered to construct biogas production facilities based on residues from agriculture, forestry and fishery /BMU 2001/. The programme also funds the production and use of biomass for energy and material in projects using biofuels and biogenic lubricants in environmentally sensitive areas.

The Renewable Energy Sources Act ("Act on Granting Priority to Renewable Energy Sources", „Erneuerbare-Energien-Gesetz“) launched in April 2000 sets a legal framework to promote renewable energy sources by market regulations /BMU 2001/. Grid operators are obliged to purchase a certain percentage of energy from renewable sources and to pay compensation depending on the energy source and region and in a degressive mode with increasing size of the power plant. Specifically, biomass-fired plants can achieve 0.17 to 0.20 DEM kWh^{-1}, and biogas receives a minimum payment of 0.13 to 0.15 DEM kWh^{-1}.

1.1.2 Agricultural policy

As shown in Table 1, agriculture belongs to the most important sources of non-CO_2 GHGs. Therefore, the political framework of agricultural production is discussed as a basis to assess the potential of GHG mitigation. Agriculture belonged to the first areas of Common European Policy and still the European political level performs the strongest control on agricultural activities in the EU Member States.

Global agricultural policy

On a global level, WTO and GATT negotiations set the framework of agricultural trade and commerce which European and national agricultural policy has to adhere to. Through market forces, global agricultural negotiations have an indirect impact on agricultural production systems and on the extent at which environmental and socio-economic issues are considered by regional and national agricultural policies.

European agricultural policy

The Common Agricultural Policy (CAP) of the European Community is based on the EEC contract of Rome (EEC contract, Art. 39, para. 1, 1957). Since then, the agricultural production increased in the EU until a surplus over the internal demand was reached in all agricultural sectors (cereals, milk and meat) in the early 1980s. The increasing costs of agricultural subsidies for exports led to CAP reforms in order to adapt the production to demand by the milk quota regulation (1984) and volontary set-aside areas with compensation payment (mid 1980s). In order to regulate agricultural production more efficiently, the CAP reform 1992 was put into force, taking account of WTO and GATT negotiations.

Important elements of the CAP reform 1992 were structured along two pillars: (1) Community-level measures towards more direct, yield-independent payments and (2) environmental, social and structural measures to be implemented at national levels. The measures to reduce the deficiency payment and the financial support of exports aim to promote the liberalisation of the agricultural market and to compensate for negative income effects by direct payment (yield-independent payment per agricultural land areas and animals) and through fixed, market-dependent rates of set-aside areas to reduce the overall agricultural production. Additional measures to address environmental, structural and social goals were introduced, which were implemented with regional differentiation by the Member States.

The Berlin Summit in March 1999 resulted in the adoption of new regulations which came in force in 2000 as Agenda 2000 CAP Reform /EC 2000b/, strengthening direct payments for compensation of further reduced yield-based market support prices. Environmental goals and a new rural development framework were integrated in the CAP. As a consequence, according to the impact analyses in /EC 2000b/, the cereal area will expand due to the reduction of the compulsory set-aside area, and the oilseeds area will decline. There will be a slight decline in beef production, and further slight decrease in dairy cattle numbers since the increase in milk quota will be lower than the increase in milk yield per cow. The increasing trend in pigs and poultry production will continue.

The CAP has a pure impact on the production in the heavily regulated agricultural subsectors of sugar and milk only, where there are quota systems. In all other subsectors, external factors influence the regulatory efficiency of the CAP reform by changes in the prices achievable on the world market and by fluctuations in the currency exchange rates /Zimmermann 1997/. These effects overcome much of the regulatory effect of the CAP measures. Therefore, any changes in the production of crops and meat will result from the combined pressure of the economic environment – global markets, GATT and WTO, and internal EU markets as well as CAP measures.

So far, the CAP has largely failed to address the mitigation of and adaptation to climate change in its measures because it has never intended to do so. However, climate change is entering slowly in the agricultural policy objectives in EU member states. New efforts for integration of climate protection in agricultural policy are being made /EC 2000a/.

German agricultural and forest policy

Most of German agricultural policy is tightly related to the EU policy but has some freedom in environmental, social and structural measures.

German agricultural policy already considers the mitigation of climate change at some extent. The German National Climate Protection Programme /BMU 2000/ estimates that the afforestation of agricultural land and maintenance of existing forest can sequester 30 Tg CO_2 (8.2 Tg C) between 1990 and 2010, and that the production of biogas by anaerobic fermentation may reduce the emissions by another 1.4 Tg CO_2-equivalents. /BML 2000/ analyses a much broader spectrum of mitigation options which fit to the Sustainability Strategy of the German Federal Ministry of Agriculture. The options concentrate on ruminants, biogas, enhanced fertiliser use efficiency, maintenance of forests and promotion of bioenergy crops. Integrated measures like the extensivation of the agricultural production, promotion of organic farming, agri-environmental measures, investments, and the promotion of regional markets will also indirectly reduce the agricultural GHG emissions.

However, the integration of environmental issues and climate change in the agricultural policy on national and European level still offers considerable scope for improvement.

1.2 Mitigation of greenhouse gas emissions

In the political arena of climate change, scientists have played a major role in arguing for the urgency of climate change mitigation and in largely enhancing process understanding and developing methods for greenhouse gas accounting and mitigation. The implementation of the Kyoto Protocol requires adequate methods and instruments to quantify, monitor, and verify GHG emissions and their reduction. Science has to provide relevant information and to develop adequate tools to quantify emissions since 1990, to quantify, monitor and verify emissions reductions during the first commitment period from 2008 to 2012 and to guarantee adequate statistical certainty to proof the fulfillment of the Kyoto commitments on project, national and European scale. Furthermore, feasible options and the potential for GHG mitigation and the possible adoption rate in all sectors of human society have to be identified.

Three complementary – and somewhat contradicting – strategies can be pursued to stabilise the atmospheric GHG concentrations. These are /IPCC 1996/:

1. Carbon sequestration by enhancing sinks and reservoirs
2. Carbon substitution (use of renewables to substitute fossil energy sources and materials)
3. Reduction of anthropogenic GHG emissions

This way of classifying GHG reduction strategies applies to all sectors, including agriculture, energy, industry and households.

The three strategies differ in their characteristics and power to achieve the long-term goal to stabilise and reduce the atmospheric concentrations of GHGs. In the following, the assumptions, environmental implications, general potential and limitations of each of the strategies are outlined. Subsequently, their advantages and disadvantages and potential conflicts between them are discussed. This section aims to describe the general context of GHG mitigation in order to set the framework for the detailled analyses in the subsequent chapters.

1.2.1 Carbon sequestration

Carbon sequestration means to remove CO_2 from the atmosphere and store it in non-atmospheric pools such as the biosphere, lithosphere or in oceans. There is a set of theoretical options: large-scale iron fertilisation of oceans in order to enhance biological activities, and capturing and subsequent storage of fossil fuel-derived CO_2 emissions in the deep ocean or in geological reservoirs, e.g. in reservoirs from which natural gas is extracted /Watson et al.

2000/. The only option which is technically feasible at large scale at present state, however, is carbon sequestration in the terrestrial biosphere.

Assumptions

Carbon sequestration relies on the increase of non-atmospheric carbon pools. The natural global carbon cycle shows a large exchange between the oceans, terrestrial biosphere and the atmosphere. Human activities have increased the flux from the carbon pool in the lithosphere to the atmosphere by fossil fuel consumption and cement production. Carbon sequestration aims to revert the flux and, in turn, increase the carbon flux from the atmosphere to the oceans and biosphere. However, whilst carbon is bound in the lithosphere for millions of years, the biosphere represents an active carbon pool with turnover rates in the order of months to decades. Hence, carbon sequestration in the biosphere to offset fossil carbon emissions means to postpone the CO_2 increase in the atmosphere rather than reverting the trend by real long-term mitigation.

Environmental implications

General environmental implications of carbon sequestration measures are difficult to assess because they depend on the characteristics of the measures taken. If the sequestration measure aims at the conservation of the established carbon stocks over long periods of time, it will lead to a reduction of disturbance and of the intensity of human interference. In contrast, if a measure is optimised towards high short-term sequestration rates, it may go along with intensive management of plantations, fertilisation and drainage, which may lead to an intensification of disturbance. Potential indirect effects outside the scope of a sequestration project may occur, driven by changes in wood supply and market pressures. Therefore, carbon sequestration measures need a careful evaluation of the entire forest and agrifood sector to fully account for all direct and indirect effects on the carbon balance.

Potential and limitations

Carbon sequestration in terrestrial ecosystems has been extensively reviewed and debated /Watson et al. 2000/. Globally, the terrestrial biosphere is presently removing CO_2 from the atmosphere at a rate of 2.3 ± 1.3 Pg a^{-1} C (8.4 ± 4.8 Pg a^{-1} CO_2) /Watson et al. 2000/, which is equivalent to one third of the anthropogenic emissions of fossil CO_2. However, this largely natural sink is highly variable and uncertain in its present and future magnitude (e.g. /Watson et al. 2000/; /Valentini et al. 2000b/).

Assessing the potential for carbon sequestration in the terrestrial biosphere for meeting Kyoto commitments is presently hardly possible as long as the political framework has not yet been fully decided /UNFCCC 2000/. Policy makers have to resolve definitional issues: forest definitions, defintions of afforestation, reforestation, and deforestation, definitions of land management. These settings of political nature largely determine the magnitude of the terrestrial carbon sink accountable in the frame of the Kyoto Protocol. /Watson et al. 2000/ give a number of definitions and try to assess the impact of definitions on accountable sinks.

Further technical issues remain to be solved in the context of compliance with the Kyoto Protocol /Valentini et al. 2000b/:

- Uncertainty and interannual variability. The uncertainty in the terrestrial carbon sink is already high at global scale, but even increases at continental scales /Valentini et al. 2000b; P. Ciais, unpubl./. In addition, the sink strength shows an interannual variability in the order of 100 % of the sink itself, driven by climate variation. However, the direct drivers and their links to the carbon balance are not yet sufficiently understood.
- Attribution. A considerable part of the terrestrial carbon sink may originate from natural processes and at present, there are no methods available to distinguish between this "background", indirect human-induced (e.g. by nitrogen deposition, increased atmospheric CO_2 concentration) and direct human-induced carbon sequestration /Valentini et al. 2000b/.
- Risk of leakage. Depending on the political definition of carbon sequestration projects, there is considerable risk that carbon sinks taking place at the project scale will cause and increase in carbon emissions elsewhere by shifting the pressure on the land or on timber markets.
- Non-permanence. Viewing the limited mean residence time of carbon in terrestrial ecosystems /Schulze et al. 2000/, carbon sequestration by land use change and changes in land management represents a non-permanent strategy to stabilise the atmospheric CO_2 concentration and may be valid for some decades.
- Future behaviour. The terrestrial biosphere is sensitive to climate change and atmospheric CO_2 concentrations. There is some evidence from global models that the terrestrial carbon sink may decline or be even reverted to a source after some decades /Cox et al. 2000; P. Friedlingstein, unpubl./.

Despite the fact that there is considerable potential for human induced carbon sinks in European forests /Martin et al. 1998; Valentini et al. 2000a/ and agricultural ecosystems /Batjes 1998; Smith et al. 2000/ the scientific basis for accounting of carbon sinks in order to fulfill the commitments under the Kyoto Protocol is to be greatly improved. Carbon sequestration cannot compensate for fossil carbon emissions in the medium and long run, but can facilitate the adaptation to and mitigation of climate change by buying time.

1.2.2 Carbon substitution

Carbon substitution means the use of renewables replacing fossil carbon compounds in industrial production and for the provision of energy.

Assumptions

The carbon substitution strategy relies on the assumption that the carbon bound in agricultural and forest products takes part in the short term global carbon exchange between the atmosphere and the biosphere and a release of this carbon to the atmosphere as CO_2 by combustion will not increase the concentration of atmospheric CO_2. The CO_2 emitted from the combustion of e.g. biofuels has been previously taken up by the plants through photosynthesis, and will be taken up again during the next rotation period. This assumption is only true if the biomass is produced on an area where the carbon cycle is in equilibrium and if all carbon-containing byproducts are considered. This requires a constant amount of carbon stocks in biomass, litter and soil at landscape or national level, and constant carbon turnover rates over a period of the mean residence time of carbon in the atmosphere, i.e. of 100 to 150 years /IPCC 1996/. In particular, a constant amount of litter input to the soil and long-term steady state in soil disturbance by harvest, tillage and drainage characterise such equilibrium conditions. Soil memorizes disturbances over decades. Changes in land use and land management practice, associated with the increased production of energy crops, as well as interannual climatic variability affect the local carbon cycle, so the assumption of CO_2 neutrality is oversimplified for most biomass production systems.

Whilst the hypothesis of CO_2-neutrality has been common sense among promoters of renewable raw materials and bioenergy, it evidently contradicts the existence of a global and regional carbon sink in the terrestrial biosphere, i.e. its present non-equilibrium state /Watson et al. 2000/. Nevertheless, the hypothesis is valid if some restrictions are fulfilled: Fossil carbon released from the consumption of coal, oil, natural gas and limestone increases the global pool of active carbon since the reverse process of e.g. limestone or lignite formation acts at much longer time scales. In contrast, biospheric carbon is part of the active carbon pool. The biomass must be produced in a sustainable way, i.e. the removal of carbon at some area must be compensated by the growth of biomass stocks elsewhere. This precondition will be widely fulfilled in the intensively managed land use systems of Europe, but not entirely when savannahs and natural forests are used /Kaltschmitt 2000/. Also on-site effects by harvest on soil carbon stocks and their dynamics as well as trade-offs with regard to other GHGs and other environmental issues need to be considered /IEA Bioenergy 2000/. Within these restrictions and subtracting all possible “leakage”, the effect of carbon substitution on the atmospheric CO_2 concentration is permanent and irreversible.

Environmental implications

The European political and economic environment favours the production of biomass for energy on set-aside areas (Common Agricultural Policy, Agenda 2000, Rural Development Regulation 1257/1999), and the likely future increase in the area under biofuel production will change the common agricultural practice in terms of crop species and crop rotations. Options introducing herbaceous or woody perennials on arable land or reclaiming degraded land will sink carbon in biomass and soil, while options increasing the removal of biomass or the frequency of soil disturbance will lead to carbon losses /Mclaughlin and Walsh 1998; Borjesson 1999/. An analysis of the environmental effect of bioenergy must carefully consider any changes in biomass and soil carbon stocks driven by an increased demand for biomasss for energy. So far, the political and scientific perception of bioenergy has focused on carbon substitution (e.g. /Boman and Turnbull 1997; IEA Bioenergy 1999; Kaltschmitt 2000/) neglecting other greenhouse gases and other environmental effects except for approaches based on life cycle assessment (LCA) /Kaltschmitt and Reinhard 1997; Hanegraaf 1998/.

Biofuels and biogas produced in agriculture risk environmental trade-offs similar to common cultures. Agricultural production necessarily relies on input of energy and nutrients, so there are intrinsic environmental trade-offs in any crop, including biofuels. The agricultural nitrogen cycle leaks to groundwater (nitrate) and to the atmosphere (N_2, N_2O, NO, NH_3). Indeed, most of the nitrate in groundwater and surface waters is attributed to leaching and runoff from agricultural land. Biogas production is associated with the fermentation of waste, in particular with slurry from livestock production. The nitrogen cycle in the animal sector is even more leaky than in the crop production sector. For instance, 92 % of NH_3 emissions in European countries originate from agriculture /EEA 2000/. The nitrogen losses depend on the concentration of mineral nitrogen in the soil solution. The increase of mineral nitrogen, e.g. by fertilisation or by enhanced mineralisation rates, stimulates nitrogen losses from soil /Skiba and Smith 2000/. Annual bioenergy crops tend to demand higher nitrogen fertilisation and to have a lower nitrogen use efficiency than perennial grasses and woody coppice plantations, promoting hence nitrogen losses /Hanegraaf 1998/. The release of reactive compounds from agriculture to the regional environment results in eutrophication and acidification. Whilst the “classical” environmental effects of nitrogen losses have been considered in several studies /Kaltschmitt and Reinhard 1997; Bransby et al. 1998; Hanegraaf 1998/, biogenic emissions of N_2O from agricultural soils have been widely disregarded so far. This omission of one of the potentially most important sources of GHGs in the life cycle of bioenergy may significantly overestimate the GHG reduction efficiency of bioenergy options.

Biomass for energy has environmental implications. The environmental trade-offs associated with its production and use must be related to the overall effects in the entire life cycle, and assessed against the respective effects in the life cycle of fossil fuel options. In particular, all greenhouse gas species including biogenic N_2O emissions from the soil on which the bioenergy crops are produced need to be taken into account in order to adequately assess the value of carbon substitution by bioenergy for the abatement of GHG emissions.

Potential and limitations

The energy sector releases 92 % of the anthropogenic CO_2 emissions of the European Union /Ritter 1999/, so there is a high potential for reduction of fossil fuel emissions by substitution with renewable energy carriers. Indeed, the European Commission has set a target to double the share of renewables in the energy consumption to reach 12 % by the year 2010. The main contribution will come from biomass, tripling its present level of use /EC 1997/. Bioenergy will be analysed here due to its high potential and the moderate costs.

The potential reduction of fossil fuel-derived CO_2 emissions by introduction and enhancement of bioenergy ranges between 1.1 and 2.7 Pg a^{-1} C (4.0 to 9.9 Pg a^{-1} CO_2; 75 to 114 EJ a^{-1}) globally and between 0.13 and 0.24 Pg a^{-1} C (0.48 to 0.88 Pg a^{-1} CO_2; 8.8 to 9.9 EJ a^{-1}) in Europe /Kaltschmitt 2000/.

Carbon substitution can be seen as a repeatable long-term strategy with a significant potential for GHG mitigation in Europe. Environmental trade-offs and leakage with regard to the carbon cycle and the energy substitution effect have to be carefully taken into account. Life cycle assessment of promising substitution options offers a valuable tool for designing most energy-efficient and environmentally friendly carbon substitution systems.

1.2.3 Reduction of greenhouse gas emissions

Reducing GHG emissions attacks the anthropogenic contribution to climate change at its very source.

Assumptions

The reduction of anthropogenic GHG emissions represents the most straightforward strategy to mitigate human induced global warming. Reducing the input of GHGs to the atmosphere will dampen the increase in GHG concentrations and will stabilise or even decrease the atmospheric concentrations in the medium to long term if the reduction target is large enough.

Environmental implications

Measures to reduce N_2O emissions by optimising fertilisation rates with regard to nitrogen losses have positive side-effects on the release of other reactive nitrogen compounds such as ammonia (in the case of synthetic fertilisers containing ammonium or urea and of manure and slurry), nitrate leaching and associated environmental impacts on adjacent ecosystems. Such optimisation can mean reduced fertiliser doses, timing according to crop demand by frequent application of small doses, use of slow-release fertilisers and in particular, accounting for organic nitrogen in crop residues, manure and slurry. There is no such general rule for win-win options in the field of measures to reduce CH_4 from animal husbandry since some of them might oppose animal welfare or increase N_2O emissions. Therefore, in general, the whole basket of GHGs and additional environmental effects on farm and landscape scale has to be considered. Potential leakage through enhanced import of food and feedstuff can be assessed by monitoring of the national trade balance in the agrifood sector.

Potential and limitations

The agricultural sector offers a wide range of options to reduce the emissions of CH_4, N_2O and to a minor extent of CO_2 many of which are technically feasible /Minami 1997; OECD

1997; Kroeze 1998; Mosier et al. 1998a; Velthof et al. 1998; Löthe 1999/ and economically viable /Bates 2001/. However, there is a difference on principle in the controls of GHG emissions between agricultural and industrial sources. Industrial emissions arise from technical processes in which the physical and chemical characteristics are relatively well defined and the locations readily identifiable. In contrast, the major share of the agricultural GHG emissions originates from microbial processes in the rumen of animals, in manure and in soil, exhibiting a large spatial and temporal variability in flux rates. Farm management contributes to the emissions, but the final driving forces of the microbial processes – substrate and oxygen availability, moisture and temperature – are intimately linked to climate, weather, site properties /Smith et al. 1998; Skiba and Smith 2000/ and land use history /Mosier et al. 1998b/, many factors of which lie almost if not completely beyond farmer´s control. This particularity limits the maximum rate by which agricultural GHG emissions can be reduced and also makes a reliable prediction of the reduction efficiency difficult. In general, in Europe, CH_4 originating mainly from animals may offer the largest potential for reduction in the short run. The reduction of N_2O emissions is more complicated but feasible. Biogenic CO_2 emissions arise from drained peatlands, so their reduction is limited to these areas.

Uncertainty in emission inventories as a major obstacle

From a scientific point of view, the effect of GHG reduction must exceed the level of uncertainty in the trend of the GHG emissions estimate in order to be verifiable. The current uncertainty in national GHG emission inventories for the agricultural sector is estimated at 70 to 80 % for N_2O from agricultural soils and CH_4 and N_2O from manure management but only 30 % for CH_4 from enteric fermentation /Lim et al. 1999/, and additional uncertainty is introduced by trend analysis. In contrast, in order to fulfill the commitments under the Kyoto Protocol and specified in the European Burden Sharing Agreement, EU Member States have to verify changes in their national total anthropogenic emissions in the range of 0 to –28 %. Evidently, the uncertainty in the agricultural emissions estimates must be considerably reduced before the effect of measures to reduce GHG emissions in the agricultural sector can be documented and verified at the national scale. The present standard methodology for GHG inventories /IPCC 1997/ distinguishes in the agricultural sector between enteric fermentation, manure management, rice paddies and burning of residues as CH_4 sources and agricultural soils, manure management and burning of residues as N_2O sources, respectively. The emission factors and default values for some activity data have been designed for global application or for large regions such as “developed countries” or “Western Europe” /IPCC 1997/. There is hence considerable scope for improvement with regard to completeness, the degree of detail and disaggregation of sources and climatic regions of Europe and accuracy and precision of the emission factors and activity data in order to make the inventory suitable to monitor emissions reductions in agriculture. Intensive and widespread measurement programmes of N_2O emissions from agricultural soils during the late 1990s have created a valuable experimental basis to develop a more detailed methodology adjusted to the European agri-environmental conditions. Unfortunately, most of the long-term experiments rely on manual chamber measurements performed in average on weekly intervals, so their value for an enhanced understanding of the rapidly changing microbial processes is limited. The data are, however, suitable for the analysis of statistical relations between N_2O emission rates and controlling factors. Long-term studies of N_2O and CH_4 emissions from manure management have been performed by few groups within Europe only. Therefore, there is only restricted potential for improvement of emission factors. Animal houses represent an omitted source of GHGs to include in future emission inventories.

1.2.4 Comparison of strategies

Carbon sequestration, carbon substitution and the reduction of GHG emissions represent all feasible, but contrasting and sometimes contradicting strategies to mitigate climate change. In practice, a mixture of the approaches will be needed in order to assure timely action and sufficient social acceptance of measures to fulfill the commitments of the Kyoto Protocol.

In order to develop efficient and reliable GHG mitigation strategies, the nature and environmental implications of the strategies have to be fully assessed and compared (Table 2). Among the characteristics summarised in Table 2, the most critical differences arise from:

- Different nature: Carbon sequestration may only buy time while both carbon substitution and GHG reduction represent long-term, cause-oriented solutions. Carbon substitution has to fulfill the criteria of sustainable production of the biofuels or biomaterials and minimise trade-offs with regard to non-CO_2 GHGs. If reduction options directly address all GHGs, no major indirect effects on the climate are to be expected.
- Ease of implementation: Carbon sequestration measures and to some degree also carbon substitution do not affect dramatically the daily life of people in industrial countries, so their acceptance will be high. In contrast, some of the GHG reduction options will require significant changes in agricultural production and human consumption patterns as well as in the daily energy consumption.
- Ease of assessment, monitoring and verification: The different nature of the three strategies and the risk of leakage and negative environmental side effects require adequate, adjusted methods and instruments for assessment, monitoring and verification of the impact of the strategies. In the case of carbon sequestration, all stock changes or fluxes in the forestry and agrifood sector, including trade balances, have to be accounted. A similar method is needed to ensure that biofuels really replace fossil fuels rather than being used as an additional source of energy. In this case, the entire national and European energy sector, including the trade balance, has to be considered. Life cycle assessment of the various carbon substitution options has to identify the rate at which carbon substitution effects are counteracted by emissions elsewhere in order to derive a discount factor for accounting under the Kyoto Protocol. The assessment, monitoring and verification of

GHG reduction measures can restrict itself to the scope of the respective measure in a given sector.

Table 2 Comparison of GHG mitigation strategies

	Carbon sequestration	**Carbon substitution**	**GHG reduction**
Strategy	Increase non-atmospheric carbon stocks	Substitute carbon from passive, fossil sources by carbon from reactive biogenic sources	Reduce anthropogenic GHG emissions
Time horizon for applicability	< 50 to 100 years	Infinite	Infinite
Quality of strategy	Reversible Carbon sink saturates Potential other environmental benefits	Irreversible Partial trade-offs against other GHGs and environmental issues	Irreversible Potential other environmental benefits
Repeatability	Not repeatable	Repeatable	Permanent effect
Reliability	High uncertainty	Moderate uncertainty	Moderate uncertainty
Risk of leakage*	Yes National carbon budget or forest and agricultural trade balance can minimise the risk.	Yes National energy budget can minimise the risk.	Minor, only if agricultural production mismatches the food and feed demand. Watch the national trade balance in the agrifood sector.
Implementation	Immediate or with some years delay. Financial incentives and area available for sequestration measures needed.	A few years delay to produce sufficient biofuels and to develop the market.	Immediate or with some years delay to change the management. Financial incentives and favourable agricultural policy and market needed.
Costs			
Implementation	Very low	Low to moderate	Low to high
Monitoring	?, depends on political set of definitions and on required level of uncertainty	Low	?, depends on measure and on required level of uncertainty
Verification	?	Low	?
Accountability for Kyoto commitments	All or part of carbon sequestered may be accountable, depending on measures and caps. Temporary option only in order to gain time (decade(s)) to develop permanent strategies.	Large part of carbon substitution can be accounted, depending on the energy carriers in the substitution system. Biofuels must be produced in a sustainable way.	Fully accountable
Link to other strategies	Opposes carbon substitution	Opposes carbon sequestration. May go in line with GHG reduction measures.	Offers carbon substitution through anaerobic digestion of farm wastes. May offer areas for carbon sequestration or to grow biofuels.
Sectors affected	Agriculture, forestry, food	All sectors	Sector of measure, e.g. agriculture

* in this context, leakage means the indirect release of GHG outside of the GHG mitigation project, but triggered by its implementation

Clearly, short-term benefits with regard to the ease of implementation have to be valuated against long-term costs and efforts to account for reliable measures.

1.2.5 Indirect greenhouse gas emissions from agriculture

The agricultural system also shows indirect feedbacks with the climate system. The increased atmospheric CO_2 concentration stimulates photosynthesis – an effect potentially larger in agricultural systems than in natural ecosystems facing nutrient or water limitations. Changes in the carbon and nitrogen cycles have been recorded for a wide range of ecosystems /Torbet et al. 1997; Korner 2000/. However, they can presently not be adequately quantified to be considered in the assessment of GHG mitigation options in agriculture.

Nitrogen cycling in the agricultural and food chain is highly inefficient in Europe. Agricultural land represents a source of reactive nitrogen compounds affecting other ecosystems through eutrophication and acidification. The indirect fertilisation effect of the airborne deposition of NO_y and NH_y alters the carbon and nitrogen cycle in natural ecosystems with still unknown effects on long-term changes in biodiversity and ecosystem functions. The deposition of NH_y represents a clearly agricultural indirect effect due to the emission of NH_3 from animal excreta and fertiliser. Accordingly, the eutrophication of groundwater and surface waters by nitrate leached from fertilised soils has to be attributed to the agricultural system. Eutrophication promotes the formation of N_2O in natural and other ecosystems affected by agricultural nitrogen losses.

The /IPCC 1997/ guidelines for national greenhouse gas inventories therefore assign emission factors to these indirect N_2O sources and specify the impact by airborne nitrogen deposition (cf. Chapter 3) versus nitrogen leaching. According to the /IPCC 1997/ methodology, it is assumed that on average, 30 % of the nitrogen input to agricultural land is lost by leaching and runoff. The nitrate is hypothetically traced to the ocean by an emission factor of 0.025 kg N_2O-N (kg N input)$^{-1}$, which is partitioned to groundwater (0.015 kg N_2O-N (kg N input)$^{-1}$), rivers (0.0075 kg N_2O-N (kg N input)$^{-1}$) and estuaries (0.0025 kg N_2O-N (kg N input)$^{-1}$) /Groffman et al. 2000/. Applying the /IPCC 1997/ methodology to a global N_2O budget, /Mosier et al. 1998a/ attribute about a third of the agricultural emissions to indirect sources, 75 % of which are in turn released from leaching and runoff. The methodology has been heavily criticised /Nevison 2000; Groffman et al. 2000/ because of a poor data base from which the emission factors were derived and for the general misconception that the N_2O-to-nitrate ratio in agricultural drainage and groundwater would not change during transport. Most of the studies cited in /Groffman et al. 2000/ suggest that the emission factor for groundwater in the /IPCC 1997/ guidelines overestimates the actual emissions by an order of magnitude.

In contrast, /Naqvi et al. 2000/ warn about potential long-term increase of N_2O formation in coastal marine waters, which have so far been disregarded in the /IPCC 1997/ approach. Eutrophication of surface waters and hypoxia in bottom water has been increasing in many coastal areas. Extrapolating from intensifying anoxia and N_2O production over the western Indian continental shelf, /Naqvi et al. 2000/ hypthesize that eutrophied coastal marine waters could contribute significantly to the indirect anthropogenic global warming.

However, more measurements and integrated studies on watershed level are needed before the indirect agricultural N_2O emissions can be captured in inventories at adequate uncertainty levels.

Therefore, subsequently, the analysis will focus on direct GHG sources and sinks in agriculture only. Indirect GHG emissions will be treated as environmental side effects.

1.2.6 Potential and limitation of agriculture for greenhouse gas mitigation

Agriculture belongs to the most important sources of CH_4 and N_2O, so there is also significant scope for GHG mitigation. The European Environmental Agency recently summarised the initiatives within EC member states to reduce CH_4 and N_2O emissions /EEA 1999/. There were only few and scattered references to agriculture, and evidently no coordinated effort to involve agricultural production systems in mitigation strategies.

Potential role of agriculture in the three mitigation strategies

Carbon sequestration. Afforestation and revegetation of abandoned agricultural land and of long-term set-aside areas as well as changes in land management towards the conservation and increase of soil carbon offer a range of options for carbon sequestration.

Carbon substitution. Renewable materials and biofuels grown on agricultural land as well as the production of energy by anaerobic digestion of farm wastes currently belong to the most promising options for GHG mitigation.

GHG reduction. The role of agriculture in GHG mitigation has already been appreciated by some official EU documents, but the full extent at which GHG reduction measures could and should be implemented has not yet been comprehensively discussed.

Methane. The reduction of CH_4 emissions represents an efficient and fast-response mitigation measure /EC 1996/ due to the high Global Warming Potential (GWP) and the relatively short residence time of CH_4 in the atmosphere. Some of the most promising CH_4 mitigation options mentioned in this political paper address agriculture – slurry covers, anaerobic digestion, and changes in feed composition.

Nitrous oxide. According to /EC 1999a/, ”Common and co-ordinated measures should be taken to reduce N_2O emissions from agriculture”. As low cost measures, reduced fertiliser applications and improved manure management are mentioned.

Carbon dioxide. Although there is considerable potential to reduce CO_2 emissions from farmed peatlands, none of the presently discussed GHG mitigation strategies address this source.

Scientific obstacles to implementation

Except for the promotion of bioenergy including biogas, few of the potential GHG mitigation strategies have been adopted at large scale and long-term change in management in European or German farming practice. Apart from political and socio-economic inertia, the lack of scientific knowledge, particularly the large uncertainty in the quantification of emissions and emissions reductions has still prevented the implementation of GHG mitigation measures although some of them are feasible and viable. Therefore, it is crucial to reduce the scientific and methodological uncertainty in the quantification of sources and sinks and to develop adequate tools for assessing environmental trade-offs such as leakage, indirect GHGs and other environmental side effects. There are two undispensable prerequisites for making GHG mitigation in agriculture attractive for implementation:

1) Reducing the uncertainty in the quantification of sources and sinks to the level required for meeting the German and European Kyoto commitments. At present, the uncertainty is particularly high whenever biogenic processes are involved in GHG sources and sinks.
2) Developing tools that make potential mitigation measures transparent and verifiable.

1.3 Aim and structure

As shown in the preceding chapters, climate change has led to a response by policy at global, European and national level. So far, climate policy has defined GHG reduction targets. The implementation of climate policy depends on an agreement about the rules for the implementation of the Kyoto Protocol at global level and on concrete actions at European and national level. Political action may efficiently reduce GHG emissions in the heavily regulated agricultural sector from which significant quantities of N_2O and CH_4 arise. Agriculture offers a wide range of potential GHG mitigation options, ranging from carbon sequestration, carbon substitution to the direct reduction of GHG emissions. Science must contribute to these processes with relevant information and adequate methods to quantify, monitor and verify GHG emissions and emissions reductions that allow to proof in a transparent, reliable manner whether the parties of the Kyoto Protocol fulfill their commitments.

1.3.1 Aim and scope

This dissertation contributes scientific information and methodologies to support the needs of policy to fulfill Article 4.1 (GHG reporting) and Article 4.2 (GHG mitigation) of the UNFCCC /UNFCCC 1992/ as specified in the Kyoto Protocol /UNFCCC 1997/. It aims to analyse essential components of efficient potential GHG mitigation strategies in the agricultural sector of the EU by identifying where important mitigation potential is located and what uncertainty, environmental ancillary effects and costs are associated with it. Measures analysed here encompass carbon sequestration, carbon substitution by bioenergy, and GHG reduction in agriculture through technical, socio-economic and political means. In order to achieve this, literature reviews, further development of methodologies for environmental assessment and GHG accounting, and calculations of GHG emissions and reduction potentials are performed. The lack of political guidance for definitions and accounting rules hampers the assessment of carbon sequestration, so here, the state of scientific knowledge is synthesized only. Bioenergy is analysed by a case study of the production and utilisation of an exemplary bioenergy crop. As a basis, the methodology of life cycle assessment is further developed and applied for evaluating environmental trade-offs of carbon substitution. The detailed assessment of agricultural GHG emissions represents the undispensable basis for decisions upon greenhouse gas mitigation strategies on national and European level. Therefore, a cause-oriented spatialized methodology for national GHG inventories is developed to allow a regionally adjusted quantification of biogenic GHG emissions and to reduce the uncertainty in the national and European estimates of agricultural greenhouse gas emissions. Based on the detailed inventory, the GHG mitigation potential in the agricultural sector of the European Union is quantified and assessed.

More specific, among the wide range of issues to be considered with regard to GHG quantification and mitigation, the following hypotheses are addressed in depth.

1. *There is some potential for carbon sequestration in agricultural soils, which may even provide environmental benefits. However, uncertainties associated with the sink strength, the anthropogenic contribution to the sink and its future behaviour are as high as the sink strength itself.*

2. *Emissions of greenhouse gases, especially of* N_2O*, and other environmental trade-offs associated with the production and use of biomass from annual crops as an energy carrier significantly reduce its benefit of being a largely climate-neutral option for carbon substitution.*

3a. *The uncertainty in the quantification of* N_2O *emissions from agricultural soils on national and European scale is significantly reduced by improving the methodology for the inventory calculation. It is important to distinguish at sub-national scale between climatological, pedological and management factors that control the annual* N_2O *release.*

3b. *A more detailed methodology compatible to the IPCC methodology to quantify biogenic* N_2O*,* CH_4 *and* CO_2 *emissions from agriculture allows to reduce the uncertainty in the national and European inventories to a degree that makes reductions verifiable under criteria of the Kyoto Protocol.*

4. *European agriculture theoretically offers a significant potential for GHG mitigation via a range of technical and market-based measures. Their implementation is most effective if based on local environmental and socio-economic conditions and if clear incentives are provided.*

1.3.2 Structure

Separate chapters are assigned for each of the highlighted issues. This structure follows the aspects under which agricultural greenhouse gas emissions are investigated in this dissertation – carbon sequestration, bioenergy for carbon substitution, statistical modelling of N_2O emissions from agricultural soils, development and application of a detailed methodology for inventories of GHG emissions from agriculture on national and European scale, and the subsequent application for the assessment and discussion of the GHG mitigation potential in agriculture.

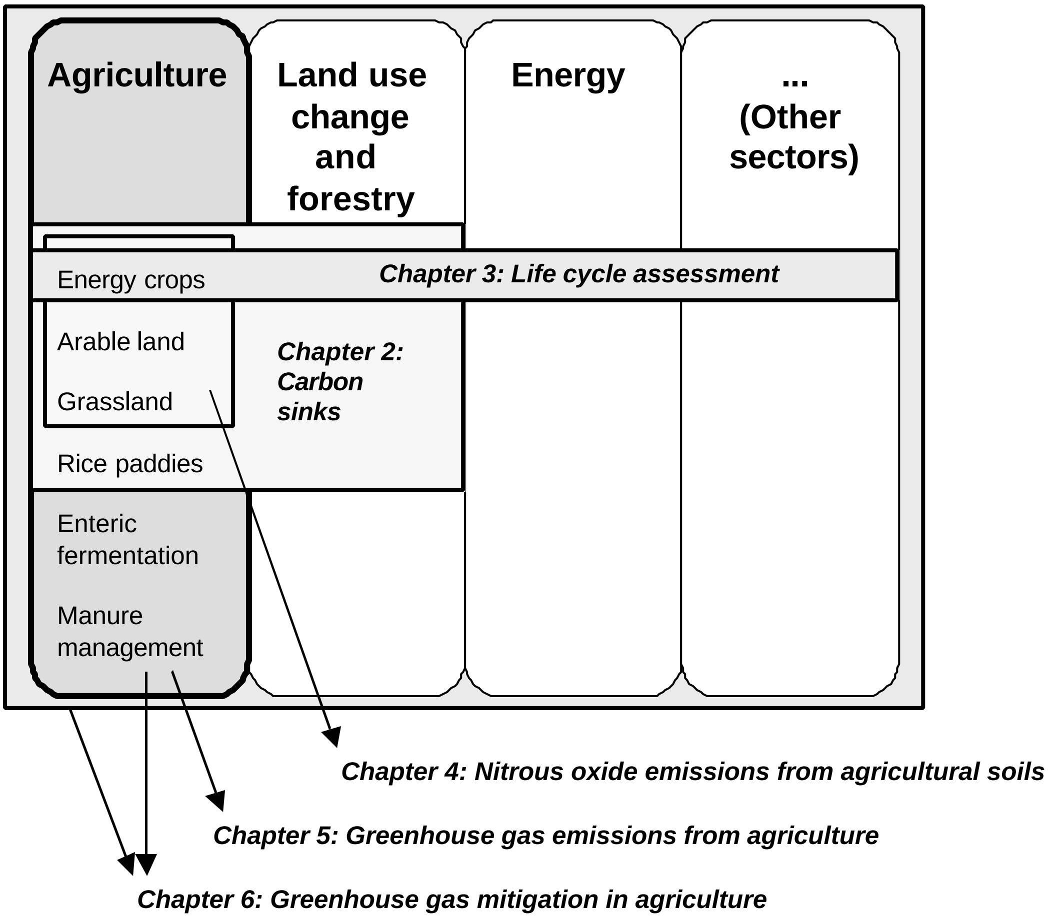

Figure 1 Scope of the aspects and related chapters with regard to the /IPCC 1997/ categories for national GHG inventories

Figure 1 illustrates how the different aspects and related chapters fit in the economic sector categories subscribed for reporting national GHG inventories according to the /IPCC

1997/ guidelines and how the methods to evaluate the three mitigation strategies carbon sequestration, bioenergy for carbon substitution, and GHG reduction address the various economic sectors. Carbon sequestration represents a mitigation option affecting parts of the agricultural and land use change and forestry categories. Here, the forestry sector is largely excluded from the analysis. But for a full monitoring and verification of carbon sequestration, carbon flows in the entire sectors, including trade, have to be considered in order to avoid leakage (cf. Chapter 1.2.1). In contrast, understanding the implications of carbon substitution by substituting fossil fuels through energy crops requires an in-depth analysis across all sectors in each of which it addresses small components only. Life cycle assessment (LCA) is an adequate tool to assess environmental trade-offs associated with carbon substitution (cf. Chapter 1.2.2) and will be used here. The verifiable reduction of GHG emissions from agriculture requires an improvement in the methodology for inventory calculation, reducing the uncertainty to a level that allows to detect reduction measures. Therefore, agricultural soils, being the largest source of biogenic GHGs, are studied in more detail, concentrating on N_2O rather than on the whole suite of GHG emissions. This is justified because N_2O dominates the GHG emissions from agricultural soils with a relative contribution of about 80 % (cf. Chapter 5). In order to develop a more detailed methodology for quantifying GHG emissions with reduced uncertainty, the entire agricultural sector as defined by /IPCC 1997/ is addressed. The results are discussed against the emissions from the entire national and European economy /Ritter 1999/ and in terms of potential opportunities for GHG mitigation and their environmental consequences.

Chapter 2 reviews the options for carbon sequestration in agricultural soils through changes in land use and land management and discusses their potential, environmental implications and suitability to form measures accountable under the Kyoto Protocol.

Chapter 3 investigates important but largely neglected environmental aspects of bioenergy by the example of a heat provision from whole cereal crops substituting light oil. The emissions of the volatile nitrogen species nitrous oxide (N_2O), nitric oxide (NO_x), and ammonia (NH_3) respectively their resulting environmental impacts are emphasized. First the applied methodology of a life cycle assessment (LCA) is discussed. The emissions of volatile nitrogen species and their resulting environmental impacts are subsequently quantified. Sensitivity analyses are performed to quantify the effects of uncertainty in the underlying assumptions and emission calculations, which is particularly high for N_2O.

Given the importance of N_2O emissions from agricultural soils and the present uncertainty in the IPCC emission factors, a more accurate and precise statistical methodology to quantify the N_2O release under European agri-environmental conditions is developed in **Chapter 4**. Based on a review of long-term N_2O field studies in Europe, major soil, climate and management controls of N_2O release from agricultural mineral soils in the European Union are identified. In order to make the methodology feasible for application by national administrative bodies, all parameters used in the models shall easily be gathered from statistical services. Using stepwise multivariate linear regression analysis as far as possible, empirical models of N_2O emissions are established which allow – in contrast to existing large-scale approaches – a regionally disaggregated estimation of N_2O emissions from agricultural soils on sub-national to continental scale in Europe.

Chapter 5 quantifies the biogenic direct greenhouse gas sources and sinks in the agriculture of EU member states on a sub-national (NUTS 1) /EUROSTAT 1999/ resolution by using Europe-specific emission factors and refining the /IPCC 1997/ approach where sources and sinks are missing (animal houses, CH_4 sink in soils), where emissions can be further disaggregated (manure management) and where local conditions interfere with emission factors (N_2O emissions from agricultural land). The results are used to test the credibility of the national reports submitted by EU member states under the UNFCCC /Ritter 1999/. The multi-gas approach pursued here represents a valuable tool for decisions upon GHG reduction strategies because it allows to quantify side effects of potential measures and interactions between the greenhouse gases and creates basic information required for the development of a cost-efficient integrated climate protection policy.

Chapter 6 compiles all findings of the preceding chapters in order to discuss comprehensively the importance of biogenic GHG emissions from agriculture and of their mitigation in the context the total anthropogenic emissions in European member states and the European Union as a whole. Following the description and assessment of promising GHG mitigation measures in agriculture, conclusions are drawn whether and how the findings can serve as a basis for decisions upon greenhouse gas mitigation strategies involving the agricultural sector.

The procedure outlined above allows optimally to assess the GHG mitigation strategies in agriculture in a comprehensive way in order to provide relevant scientific tools and information that may help to improve the quality of the national and European GHG inventories and to implement climate policy in the agricultural sector.

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2 Potential and Risks of Carbon Sequestration in the Biosphere

Abstract

Based on a review of the carbon sequestration options suitable for implementation in the agricultural system of the European Union, the fundamental limitations of offsetting fossil fuel emissions by biological carbon sinks are discussed. Options for carbon sequestration by land use change and changes in land management are described and assessed with regard to magnitude of the carbon sink, the possible implications for the release of non-CO_2 *greenhouse gases and other environmental effects. There is significant potential for carbon sequestration in Europe, which is maximised by a mix of measures including conservation tillage, organic amendments, and in particular regeneration of woodlands and afforestation. However, there is a fundamental drawback of carbon sequestration in the long run. The sink may saturate or reverse to a source under changing climatic or management conditions. So carbon sequestration in the terrestrial biosphere only buys some time, but cannot offset fossil fuel emissions in the sense of a full greenhouse gas mitigation option. Some measures, however, could be adopted for other environmental reasons. As a consequence, accounting for carbon sinks in the frame of the Kyoto Protocol has been politically approved but still awaits a scientifically sound set of rules for accounting, monitoring and verification.*

2.1 Introduction

Globally, terrestrial ecosystems are presently taking up carbon from the atmosphere at a rate of 2.3 ± 1.3 Pg C a^{-1} /Watson et al. 2000/, which is equivalent to one third of the anthropogenic emissions of fossil CO_2. However, this sink varies interannually on plot to global level by close to 100 %. The terrestrial carbon cycle is strongly driven by natural climatic variation such as oceanic oscillations that occur on decadal time scales. The uncertainty in the sink strength is not only inferred by such periodical short- and medium-term variability but also in longer time horizons. The unknown feedback between the biosphere and changing radiation, water cycling and atmospheric CO_2 concentration makes predictions of its future behaviour highly speculative /Cox et al. 2000/. Viewing also that the average mean residence time of carbon in terrestrial ecosystems is between 15 and 100 years /Schulze et al. 2000/, carbon sequestration by land use change and changes in land management will most probably represent a non-permanent strategy to stabilise the atmospheric CO_2 concentration for some decades.

2.1.1 Carbon stocks

For the estimation of current and future carbon sequestration potential it is essential to analyse both the carbon stocks and carbon fluxes as well as turnover rates of terrestrial ecosystems /Watson et al. 2000/. Carbon sequestration is defined as a continuous increase of the carbon stocks in biomass and soil.

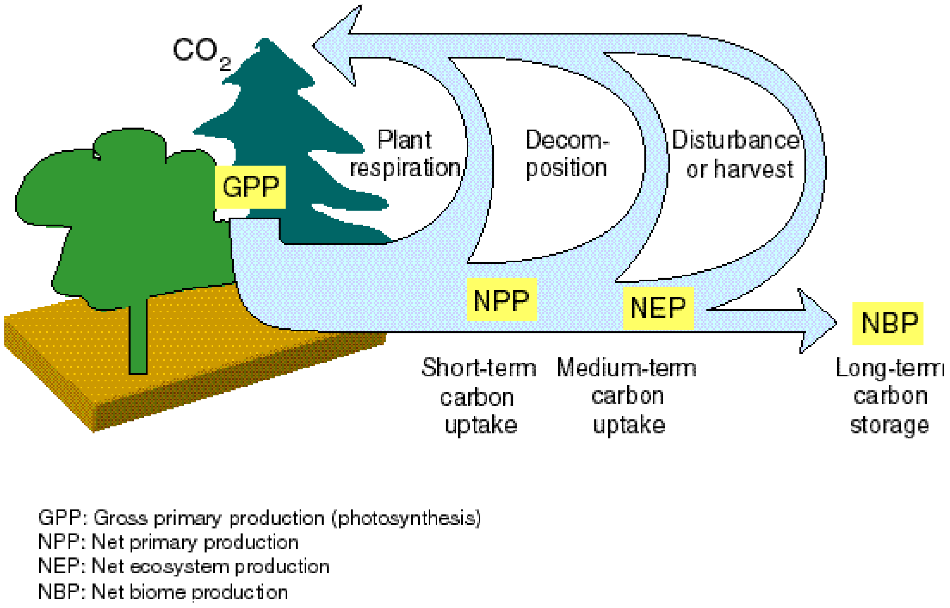
Carbon stocks identify the present reservoirs and indicate historical changes in the terrestrial carbon stocks by land use change. Table 3 summarises the global carbon stocks in major terrestrial ecosystems. Soil represents the major organic carbon reservoir in the terrestrial biosphere. Wetlands have the highest carbon density, so despite their relatively small spatial extension, the protection of carbon stocks in wetlands represents an important tool to avoid further release of CO_2 from the terrestrial biosphere. The high soil carbon stocks of terrestrial grasslands have accumulated and are still accumulating in steppe climates where decomposition is limited by drought and low winter temperatures. Typical Western European grasslands tend to show lower soil carbon stocks than given in Table 3.

Table 3 Carbon stocks in global ecosystems /WBGU 1998/

Biome type	Area	Global carbon stocks [Pg C]			Carbon stock per unit area [Mg C ha^{-1}]		
	[Mio ha]	Total	Soil	Vegetation	Total	Soil	Vegetation
Grasslands	1250	304	195	9	243	236	7
Cultivated land	1600	131	128	3	82	80	2
Wetlands	350	240	225	15	686	643	43
Terrestrial biosphere	15115	2477	2011	466	164	133	31

2.1.2 Carbon fluxes

Carbon fluxes have to be related to pools and turnover rates (Figure 2). Plant photosynthesis ("gross primary production", GPP) removes CO_2 from the atmosphere to the biosphere. Half of the CO_2 is quickly returned to the atmosphere by autotrophic respiration. The reminder is the net production of organic matter by plants in the ecosystem, called "net primary production" (NPP). Decomposition of organic matter in the ecosystem, losses by herbivores and heterotrophic respiration in soil further deplete the carbon pool. The "net ecosystem production" (NEP) denotes the remaining net annual accumulation of carbon in an ecosystem, which amounts to 10 to 75 % of NPP /Schulze et al. 2000; Watson et al. 2000/. We can assume that NEP carbon will stay in natural ecosystems for years to decades, which is equivalent to the time frame of commitment periods in the Kyoto Protocol. Agricultural ecosystems, managed forests and natural ecosystems also experience carbon losses through disturbance by harvest, pest calamaties, fire or windthrow. The real carbon sink that occurs at landscape scale is the so-called "net biome production" (NBP), which is globally about 1 % of NPP and about 10 % of NEP /Watson et al. 2000/.



GPP: Gross primary production (photosynthesis)
NPP: Net primary production
NEP: Net ecosystem production
NBP: Net biome production

Figure 2 Carbon budget of terrestrial ecosystems (adapted from /WBGU 1998; Watson et al. 2000/

2.1.3 Carbon sinks in the Kyoto Protocol

The Kyoto Protocol /UNFCCC 1997/ allows the accounting of carbon sequestration in terrestrial ecosystems to fulfill part of the parties´ commitments. According to the Kyoto Protocol, the afforestation of agricultural land since 1990 would fulfill the requirements to be accounted under Article 3.3 while all other agricultural measures like no-till systems, improved pasture management etc. apply to Article 3.4.

2.1.4 Aim and scope

Against this background of science and policy, the literature is reviewed in order to describe measures for enhanced anthropogenic carbon sequestration and to assess the sequestration potential and associated environmental risks. Focus lies on agricultural ecosystems in Europe. The analysis anticipates also methods and results of Chapters 4 and 5 and discusses them in the context of their potential to fulfill the national commitments under the Kyoto Protocol. Accounting for biological carbon sinks is a non-trivial issue, so the basic underlying concepts and methodologies are also discussed.

2.2 Options for carbon sequestration

Ecosystems sequester carbon whenever the carbon uptake exceeds the losses. Carbon sequestration is determined at a decadal time scale by an increase in carbon stocks, or at shorter time scales by a net flux from the atmosphere into the ecosystem. Human activities affect the rates of uptake and loss, providing options for enhancing the terrestrial carbon sink. Two categories of options can be distinguished: 1) land use change and 2) land management.

Land use change refers to any land of which the main function is altered through human activities. Typical land use change activities encompass afforestation of agricultural land, deforestation of pristine forests, conversion of permanent pasture to arable land, or abandonment of agricultural land.

Land management is the system of practices on a piece of land. If land management changes in order to enhance the carbon sequestration rate, the main productive purpose of the land remains the same as prior to the change.

2.2.1 Land use change

The conversion of forests and temperate grasslands to cropland in historical times has already released considerable amounts of carbon from vegetation and soil. In general, the cultivation of a native soil to continuous cropping results in the disruption of large aggregates and the loss of organic carbon and associated nutrients /Tisdall and Oades 1982/. Processes involved are the mineralisation of organic matter formerly protected within large aggregates /Elliott 1986/, enhanced turnover of soil organic carbon /Dalal and Mayer 1986/, and low input of litter and root biomass /Oades, 1984/. /Matson et al. 1997/ estimate that 50 % of the carbon stocks in the 0 to 20 cm topsoil layer of a temperate grassland is depleted by cultivation.

Conversion of cropland to grassland can lead to an increase of the carbon content in the topsoil. In order to maintain this grassland, it must be used either to feed the existing herd of domestic ruminants, or to increase the herd. Both options may be connected with enhanced methane emissions from animal metabolism. These trade-offs will probably overcompensate the benefits of carbon sequestration in terms of global warming. Alternatively, if the area is taken out of the food and feed production, the mown hay can be used as biofuel - which has so far rarely been done for economic reasons, although it is technically feasible.

Set-aside areas represent a short term sink of carbon since under conditions of the present Common Agricultural Policy (CAP), the major part of which it be cultivated again after one or few years. In the strict sense of carbon sequestration, they do therefore not represent a suitable option for climate change mitigation worth pursuing in the context of meeting the commitments under the Kyoto Protocol. In addition, the potential dimension of set-aside areas is limited by the pressure from other types of land use.

Afforestation is currently seen as the most effective option for carbon sequestration on large areas. Young, growing forests act as a significant sink of carbon by biomass growth while the increase in the soil carbon content is delayed. The soil carbon pool could even be depleted during the first years after establishment. Afforestation can effectively sequester carbon on a short to medium term time scale (decades) although in the start-off phase, the plantation may act as a source of carbon /Thuille et al. 2000/. But once the new equilibrium of

CO_2 assimilation and release is established, the long-term target of sustainable forest management and forest protection will result in a very slow or negligible further increase in C storage only. So the short and medium term climatic benefit goes along with a long-term demand for forest maintenance to be effective and to represent a real sink through increased carbon stocks. Extracting wood from such a forest would annulate the major part of the sink and would hence need to be debited to the national carbon balance. For illustration, the above-ground carbon stock in the vegetation of a cereal field amounts to about 4.5 Mg C ha^{-1} /FNR 1999/, which is annually produced and also at large extent removed from the field. The carbon stock in the standing biomass of a mature temperate forest is about 60 Mg C ha^{-1} (40 to 180 Mg C ha^{-1}) /Larcher, 1984/, of which carbon in wood has been accumulating over an 80 to several hundred years period. Furthermore, litter input to forest soil exceeds by far the input by crop residues and manure on farmed soils, and typically a litter layer forms above forest soils, with turnover rates of months to years. Changes in the soil profile and the apparent density of soil horizons make direct comparisons of soil carbon stocks in agricultural and forest soils difficult. Moreover, the soil carbon content generally depends highly on litter input. In the topsoil (0 to 20 cm) of temperate forests, about two thirds of the typically 60 to 70 Mg C ha^{-1} have turnover rates of 20 to 50 years while one third or less is relatively stable with a turnover rate above 2000 years /Perruchoud et al. 1999/. Hence, a great part of sequestered carbon in soil will rapidly react on sudden changes in litter input as occur after logging, harvest or deforestation.

There is one single irreversible win-win option of land use change for a limited agricultural land area: In order to reduce CO_2 release from peat oxidation by drainage of organic soils and wetlands, an extensivation of land use towards ungrazed grassland and preferably the complete abandonment of agriculture with subsequent rewetting of the areas will turn a strong CO_2 emittor into a more efficient and longer term sink than mineral soils will ever be. Rewetting also dramatically reduces N_2O emissions. These benefits with regard to global warming are slightly reduced by an increase in CH_4 emissions. The restoration of cultivated organic soils towards fens and bogs, however, is a long process taking decades to centuries.

The main pressure on soils comes from types of land use other than agriculture or forestry. Changing agricultural land to construction sites will irreversibly reduce the carbon supply on these areas.

2.2.2 Land management

Carbon stocks in cropland can be increased by increasing the volume of biomass and constraining carbon decomposition in soil. Various options are available for this. Their application depends upon regional conditions such as temperature, distribution and level of precipitation, and water limitation. Examples for management options in Europe are /Watson et al. 2000/:

- Improved crop production and erosion control
  - Partial elimination of bare fallow
  - Irrigation, water management
  - Organic amendments (biosolids, manure, straw)
  - Forages in rotation
  - Ley-arable farming
  - Conservation tillage
  - Grazing management
- Species introduction (including legumes and deep-rooted species)
- Fire management on extensive pastures
- Afforestation for renewable materials

At best, a combination of improved or reduced tillage, nutrient and water management and erosion control seems appropriate for any conditions.

2.3 Potential and risks of carbon sequestration

2.3.1 Methodology and assumptions

Based on the present state of knowledge, the potential for carbon sequestration in agricultural soils in the EU is estimated. Key practices and the potential rate of annual carbon gain (Table 4 column A) are derived from /Paustian et al. 1998; Smith et al. 2000; Watson et al. 2000; Sauerbeck 2001/, all of which have performed global reviews of the carbon sink potential in agriculture. The mitigation rate (Table 4 column B) is calculated by a conversion of the annual rate of carbon gain to CO_2-equivalents. In this step, effects of carbon sequestration on other GHGs are taken into account. The soil-borne emissions of CH_4, CO_2 and N_2O are derived from a conservative application of the methodologies in Chapters 4 and 5. Column C indicates the period during which carbon sequestration will take place before the sink saturates. The potentially available agricultural area in Europe (Column D) is estimated from /EUROSTAT 1999/ and /Smith et al. 2000/. Then, the maximum annual mitigation in Europe can be quantified. The rate combines the non-permanent effects of carbon sequestration with the effects on CH_4 and N_2O. Alternatively, the total carbon sink in Europe over the entire sequestration period is calculated.

The global classification of ecozones and related carbon sequestration rates given in /Watson et al. 2000/ (Table 4) is translated to EU member states as follows:

- Temperate-dry: Portugal, Spain, Italy, Greece
- Temperate-wet: Ireland, UK, Belgium, Netherlands, Luxembourg, France, Germany, Austria, Denmark
- Boreal: Sweden, Finland

2.3.2 Land use change

Land use change areas represent a minor fraction of the European agricultural surface /Rabbinge and van Diepen 2000/. /Liski et al. 2000/ estimate the effect carbon in trees of afforestation, reforestation and deforestation activities (ARD) in the EU to be a source of 5.4 Gt a^{-1} C or a sink of 0.1 Gt a^{-1} C depending on the definition of ARD in the Kyoto Protocol.

Nevertheless, there is some potential for carbon sequestration by land use change (Table 4). The regeneration of natural woodlands has by far the largest potential because this is the only option in which both aboveground biomass (wood), belowground biomass and soil organic carbon are enhanced. There is some evidence that this process is currently ongoing at relatively large areas of the Mediterranean, where abandoned agricultural land reverts to Macchia. Here, it is assumed that the agricultural set-aside lands are permanently abandoned.

Restoration of wetlands offers the second largest potential, which basically refers to the rewetting of drained peatlands. In detail, the conversion of wetlands from farmed organic soils to restored wetland will reduce the emission of CO_2 and N_2O by 13 to >20 Mg ha^{-1} a^{-1} CO_2-equivalents (Chapter 5), but provoke additional CH_4 emissions in the order of 1 to 4 Mg ha^{-1} a^{-1} CO_2-equivalents, which represents the average and higher flux rates of boreal fens and bogs given in /Bartlett and Harriss 1993/. Hence, most of the mitigation effect originates from GHG reduction rather than from carbon sequestration.

All other options of land restoration or conversion of cropland to grassland have a limited potential for carbon sequestration although some systems can act at high initial rates. Restoration of salinated areas may go along with increased fertilisation promoting in turn N_2O release, which compensates the carbon gain.

As a conclusion, if land use change continues in the same direction as it is occurring in Europe at present days, the net result of land use change is more likely to be a reduction in carbon stored in vegetation and soils or neutral than an uptake of atmospheric carbon compounds. However, dedicated measures of carbon seqestration through land use change on present agricultural land will sink up to 28 Tg a^{-1} C in Europe, while the concomitant GHG mitigation could achieve 140 Tg a^{-1} CO_2-equivalents. This is equivalent to 35 % of the agricultural GHG emissions from EU-15.

Table 4 Potential of carbon sequestration by land use change and associated impacts (1 Mg = 1 t; 1 Tg = 1 Mt)

Ecozone	Key practices	Rate of C gain [Mg C ha^{-1} a^{-1}] (Confidence)[a]	Mitigation rate [Mg CO_2-equ. ha^{-1} a^{-1}]	Duration [a]	Available agricultural area in Europe [Mio ha]	Max. annual mitigation in Europe [Tg CO_2-equ. a^{-1}]	C gain in Europe [Tg C][b]	Associated impacts
		A	B	C	D	=B*D	=A*C*D	
Land use change								
All /1/	Restoration of wetlands	0.1-1 (L)	10-20	>100	3.69	37-74	37	++ CH_4, --CO_2, --N_2O +A, +B, +D
All /1/	Restoration of eroded and salinated land	0.1-7 (M)	0	30	3.69	0	11	+N_2O
Temperate -dry /1/	Marginal cropland re-seeded to grassland	0.3-0.8 (H)	0.3	50	1.26	0.4	19	-CH_4, -N_2O? +A, +E
Temperate -wet /1/	Surplus-cropland seeded to grassland	0.5-1.0 (M)	0.5	50	4.36	2.2	109	-CH_4, -N_2O +A, +E
EU-15 /3/, /Chapter 5/	Natural woodland regeneration	2-4	8-16	50	7.4	60-120	750-1500	--N_2O +A, +C, +E, -G, +H, +J, +K

/1/ /Watson et al. 2000/; /2/ /Sauerbeck 1993/; /3/ /Smith et al. 2000/

[a] L low; M medium; H high; [b] Agricultural land only

Impacts:

CH_4, CO_2, N_2O		+ increase; - decrease
A	Biodiversity	+ increase; - decrease
B	Water quality	+ increase; - decrease
C	Soil quality	+ increase; - decrease
D	Flooding	+ decrease; - increase
E	Erosion	+ decrease; - increase
F	Salinity	+ decrease; - increase
G	Productivity	+ increase; - decrease
H	Fertiliser use	+ decrease; - increase
J	Nitrate leaching	+ decrease; - increase
K	Pesticides	+ decrease; - increase

2.3.3 Land management

At global scale, optimised soil management on the arable soils of the temperate zone can increase the carbon stock by 10 Mg ha^{-1} C over a period of 50 to 100 years /Sauerbeck 1993/, corresponding to an increase in soil carbon levels by some 10 %. According to /IPCC 1996/ estimates, improved management, if maintained over a period of 50 to 100 years, would create a global sink potential for the sequestration of 20 to 40 Pg C (0.43 to 0.88 Pg a^{-1} C). 50 to 75 % of the lost soil carbon can be sequestered again /IPCC 1996/. This global sink in croplands could at best compensate about 28 % of the fossil CO_2 emissions from the European Union /Ritter 1999/ or 11 % of the global anthropogenic CO_2 emissions /Watson et al. 2000/. However, the capacity of agricultural soils to sequester additional carbon is finite and limited to the next 50 to 100 years until a new soil organic carbon equilibrium is reached /Paustian et al. 1997/.

Constraining the analysis to Europe, the following assumptions are made: The maximum annual mitigation and maximum carbon gain in Europe rely on the assumptions that European agriculture tends to be more optimised with regard to farm input and maintenance of the soil carbon balance than agriculture in other parts of Eurasia. Therefore, the lowest rate of C gain given in column A (Table 5) is taken. In order to calculate the potential, the maximum available agricultural land in a given climate region of Europe for a respective option is taken. This means the restoration of all farmed organic soils, the assumption that 5 % of croplands are eroded or salinated, 5 % of croplands are marginal, and 10 % of croplands are surplus according to the rate of set-aside land under the Common Agricultural Policy (CAP) and areas for improved land management as given by /Smith et al. 2000/. There are contrasting results whether organic amendments increase the release of N_2O or not. The effect is assessed as a slight increase of N_2O by 0.24 Mg ha^{-1} a^{-1} CO_2-equivalents, the effect of increased fertilisation by 0.48 and 0.72 Mg ha^{-1} a^{-1} CO_2-equivalents in temperate-dry and temperate-wet conditions, respectively.

Changing cropland management to no-till or low-till, optimal use of organic amendments at relatively high rates on part of the croplands, and shifting organic amendments from grassland to cropland allow to mitigate up to 100 Tg a^{-1} CO_2-equivalents. In contrast, carbon sequestration by more intense use of grassland can even lead to a source of GHGs through increased energy use and fertilisation. The most important land resource for carbon sequestration is surplus arable land if it is converted to long-term alternative land use such as woodland or bioenergy coppice. No single option as shown in Table 5 can sequester the optimum rate of carbon whereas integrated combinations of measures, especially those including bioenergy options and the restoration of peatlands, show a considerable (non-permanent) potential for carbon sequestration of up to 60 Tg a^{-1} C (220 Tg a^{-1} CO_2-equivalents) during the coming 50 to 100 years in EU-15 /Smith et al. 2000/, which is equivalent to 6 % of the European anthropogenic CO_2 emission from fossil sources.

The potential for carbon sequestration by forest management seems to be larger than in agriculture.

Table 5 Potential of carbon sequestration by cropland management and associated impacts

Ecozone	**Key practices**	**Rate of C gain [Mg C ha^{-1} a^{-1}]** (Confidence)[a] A	**Mitigation rate [Mg CO_2-equ. ha^{-1} a^{-1}]** B	**Duration [a]** C	**Available agricultural area in Europe [Mio ha]** D	**Max. annual mitigation in Europe [Tg CO_2-equ. a^{-1}]** =B*D	**C gain in Europe [Tg C][b]** =A*C*D	**Associated impacts**
Cropland management								
Boreal /1/	Ley, perennial forage crops, organic amendments	0.3-0.6 (M)	0.5	30	4.81	2.4	43	+N_2O +C, +G
Temperate -dry /1/	Low-till, less bare fallow, irrigation	0.1-0.3 (H)	0.14	30	25.3	3.5	76	+N_2O +C, +E, +G, -K
Temperate -wet /1/	Low-till, fertilisation, cover crops	0.2-0.6 (H)	0.5	25	43.6	22	220	+N_2O +C, +E, +G, -K
EU-15 /1,3/	Conservation or no-tillage	0.34-0.40 (H)	1.0-1.8	50	63.8-73.7	74-83	1100-1300	+N_2O? +C, +E, -K
EU-15 /1,3/	Organic amendments[c]	0.2-1.0 (H)	0.3-1.8	50	< 73.7	22	600-700	+N_2O? +C, +E
EU-15 /3/	Extensification	0.63	2.7	50	21.3	58	670	-N_2O +A, +C, +E, -G, +H, +J, +K
EU-15 /2,3/	Opt. soil management	0.1-0.2		50-100	73.7		370-550	
EU-15 /3/	Optt. land management			50	73.7		2900[d]	
Grassland management								
Temperate -dry /1/	Grazing management, fertilisation, irrigation	0-0.3 (L)	-0.3 (source)	50	13.6	source	0	+N_2O, +CH_4? +G, -F
Temperate -wet /1/	Grazing management, fertilisation, new species	0.4-2.0 (L)	0-0.1	50	30.5	1.5	610	++N_2O, +CH_4? +G

/1/ /Watson et al. 2000/; /2/ /Sauerbeck 1993/; /3/ /Smith et al. 2000/

[a] L low; M medium; H high; [b] Agricultural land only; [c] cereal straw, animal manure, sewage sludge;

[c] scenario includes afforestation, organic amendments and low tillage.

Impacts:

CH_4, CO_2, N_2O		+ increase; - decrease
A	Biodiversity	+ increase; - decrease
C	Soil quality	+ increase; - decrease
E	Erosion	+ decrease; - increase
F	Salinity	+ decrease; - increase
G	Productivity	+ increase; - decrease
H	Fertiliser use	+ decrease; - increase
J	Nitrate leaching	+ decrease; - increase
K	Pesticides	+ decrease; - increase

2.4 Environmental effects of carbon sequestration

Most of the measures analysed in Table 4 and Table 5 have significant environmental consequences with regard to other GHG species and local or regional environmental issues. Extensivation, the abandonment of surplus agricultural land and of farmed organic soils and perennial cultures for bioenergy as a more climate-friendly fuel source (Chapter 3) imply additional benefits for the reduction of biogenic GHG emissions besides the carbon sequestered. These range from reduced energy and nutrient losses to reduced nitrogen leaching and a potentially enhanced biodiversity on the field. In contrast, options involving the application of organic nitrogen sources such as amendments bear the risk of increased N_2O emissions which may even compensate the benefit of the carbon gain in soil if the sequestration rates are low. Many options tend to enhance the soil quality and reduce the risks of erosion while negative trade-offs are to be expected with regard to productivity and occasionally use of energy and pesticides.

Carbon sequestration in agricultural soils, however, goes along with simultaneous N sequestration which has the risk of enhanced N_2O and NO emissions through remobilization and denitrification. However, this effect will be probably small enough not to compensate or revert the carbon sink. Furthermore, no-till systems demand for higher input of chemicals than systems with mechanical weed and pest control.

2.5 Uncertainties in carbon sequestration

Unlike the emissions fossil fuel combustion, sinks constitute a major problem in accounting. Not only the effects of human induced management play a role in their behaviour, but so does natural and climate induced variability. Also, sinks are not necessarily a permanent feature of the biosphere. The carbon taken up by land management as discussed above may be released again through changing biological processes in the future or as a natural consequence of carbon cycling in the biosphere and thus be only a short term relieve measure. Whilst increasing the biological sinks is technically feasible, the challenge of a scientifically sound accounting and verification system in order to proof that the sequestration takes place and the sink is stable remains unsolved. Such technical issues are /Valentini et al. 2000b/:

- How can the extra carbon sink induced by human activities be separated from the natural sink (attribution)? Recent work suggests that a large part of the observed sink in the Northern Hemisphere may be due to changes in land use and land use management /Schimel et al. 2000/.
- Does a given carbon sink activity lead to carbon losses elsewhere (leakage)? The answer depends on political settings and definitions of carbon sequestration activities still to be decided.

- When will the carbon be released again (non-permanence)? Carbon pools in the terrestrial biosphere have an average residence time in the order of decades, so evidently, the terrestrial carbon sink not only saturates at the landscape scale, but alternates, at the plot scale, between source and sink phases. Any change in land use or land management affects the carbon stocks in the ecosystem. Disturbance rapidly depletes soil carbon and reduces biomass, which subsequently recovers slowly to the original stocks or to reach a new equilibrium. Translating this general mechanism to carbon sequestration on agricultural land means that carbon sequestered through land use change or land management will rapidly turn back to the atmosphere if land use changes again or if the carbon management on cropland is abandoned, e.g. if a no-till system is ploughed again after some years. There is hence a considerable risk that sequestered carbon, in particular in intensively managed agricultural systems, is rapidly released again to the atmosphere.
- How can the the carbon sink be verified in the commitment period 2008-2012 (verification)?
- Who takes the long-term responsibility for that sink (compliance)?

None of these issues can be answered in a satisfactory quantitative way at the moment.

2.6 Carbon sequestration in the context of the Kyoto Protocol

The European Community has agreed to reduce GHG emissions by 8 % in the Commitment Period 2008 to 2012 as compared to 1990 in accordance with the Kyoto Protocol /UNFCCC 1997/. In concrete terms, Article 3.3 and Article 3.4 of the Kyoto Protocol refer to agricultural measures applicable in Europe.

Article 3.3. "*The net changes in greenhouse gas emissions by sources and removals by sinks resulting from direct human-induced land-use change and forestry activities, limited to afforestation, reforestation and deforestation since 1990, measured as verifiable changes in carbon stocks in each commitment period, shall be used to meet the commitments under this Article of each Party included in Annex I. The greenhouse gas emissions by sources and removals by sinks associated with those activities shall be reported in a transparent and verifiable manner and reviewed in accordance with Articles 7 and 8.*"

Article 3.4. "*… The Conference of the Parties serving as the meeting of the Parties to this Protocol shall, at its first session or as soon as practicable thereafter, decide upon modalities, rules and guidelines as to how, and which, additional human-induced activities related to changes in greenhouse gas emissions by sources and removals by sinks in the agricultural soils and the land-use change and forestry categories shall be added to, or subtracted from, the assigned amounts for Parties*

> *included in Annex I, taking into account uncertainties, transparency in reporting, verifiability, the methodological work of the Intergovernmental Panel on Climate Change, the advice provided by the Subsidiary Body for Scientific and Technological Advice in accordance with Article 5 and the decisions of the Conference of the Parties. Such a decision shall apply in the second and subsequent commitment periods. A Party may choose to apply such a decision on these additional human-induced activities for its first commitment period, provided that these activities have taken place since 1990.”*

The afforestation of agricultural land fits to measures within Article 3.3 of the Kyoto Protocol if the activity has started in 1990 or later. Afforestation is accountable among the measures to fulfill the Kyoto commitments. All other management measures apply to Article 3.4 of the Kyoto Protocol to which a complicated accounting scheme with caps and discounts applies in the first commitment period 2008 to 2012. Accounting for carbon sinks activities under Article 3.4 is further complicated by the need to verify the additionality of the measures against a business-as-usual baseline.

The spirit of the United Nations Framework Convention on Climate Change (UNFCCC) is such that any measure negotiated within the Kyoto Protocol should have a clear effect on the stabilization of the CO_2 concentration in the atmosphere. Thus, a change in the biospheric sink through land use change and management should be clearly detectable as an atmospheric signal at some time and present a real reduction in emissions. If the scientific basis of monitoring in the context of the carbon cycle is not rigorous with respect to the overarching aim of reducing CO_2 concentrations in the atmosphere, the climatic impact may not be detectable /Valentini et al. 2000b/.

Evidently, the use of sinks in the Kyoto protocol be only a temporary measure in the spirit of a stabilization of the atmospheric CO_2 concentration to buy time for a long-term strategy to mitigate climate change by other means.

2.7 Conclusions

Despite the evident carbon sequestration potential in agricultural soils for the next 100 years the long-term benefit for the climate is questionable. Also terrestrial ecosystems can only be used to a limited extent as carbon sinks. It seems impossible to increase, over the long term and durably, the sink potential of ecosystems used for agricultural and forestry purposes at an extent that considerably exceeds the sink potential of natural potential vegetation /WBGU 1998/.

Although there is considerable potential and feasibility for human induced carbon sinks in European forests /Martin et al. 1998; Valentini et al. 2000a/ and agricultural ecosystems /Batjes 1998; Smith et al. 2000/ scientific arguments hence stand against this strategy in the context of long-term GHG mitigation. The terrestrial carbon sink is highly vulnerable to climate change and human interference. Its magnitude varies at all spatial scales from year to year in the order of 100 %, so the quantification of the sink strength involves large uncertainties. Furthermore, the inherent non-permanence of the terrestrial carbon sink and its rapid reversibility suggest that the risks of carbon sequestration in the biosphere offset its potential.

Some measures, however, could be adopted for other environmental reasons and will offer for some years to decades a mechanism to remove CO_2 from the atmosphere.

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3 Gaseous Nitrogen Compounds in the Life Cycle of Energy Crops[1]

Abstract

The reduction of energy-based environmental pollution is a declared goal of governmental action in all industrialised countries. In this context, the aim of this paper is to investigate the emissions of volatile nitrogen species respectively their resulting environmental impacts by the example of a heat provision from whole cereal crops substituting light oil. First the applied methodology of a life cycle assessment is discussed. The emissions of volatile nitrogen species and their resulting environmental impacts are subsequently quantified putting a special focus on the emissions of trace gases from biogenic sources in the agroecosystem. The comparison of a heat provision based on whole cereal crops versus light oil over the entire life cycle shows that heat from biomass results in a lower environmental impact regarding the anthropogenic greenhouse effect, but a higher one with respect to eutrophication, acidification and the depletion of the stratospheric ozone layer. These results may change if the energy crops are produced in extensive agricultural systems or if other mitigation measures are undertaken or if the emissions from the combustion in the biomass fired boiler are reduced.

3.1 Introduction

The substitution of fossil fuels by renewable energy sources represents one of the most feasible strategies to reach Germany´s commitments to reduce the anthropogenic greenhouse gas (GHG) emissions under the United Nations Framework Convention on Climate Change (UNFCCC) /UNFCCC 1992/ and the Kyoto Protocol /UNFCCC 1997/. Due to the significant potential of biomass as compared to other sources of renewable energy and due to the comparably low costs associated with its production and utilisation, biomass ranks among the most promising options /Kaltschmitt and Wiese 1997; Kaltschmitt 1997/. Whilst the public perception has been constrained to CO_2 emissions, a comprehensive ecological assessment of the substitution of fossil fuels by renewables must set its target beyond the CO_2 issue and consider a wide range of environmental consequences. If – as frequently argued in the

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[1] Published in German in: Umweltwissenschaften und Schadstoff-Forschung. Zeitschrift für Umweltchemie und Ökotoxikologie 10(6), 1998: 353-365 [Gasförmige Stickstoffemissionen im Lebensweg von Getreide zur Energiebereitstellung] Co-author: M. Kaltschmitt. Literal translation of the German paper.

political arena – biomass shall make a greater contribution to a future environmentally sound and climatically safe energy provision system, none of the environmental indicators shall deteriorate by the substitution of fossil fuels through biomass. In this context, a product-based ecological life cycle assessment (life cycle analysis, LCA) represents an adequate and well established method to compare the environmental impact of the production of a certain benefit (e.g. heat from bioenergy) with a substitutable one (e.g. heat from light oil).

Against this background, the aim of the following undertaking is to assess the release of gaseous nitrogen compounds (N_2O, NH_3, NO_x) throughout the entire life cycles of biomass and a fossil energy carrier by means of LCA because these species have attracted considerable attention in the context of the protection of Man and Biosphere. The analysis is performed as a case study of heat provision by annual crops versus light heating oil, which represents a typical, feasible option with relatively high risk of averse effects on the nitrogen cycle.

Existing studies used to neglect or drastically simplify the efflux of gaseous nitrogen compounds from nitrification and denitrification processes in soils induced by the application of fertiliser (e.g. /Kaltschmitt and Reinhardt 1997/). New findings in recent years make it now possible for the first time to establish a complete detailed balance of gaseous nitrogen compounds. Since knowledge about the non-energetic greenhouse gas emissions caused by energy crop production is slowly emerging, the biogenic agricultural sources deserve special attention. Keeping in mind the still great uncertainty in the data, the analysis infers the probable band width of emissions of volatile nitrogen compounds affected by varying production patterns of biomass substituting fossil fuels. In detail, the assessment addresses the life cycle of whole crops of Triticale and winter rye substituting light heating oil for heat provision in a heating plant. The emissions of volatile nitrogen species evolving in the entire life cycle are subsequently translated into their corresponding environmental effects, i.e. the depletion of the stratospheric ozone layer, the anthropogenic greenhouse effect, eutrophication and acidification of terrestrial and aquatic ecosystems. In order to quantify the range of potential emissions from the production of the cereals a “conventional” scheme with mineral fertiliser is compared with an “extensive” scheme based on legumes for biological nitrogen fixation. Moreover, sensitivity analyses are performed to test the impact of varying emissions of gaseous nitrogen compounds from soils and combustion on the overall results of the LCA. In a similar way, the effect of crop type and yields are studied.

3.2 Methodological basis

Product-based LCA is a powerful holistic concept to evaluate in a comprehensive way the potential impact of any products or service on environment and human health. Other aspects such as economic, social or cultural issues are excluded from LCA by definition /SETAC 1993; DIN 1996/. Life cycle assessment aims at a comparative evaluation of the ecological advantages and disadvantages of various alternatives serving the same purpose. The so-called life cycle comparison yields the difference in the results of the life cycle of heat provision by bioenergy to those of the life cycle relying on light heating oil based on the same useful energy which the heating plant delivers to the public heating system. The life cycle comparison hence determines whether the substitution of fossil fuel by bioenergy results in a potential environmental relieve or additional risk.

According to the concept of LCA, the entire life cycle (from cradle to grave) of a renewable energy source is compared with the one of a corresponding finite energy carrier, i.e. the environmental impacts of production, provision, use and likely recycling of the energy source (Figure 3). The same strategy applies to all products used in the life cycle. Basis of the life cycle comparison is the provision of an equal amount of utilisable heat delivered from the heating plant to a district heating system. Due to a different specific energy content, the amount needed may be different for the two compared energy carriers. The LCA is structured by several steps – goal and scope definition, inventory analysis, impact assessment according to impact categories, and interpretation /UBA 1995/. In the following, the general methodological approach to these steps is summarised and specified for this case study.

3.2.1 Goal and scope definition

General. The unambiguous definition of the aim is the undispensable prerequisite of the LCA. It determines the choice of the system parameters and of the scope; without an unequivocal definition of the aim no useful and interpretable results can be obtained.

Thereafter, further steps encompass the choice of the alternative services to be studied, the definition of the functional unit, the depth of the analysis, the contents of the balances of matter and of environmental impacts, spatial and temporal boundaries. These steps form the system definition. Since environmental effects can generally be analysed in relative terms rather than in absolute ones, a suitable basis for the comparison of alternative options has to be defined. The settings in the defintion of aim, scope and system boundaries determine largely the contents of the inventory analysis, impact assessment and interpretation.

Particular. This section describes the relevant aspects when applying the goal and scope definition to the case study of heat provision.

Goal. The LCA aims to evaluate the provision of heat from whole cereal crops produced in different ways against the use of light heating oil with respect to gaseous nitrogen compounds. Whole crops of Triticale and winter rye used for the provision of heat in a heating plant are compared with a heat provision by light heating oil in heating plants of the same size (Figure 3). Four options are studied (Table 6).

Table 6 Standard options of life cycle comparisons

Option 1	Triticale from conventional production – Heating plant	versus light heating oil – Heating plant
Option 2	Triticale from extensive production – Heating plant	versus light heating oil – Heating plant
Option 3	Winter rye from conventional production – Heating plant	versus light heating oil – Heating plant
Option 4	Winter rye from extensive production – Heating plant	versus light heating oil – Heating plant

Definition of the life cycles. The whole cereal crops studied here are produced in a conventional and extensive way, respectively, according to a management plan determining the necessary field work along the rules of good practice guidance. The means of production used in the course of the life cycle are traced back to their origins, e.g. to the production of crude oil in Saudi-Arabia. The whole cereal crops are subsequently harvested, pressed to round balls, transported, stored and finally burned in a heating plant (about 20 MW_{th}) in order to provide heat. The ash is disposed of as additive to fertiliser on the agricultural area and in a landfill, respectively. The biomass substitutes for light heating oil in a heating plant of equal thermal capacity. The mix of origins of heating oil corresponds to the contemporary German mix. Its life cycle is described in a comparable procedure.

Functional unit. All balances are calculated per unit of agricultural area (1 ha) because agricultural land is the limiting factor for bioenergy production in Germany. Thus, the basis of the life cycle comparison of biogenic and fossil energy carriers is the yield-dependent utilisable energy that can be produced by one hectare of a specific bioenergy carrier. The difference of the life cycles of biogenic minus fossil energy carrier gives the net emission gain or savings. Thus, the amount of a finite energy source substitutable by bioenergy derived from one hectare of biomass can be calculated (cf. /Kaltschmitt and Reinhardt 1997/). For further insight, not only the production area, but also the provided heat at the gate of the heating plant to the district heating system is used as a functional unit.

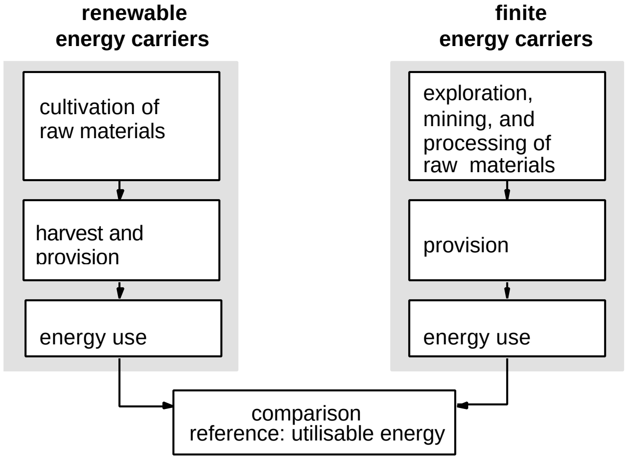

Figure 3 Comparison of bioenergy carriers with fossil energy sources

Reference system. The products or services addressed in the LCA are charged for the direct environmental effects only. In the case of the bioenergy crops, not all of the compounds released from the agricultural activities need to be considered, but only those directly evoked by the production of the energy crops themselves. The agricultural area interacts with the environment also independently from the production of energy crops. Therefore, the crop production has to be evaluated against an adequate reference system. In contrast, the reference system of the fossil energy carrier is set zero since a non-provision of light heating oil as opposed to its provision will not affect the environment.

For economic reasons, in Germany, bioenergy crops are reasonably grown on set aside areas or low productivity land. Expenditure for the production of the energy crops considered here is therefore compared with expenditure related to the management of a one-year grass fallow established according to good practice as a reference system. Emissions from this grass fallow are hence credited to the energy crop production system.

Boundary conditions. The energy crops are grown on a German virtual low quality land with a plot size of 5 ha. The soil has sandy texture, free drainage (continuous largely aerobic conditions), a topsoil pH of about 6 and about 3.5 % organic matter. The “Deutsche Reichsbodenschätzung” evaluates the yield potential as according to “Bodenzahl” 30. The site is managed by a pure crop production farm without application of manure according to common German practice. In order to analyse the impact of differing management systems, conventional management with need-oriented application of mineral nitrogen fertiliser is compared with an extensive management scheme purely based on legumes for nitrogen fixation. Major characteristics of the assumed farms are displayed in Table 7.

Table 7 Characteristics of the assumed virtual farm systems

		Triticale		**Winter rye**	
		conventional	**extensive**	**conventional**	**extensive**
Yields (whole cereal crops)	$t\ ha^{-1}\ a^{-1}$ DM	8.89	7.91	8.32	4.95
N content in dry matter	%	1.23	1.11	1.14	0.98
Nitrogen fertilisation[a]	$kg\ ha^{-1}\ a^{-1}$	113.2[b]	90[c]	108.6[b]	90[c]
Pesticides		yes	no	yes	no
Reference system: fallow		annual	bi-annual	annual	bi-annual

DM: dry matter;

[a] fertilisation according to demand;

[b] accountable application of mineral fertiliser;

[c] nitrogen fixation by a mixed bi-annual grass-legume intercrop

Only the difference between crop rotations with and without energy crops is attributed to the balance (cf. reference system). The balance covers the entire crop rotation cycle in order to account for crop residues in terms of emissions and their value as pioneer crop for the next crop in the rotation. In this sense the crop rotation with energy crops differs from the reference crop rotation with green fallow with regard to the energy crop and its preceding crop only; all other elements of the crop rotations are identical. Hence, the balance can focus on this differing part of the crop rotation only since no difference in environmental effects arises from the other elements of the crop rotation assuming identical management intensity.

When energy crops are produced without application of mineral fertiliser (*i.e.* extensive production), the energy crop can be grown on a given land area every second year only because legumes have to be grown as a pioneer crop first in order to provide the nitrogen through nitrogen fixation. As a consequence, twice as much area as in the conventional crop rotation is bound by energy crop production. For the LCA this means that the area-based yield of the energy crop is taken as half of the one given in Table 7.

An annual fallow seeded with cocksfoot (*Dactylis glomerata L.*) serves as the reference system of the conventional energy crop production system. The mown biomass remains on the surface as mulch. In the case of the extensive energy crop production system, nitrogen is provided via a mixture of grass and legumes (1:1, mainly cocksfoot and alfalfa (*Medicago sativa L.*)) maintained for more than a year. In the second year, the energy crop is grown. The related reference system is defined as bi-annual fallow seeded as mixed cocksfoot-dominated meadow.

System boundaries. The German standard in agriculture and firing technology is assumed for balancing the emissions. Products imported by Germany (e.g. fertiliser, crude oil and their respective prechains) are balanced accordingly. The latter emissions do not occur in Germany but rather in the original countries. Nevertheless, they have to be imposed to the production of heat studied here. The reference period is “mid of the 1990s”. Infrastructure (e.g. buildings, roads, machinery) is neglected here since it is assumed that it is used at a comparable degree in the course of both the life cycles of fossil fuel and energy crops, therefore avoiding significant bias in the life cycle comparison.

Uncertainty in the data and sensitivity analyses. Many of the data upon which the life cycles are based may vary in the agricultural practice in a considerable range. Therefore, the effect of the possible variation of important parameters on the result of the life cycle comparison is also investigated. As the standard life cycles comprise Triticale as well as winter rye whole energy crops, the impact of varying cultures and yields per hectare can be analysed, too.

Certain base data have a high degree of certainty; for instance, CO_2 emissions from the combustion of fossil fuels can be derived from the carbon inventory of the fuel with high precision. The opposite is true for volatile nitrogen compounds emitted from combustion. For instance, there are no measured data for NO_x emissions from the combustion of the whole cereal crop species underlying this study. As a consequence, the data measured for the combustion of winter wheat are applied in the standard life cycles. In order to assess the potential error associated with the varying nitrogen content of the biomass a sensitivity analysis is performed. Accordingly, the possible range of biogenic NO_x emissions from soil is accounted for and its effect on the result of the life cycle comparison is analysed. Whilst the N_2O emissions resulting from energetic and technical process can be estimated in a relatively certain way, /IPCC 1997/ reports an uncertainty of factor 9 for N_2O emissions from soil due to temporal variability and spatial heterogeneity. Therefore, the emission factor for N_2O from agricultural soils is varied according to /IPCC 1997/.

3.2.2 Inventory analysis

General. The inventory analysis quantifies the input and output parameters specified in the goal and scope definition in a detailed and differenciated way for all separate steps in the life cycle. In practice, reality has to be translated to a model in a way that makes the parameters of interest quantifiable. This is realised by the means of process chain analysis which allows to trace the incoming and outgoing flows of energy and matter for all partial processes and steps. In order to establish the balance the process chain is traced back far enough to ensure that neglecting further prechains and services will not introduce significant errors to the overall result.

Particular. According to the goal and scope definition, the emissions of volatile nitrogen compounds released in the course of the entire life cycles of the fossil and renewable energy carriers are quantified. Emissions from energetic and technical processes are distinguished from the biogenic sources in soil. Moreover, inventories of CO_2, CH_4, HCl and SO_2 are established, the results of which will be used in the impact assessment.

Emissions from energetic and technical processes. Emissions in this category are derived for all processes being part of the life cycles of the energy carriers studied here by according literature data (cf. /Kaltschmitt and Reinhardt 1997/).

Emissions from soil. In this category, NH_3, N_2O and NO are exlusively considered. For the non-nitrogen compounds it is assumed that the cultivation of energy crops as opposed to the reference system will not change the supply in soil nor emission rates.

Ammonia is emitted from arable soils immediately after the application of fertiliser containing ammonium. According to /Isermann 1990/ the nitrogen loss as NH_3 amounts to about 2 % of the applied mineral fertiliser at the site conditions specified above and assuming an average German mix of standard mineral fertiliser /Statistisches Bundesamt 1996/.

According to /IPCC 1997/, N_2O emissions released from ecosystems comprise emissions directly originating from the agroecosystem (N_2O_{direct}) and emissions from diffuse and diverse sources indirectly affected by agriculture ($N_2O_{indirect}$) as specified in equations (1) and (2).

$$E(N_2O_{direct}) = (F_{SN} + F_{BN} + F_{CR}) \cdot EF_1 \qquad (1)$$

with		
	E	Annual emission [kg ha^{-1} a^{-1} N_2O-N]
	F_{SN}	Nitrogen input by synthetic fertiliser [kg ha^{-1} a^{-1} N]
	F_{BN}	Nitrogen input by biological N fixation [kg ha^{-1} a^{-1} N]
	F_{CR}	Nitrogen input by crop residues [kg ha^{-1} a^{-1} N]
	EF_1	Emission factor for direct N_2O emissions [kg N_2O-N (kg^{-1} N)]

$$E(N_2O_{indirect}) = E(N_2O_{volatile}) + E(N_2O_{leaching}) = (NH_y\text{-}N + NO_y\text{-}N) \cdot EF_2 + NO_3\text{-}N \cdot EF_3 \qquad (2)$$

with		
	E	Annual emission [kg ha^{-1} a^{-1} N_2O-N]
	NH_y-N	Atmospheric deposition of nitrogen emitted in the life cycles as NH_3 and derivatives [kg ha^{-1} a^{-1} N]
	NO_y-N	Atmospheric deposition of nitrogen emitted in the life cycles as NO_x and derivatives [kg ha^{-1} a^{-1} N]
	NO_3-N	Leaching of nitrogen as nitrate from the agroecosystem [kg ha^{-1} a^{-1} N]
	EF_2	Emission factor for indirect N_2O emissions from atmospheric deposition [kg N_2O-N (kg^{-1} N)]
	EF_3	Emission factor for indirect N_2O emissions from nitrate leaching [kg N_2O-N (kg^{-1} N)]

Nitrous oxide emissions from soil are hence released either directly from the cropped field where fertiliser has been applied or at diffuse places by nitrogen input via air and ground water, indirectly induced by agriculture. Table 8 displays the respective emission factors according to /IPCC 1997/. The indirect N_2O emissions from atmospheric deposition are derived from the emissions of NH_3 and NO_x calculated in the life cycle comparison. As the energy crops are produced according to good practice, the respective amount of nitrate leached in the reference system and the energy crop production system can be set as similar. Both the fertilisation of the energy crops and breaking in the grassland of the fallow may promote nitrate leaching. Neglecting a potentially higher or lower nitrate leaching rate in the order of 10 kg ha^{-1} NO_3-N would infer an uncertainty of 0.3 kg N_2O-N in the overall balance only. Furthermore, the estimation of the indirect N_2O emission from nitrate leaching according to /IPCC 1997/ is prone to too large uncertainty to be applied in a meaningful way to a definitive fictive production site which the LCA relies on. Consequently, the respective accountable emissions are set zero (*i.e.* $N_2O_{leaching}$).

Table 8 N_2O emission factors according to /IPCC 1997/ in kg N_2O-N/(kg N-input and year)

	low	average	high
$E(N_2O_{direct})$: EF_1	0.0025	0.0125	0.0225
$E(N_2O_{indirect})$: EF_2	0.002	0.01	0.02
$E(N_2O_{indirect})$: EF_3	0.002	0.025	0.12

Oxides of nitrogen occur in soil mainly as a result of the microbial processes of nitrification and denitrification. Their formation depends therefore largely on soil temperature, soil moisture, soil oxygen availability and the level of fertilisation /Williams et al. 1992; Beese 1994/. The NO/N_2O ratio of nitrification in soil ranges between 1 and 5 /Bouwman 1990/ and of denitrification usually below 0.01 /Beese 1994/. Whilst a European estimate of soil-derived NO emissions applies an annual emission factor of 4.3 % of the fertiliser-nitrogen /Stohl et al. 1996/, recent measurements on sandy and loess substrates in the UK yielded emission factors below 1 % only /Skiba et al. 1993; Yamulki et al. 1995/. On the largely aerobic sandy production site underlying this study, nitrification will dominate N_2O release. Hence, for the standard life cycles, soil-borne NO emissions are conservatively estimated by a NO/N_2O_{direct} ratio of 0.5 – on a nitrogen basis. This is equivalent to an approximate NO emission factor of 0.6 % of fertiliser-nitrogen.

3.2.3 Impact assessment

General. The impact assessment determines the environmental consequences defined in the definition of aim, scope and boundary conditions. In general, the realisation refers to potential risks. Hence no real, actual and site-dependent effects, but rather potential consequences independent of site and ecosystem types are considered because the damage is generally virtual at the point in time when the LCA is performed. Consecutive effects or combinations of effects are presently considered at limited extent only.

Particular. The emissions of volatile nitrogen compounds quantified in the inventory analysis are assessed in this step with regard to their effects in the categories “stratospheric ozone depletion”, “anthropogenic contribution to global warming”, “acidification”, and “eutrophication”. These impact categories are defined as follows:

- Stratospheric ozone depletion. A large number of compounds contribute to the depletion of the stratospheric ozone layer. Of these, only N_2O occurs in significant amounts in the life cycles of energy crops and of the substitutable fossil fuels. Therefore, only N_2O is considered here.
- Anthropogenic contribution to global warming. Accountable greenhouse gas emissions occur in particular as CO_2 derived from the combustion of fossil fuels, as CH_4 and as N_2O. The CO_2 released from the combustion of biomass is considered as climatically neutral because it has been previously removed from atmosphere during plant growth. Methane and NO_x contribute also indirectly to global warming because they are involved in the formation of water vapour and tropospheric ozone /Brühl 1993; Lammel und Grassl 1995/. Due to the differing global warming potential of the gas species, the respective emissions are converted to so-called CO_2-equivalents. Assuming a time horizon of 20 years and German frame conditions, the emissions of CH_4 are weighted by a factor of 56 kg CO_2 (kg $CH_4)^{-1}$, N_2O by 280 kg CO_2 (kg $N_2O)^{-1}$ /IPCC 1996/ and NO_x by 126 kg

CO_2 (kg NO_2)$^{-1}$ /Lammel und Grassl 1995/. For NO_x, /Lammel und Grassl 1995/ report a possible range from 60 to 268 kg CO_2 (kg NO_2)$^{-1}$ under German conditions.

In addition, indirect climatic effects of CH_4 may occur, increasing the global warming potential by up to 30 % as a funtion of site conditions /Brühl 1993/. In contrast, aerosols react as cooling agents and hence reduce the anthropogenic contribution to global warming. However, at present knowledge, the contribution of NH_3, NO_x and their derivatives cannot be quantified.

- Eutrophication. The impact category eutrophication addresses the excess input of nutrients in soils, surface and ground waters. For the purpose of this study, the potential deposition of NH_3, NO_x and their derivatives is considered only because all other substances and compounds are not expected to yield differences in the life cycles.
- Acidification. Emissions of acidifying trace gases in the course of the life cycles underlying this study comprise basically SO_2, NO_x, NH_3 and HCl. The emissions are summarised as SO_2-equivalents (*i.e.* NO_x: 0.70 kg SO_2 (kg NO_2)$^{-1}$, NH_3: 1.88 kg SO_2 (kg NH_3)$^{-1}$, HCl: 0.88 kg SO_2 (kg HCl)$^{-1}$ /Heijungs 1992/). Nitric oxide is produced by microbial processes in soil /Williams et al. 1992/ and during combustion /Baumbach 1993/ in the form of NO, which then reacts rapidly in the atmosphere to form a stable NO/NO_2 equilibrium. The conversion of NO_x to SO_2-equivalents relies on nitric oxide in the form of NO_2.

3.2.4 Interpretation

General. Given the methodological difficulty and the limited acceptance of existing approaches to interpretation in LCA, which aim to synthesize the highly differing potential environmental impacts by an overall interpretation, this step is presently possible with great restrictions in the sense of LCA theory only /DIN 1996; UBA 1995/. Therefore, a “verbally argumentative” approach is taken here, i.e. a qualitative discussion of the advantages and disadvantages of the options investigated.

Particular. As a result of the problems mentioned above, no interpretation in the sense of LCA theory is performed here. However, the results obtained in the frame of the inventory analysis and the assessement of environmental effects are analysed and discussed in the individual steps of the life cycle and in the overall context.

3.3 Results and discussion

The individual life cycles are modelled under the frame conditions discussed in chapter 3.2 and the respective inventory analyses and their corresponding environmental impacts are quantified. The calculation of volatile nitrogen compounds released from ecosystems rely on the assumptions and steps identified in Table 7.

3.3.1 Inventory analysis

In the following, the results of the inventory analysis of volatile nitrogen compounds in the individual life cycles are illustrated (Figure 4). Table 9 documents the relative importance of the individual steps in the life cycles.

Table 9 N_2O emissions from soil (average emission factors according to /IPCC 1997/)

		Conventional production			**Extensive production**		
		Fallow	**Triticale**	**Rye**	**Fallow**	**Triticale**	**Rye**
Synthetic N fertiliser (F_{SN})	kg ha^{-1} a^{-1} N	-	110.9[a]	106.4[a]	-	-	-
N fixation (F_{BN})	kg ha^{-1} a^{-1} N	-	-	-	-	90	90
Green manure[b] (F_{CR})	kg ha^{-1} a^{-1} N	-	72	72	-	48[c]	48[c]
Crop residues (F_{CR})	kg ha^{-1} a^{-1} N	96	22	19	96	18	10
N_2O_{direct}	**kg ha^{-1} a^{-1} N_2O-N**	**1.2**	**2.5**	**3.9**	**1.2**	**2.0**	**1.8**
NH_y-deposition from LCC	kg ha^{-1} a^{-1} N		2.7	2.6		0.01	0.01
NO_y-deposition from LCC	kg ha^{-1} a^{-1} N		8.0	7.6		3.2	2.2
$N_2O_{indirect}$	**kg ha^{-1} a^{-1} N_2O-N**	**(0.1[d])**	**0.3**	**0.3**	**(0.1[d])**	**0.0**	**0.0**
accountable N_2O emission[e]	**kg ha^{-1} a^{-1} N_2O-N**		**1.6**	**1.5**		**0.8**	**0.7**

[a] non-volatile proportion: 98 % of total accountable N fertiliser; [b] Incorporation of intercrop and pioneer crop; [c] total amount of N of the grass-legume mix minus fixed N; [d] by NO from soil; [e] Emissions of the area under energy crops minus emissions from fallow; LCC: Life cycle comparison

Nitrous oxide (N_2O). Within the life cycles of energy crops, the agricultural area represents the dominant source of N_2O (Figure 4, top left). The individual emission sources from soil are differentiated by their causes and origin in Table 9. The calculated values for N_2O_{direct} agree well with field measurements (cf. /Bouwman 1996/). Evidently, nitrogen input by fertiliser and crop residues constitute the major reason for N_2O emissions from soil. The deposition of airborne nitrogenous compounds slightly adds to indirect N_2O release. Nitrous oxide is also emitted during the production of synthetic nitrogen fertiliser and from the combustion of biomass.

In contrast, N_2O is released in the life cycle of light heating oil at small amounts, dominantly during combustion.

As a consequence of the lower nitrogen input in the extensive options 2 and 4, the N_2O emissions stay below those of the conventional options 1 and 3 in virtually all steps in the life cycle. The clearest emission reduction results from the avoidance of energy- and process-borne N_2O emissions during the production of synthetic nitrogen fertiliser. In total, option 2 releases about half of N_2O as option 1 if related to the same amount of useful energy (Figure 4, bottom right). However it remains to be proven whether – as assumed here – the emission factor for fixed nitrogen compares to the one of mineral fertiliser because the actual amount of nitrogen fixed is difficult to quantify. Comparable statements also apply to the options 3 and 4.

To summarise, the N_2O balance indicates that the substitution of light heating oil for heat provision by whole cereal crops is associated with significantly higher N_2O emissions.

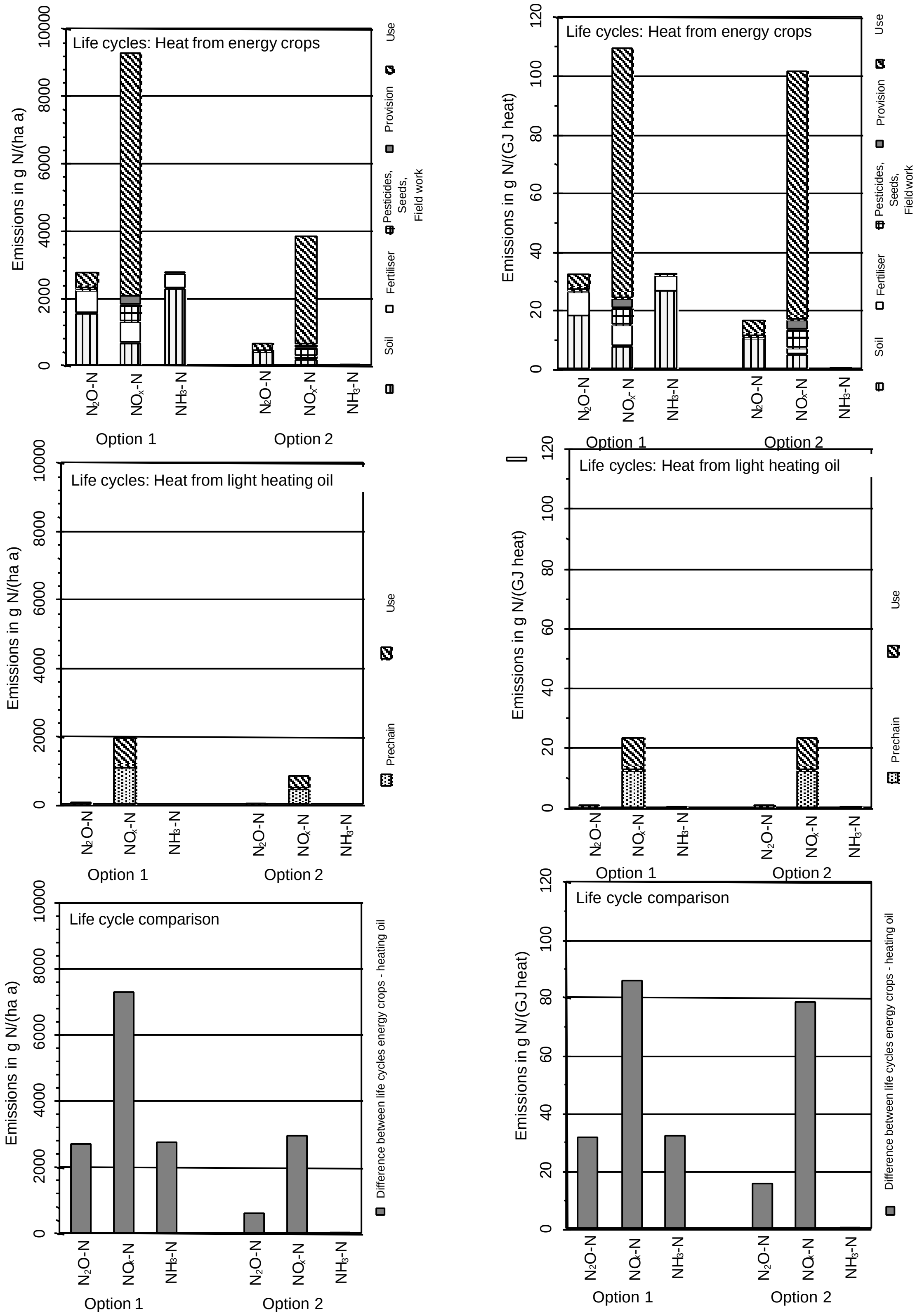

Figure 4 Inventory analysis of volatile nitrogen compounds in the life cycle of heat from

Triticale (top), light heating oil (middle) and in the life cycle comparison (bottom), related to the hectare of agricultural area (left) and useful heat at the gate of the heating plant (right)

Nitric oxides (NO_x). In the life cycle of heat from whole cereal crops, NO_x emissions originate dominantly from the combustion with only minor sources in soil, the production of synthetic nitrogen fertiliser and mechanical field work. In contrast, in the case of heat from light heating oil, NO_x emissions are related to the prechain and use in equal parts. Hence, the heating plant represents the place where the most significant additional NO_x emissions occur in the life cycle comparison. This is primarily induced by the high nitrogen content in the biomass, especially in the grains, which leads to the formation of about eight times more NO_x than the combustion of light heating oil.

The conventional and extensive options merely differ with regard to NO_x emissions because the balance is governed by the heating plant and for NO_x, prechains have a minor overall effect only.

To conclude, all options result in a considerably higher formation of NO_x when heat is provided with whole cereal crops as compared to heat from light heating oil.

Ammonia (NH_3). In the life cycle of the energy crops, NH_3 primarily volatilises from synthetic fertiliser applied to the agricultural land and results at smaller rates also from the production of synthetic fertilisers. In contrast, NH_3 is released in low amounts in the life cycle of light heating oil only. Consequently, the conventional options of bioenergy production (1 and 3) are associated with higher NH_3 emissions as opposed to heat provision from light heating oil.

The extensive options (2 and 4), however, lead to negligibly higher NH_3 emissions than the life cycles of light heating oil because NH_3 originates mainly from the synthetic fertilisers applied in the conventional production system, but not from biological nitrogen fixation by legumes.

Comparative discussion. The heat provision by whole cereal crops leads to higher emissions of NH_3, N_2O and NO_x as compared to a heat provision by light heating oil. The additional emissions originate largely from the life cycle steps of the crop production and use whereas the provision releases low amounts of nitrogenous compounds only.

The yields of winter rye (options 3 and 4) and consequently, the nitrogen input per hectare of agricultural land, are assumed to be lower than for Triticale (options 1 und 2; Table 7; Table 9). However, when compared on the basis of the same amount of useful energy at the gate of the heating plant, the results of the two crops are similar. The assumed yield losses in the extensive rye production system compensate the reduced NO_x emissions of the extensive versus the conventional cropping system if related to useful energy. The difference is only visible for Triticale for which the yield loss in the extensive system was assumed to be small.

3.3.2 Impact assessment

Subsequently, the impact categories stratospheric ozone depletion, anthropogenic contribution to global warming, eutrophication and acidification are considered separately without accounting for potential interactions (Figure 5).

Stratospheric ozone depletion. Related to the same amount of useful energy, the provision of heat by whole cereal crops substituting light heating oil results in increased N_2O emissions and hence an increased potential depletion of the stratosheric ozone layer. The increase is less pronounced if the crops are produced extensively (options 2 and 4) instead of conventionally (options 1 and 3). This finding is equivalent with the discussion of N_2O in the inventory analysis.

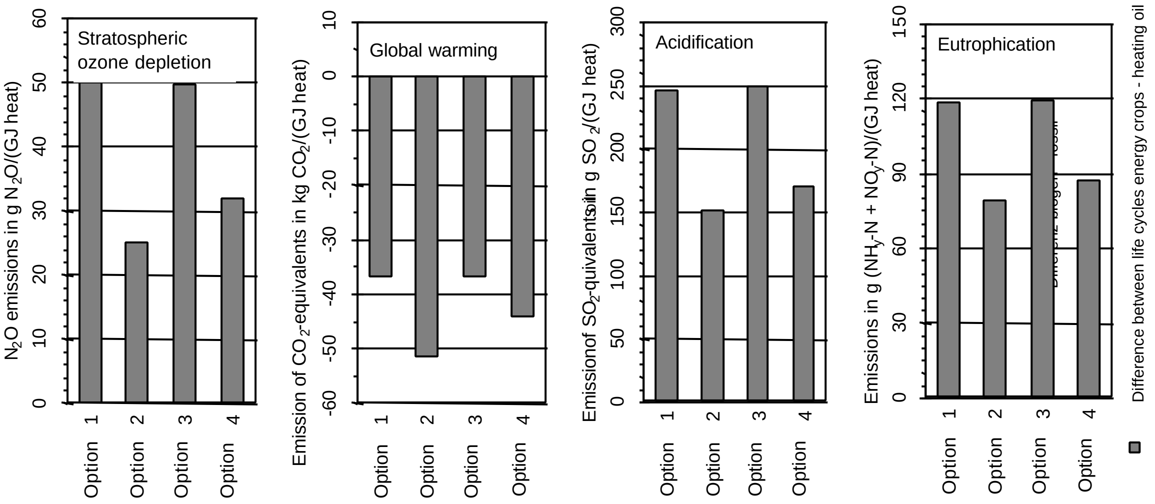

Figure 5 Comparison of impact categories

Anthropogenic contribution to global warming. In the life cycle of heat provided by energy crops, N_2O contributes close to two thirds and CO_2 from the utilisation of fossil fuels about one third to the gas emissions with direct climatic relevance. However, the direct greenhouse gases account for a third (options 1 and 3) and a quarter (options 2 and 4), respectively, of the overall release of CO_2-equivalents because NO_x emissions make the major contribution although their indirect global warming potential varies widely. If the band width given by /Lammel and Grassl 1995/ is considered, the ratio of indirectly active versus directly active greenhouse gas emissions ranges between 1:1 and 4:1. Methane is responsible for less than 2 % of the overall emissions of CO_2-equivalents, so the result is stable even if the indirect climatic effects of CH_4 /Brühl 1993/ are accounted for.

As opposed to the use of whole cereal crops, the provision of heat by light heating oil is associated with the release of almost twice as high emissions of CO_2-equivalents. Carbon dioxide produced in the heating plant represents the major compound.

The extensive options release slightly lower amounts of CO_2-equivalents than the conventional ones. This is particularly due to lower emissions of N_2O and NO_x as well as a higher energy use efficiency in the extensive crop production system.

Acidification and eutrophication. In the life cycle of heat provision via energy crops, considerably more NH_3 and particularly NO_x is released as compared to light heating oil, leading to potentially higher eutrophication. Accordingly, more acidifying gases are produced. The volatile nitrogen compounds account for 73 % the balance of SO_2-equivalents (of which 75 % as NO_x and 25 % as NH_3), whilst their share in the acidifying emissions in the life cycle of heat from light heating oil is 27 % only.

The trace gas emissions with acidifying and eutrophying impact in the conventional options exceed those of the extensive options, mainly due to the avoidance of NH_3 emissions from fertilisation in the extensive production scheme.

Comparative discussion. All environmental impact categories show a clearly lower emission level for the extensive crop production systems than for the conventional options. The provision of nitrogen to crops via biological nitrogen fixation by legumes reduces the emissions of N_2O, CH_4, CO_2 and in particular of NH_3 as compared to the application of synthetic fertiliser. However, these benefits in terms of reduced emissions are linked to a considerably higher demand for land area in the extensive crop production system.

3.3.3 Sensitivity analyses

The flows of energy and matter in technical processes of the prechains can be quantified at a relatively high degree of certainty. In contrast, the emissions of volatile nitrogen compounds from the heating plant depend on a number of factors which are more difficult to predict. This is particularly true for heating plants fired with biomass because few emission measurements have been performed so far. Therefore, adequate emission models have been established (cf. /Kaltschmitt and Reinhardt 1997/). Furthermore, the emissions N_2O and NO_x from soils can vary over a wide range (Table 8). Therefore, in the following, sensitivity analyses are used to investigate whether and to what degree the uncertainty and variability in the data affect the overall results. In order to address the most critical issues, the emissions of volatile nitrogen compounds are varied for the sources involving low to moderate certainty and in parallel, contributing significantly to the overall emissions (Table 10).

Table 10 Importance and certainty of the results of emissions of volatile nitrogen compounds in the individual steps of the life cycles

	Share of overall emissions in %			Certainty of data		
	N_2O	NO_x	NH_3	N_2O	NO_x	NH_3
Triticale, conventional						
Soil	56	7	82	**+**	**(+)**	++(+)
Fertiliser	25	7	17	++(+)	++(+)	++(+)
Pesticides	<1	1	<1	++	++	++
Seeds, Field work	1	4	<1	+++	+++	+++
Provision	<1	3	0	+++	+++	+++
Use	17	78	0	++	**++**	+(+)
Light heating oil						
Prechain	17	55	100	+++	+++	+++
Use	83	45	0	+++	+++	++(+)

Certainty of data: (+) very low; + low ; ++ moderate; +++ high

+ Variables for the parameter variation

Nitrous oxide emissions from soil. Due to the great uncertainty in the N_2O emissions from soil (impact categories: stratospheric ozone depletion and anthropogenic contribution to global warming) the effect of potential variation in the emission factors according to /IPCC 1997/ (Table 8) on the results of options 1 and 2 is analysed (Figure 6). As illustrated, the results discussed so far are qualitatively stable. In quantitative terms, however, the original average increase in N_2O emissions may vary by factor 1.7. The result of reduced emissions of CO_2-equivalents (impact category: anthropogenic contribution to global warming) is less affected, changing between –30 and +20 % around the original mean result only.

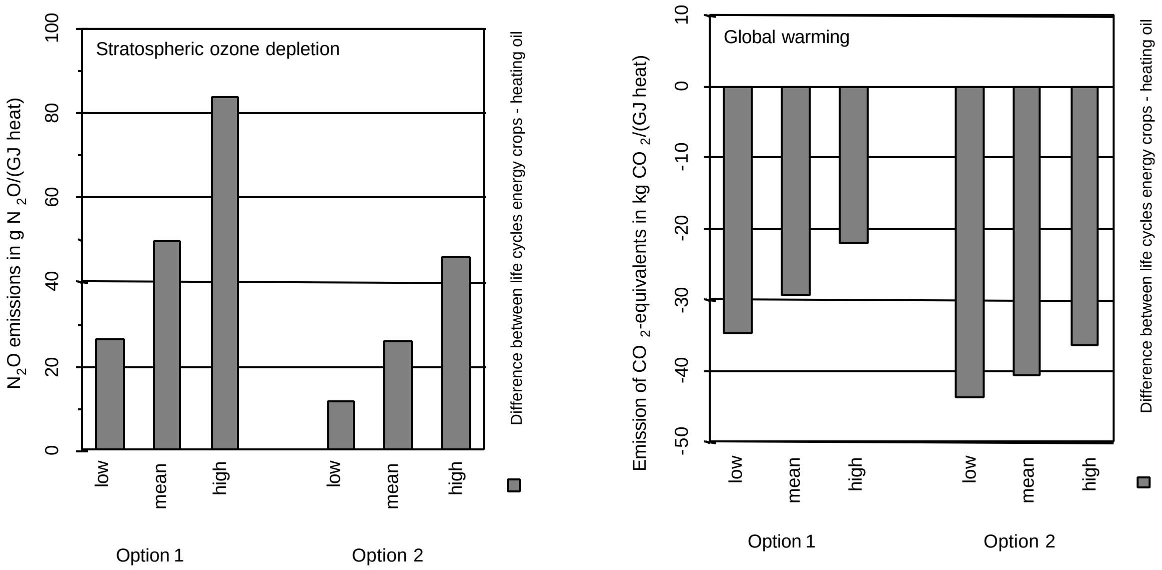

Figure 6 Variation of emission factors for N_2O from soil: Impact on the results of the life cycle comparison of options 1 and 2; left: impact category stratospheric ozone depletion; right: impact category anthropogenic contribution to global warming

Nitric oxide emissions from soil. If for the calculation of biogenic NO emissions from soil a $NO\text{-}N/N_2O\text{-}N_{direkt}$ ratio of 0 (i.e. no NO emission from soil; Figure 7 left, low) and 5 respectively (Figure 7 left, high) is assumed instead of a mean ratio of $NO\text{-}N/N_2O\text{-}N_{direct}$ of 0.5, the results of the NO_x balance of option 1 changes by –10 and +80 % respectively. In quantitative terms, this imposes an uncertainty of –7 and +60 % to the impact category eutrophication or –6 and +23 % respectively to the impact category acidification. However, the results remain qualitatively stable.

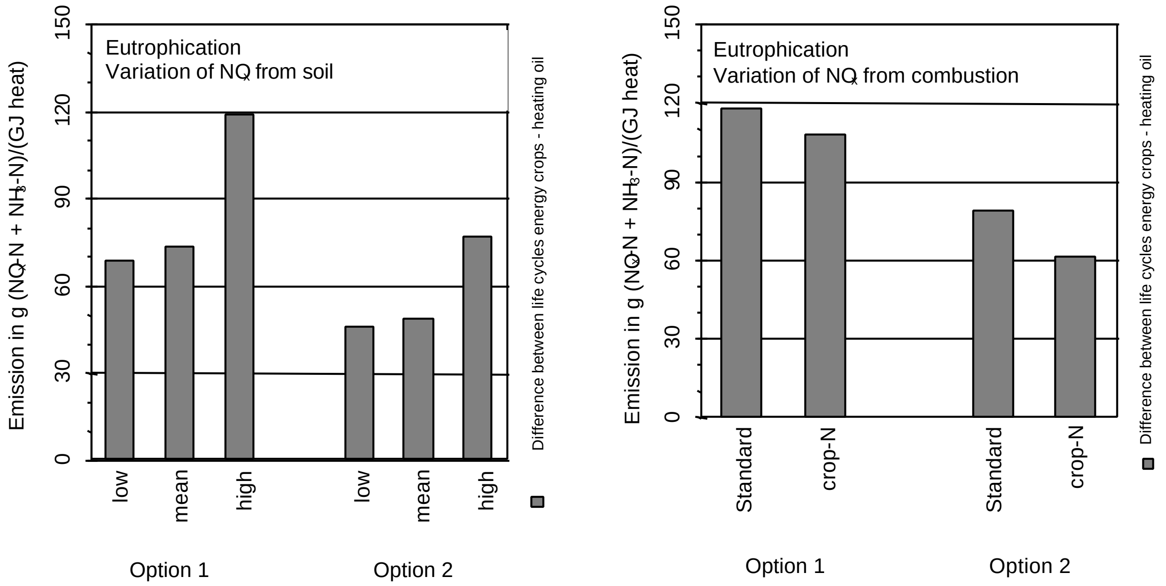

Figure 7 Variation of NO_x emissions and their impact on the results of the life cycle comparisons of options 1 and 2 in the impact category eutrophication. Left: variation of NO_x from soil; right: variation of NO_x from combustion of energy crops

Nitric oxide emissions from the combustion of whole cereal crops. The nitrogen content in Triticale exceeds the one in winter rye (Table 7). This may produce accordingly higher NO emissions from the combustion in the heating plant. So far, however, emissions have been measured for whole winter wheat crops only and in addition, the formation of N_2O and NO_x also depends on parameters like the boiler temperature and water content of the biofuel, which may overcompensate the effect of differing fuel nitrogen content. Therefore, the standard options rely on the same NO and N_2O emission factors for combustion of biofuels. This assumption is justified for N_2O emissions since in the entire life cycle, the use of the crops contributes in minor terms to the overall N_2O release and furthermore, the formation of N_2O is related to the fuel nitrogen content to a limited extent only. For the standard options, the estimation of emissions of volatile nitrogen compounds from combustion is based on a nitrogen content of 1.4 % in the whole cereal crops in accordance with winter wheat crops (Figure 7 right, standard). In order to evaluate the impact of a changing fuel nitrogen content on NO_x emissions from combustion, an approximately linear relationship between the nitrogen content of whole cereal crops and NO_x emissions /Nussbaumer 1989/ is postulated.

As illustrated in Figure 7 (right, crop-N), the overall results of the life cycle comparisons are consequently reduced by 10 % (option 1) and 20 % (option 2) as compared to the standard options.

3.4 Conclusions and outlook

Biomass is commonly seen as a promising option to meet the future energy demand, particularly with regard to the concomitant reduction of energy-borne negative effects on environment and climate. Against this background, this study aimed at the quantification of emissions of volatile nitrogen compounds in the entire life cycle of heat provided by whole cereal crops as compared with heat from heating oil. As an example, the life cycles of heat provision by whole crops of winter rye and Triticale produced in conventional and extensive systems, respectively, were compared with the life cycle of heat from light heating oil. The results gathered can be summarised as follows:

- Whole cereal crops used as biofuels to produce heat can considerably reduce the emission of greenhouse gases if they substitute light heating oil. This benefit results dominantly from the avoidance of CO_2 emissions from fossil fuels because the CO_2 released from the combustion of biofuels does not contribute to human-induced global warming if the crops are produced in a sustainable manner. In contrast, the life cycle of whole cereal crops is associated with considerably higher emissions of the climatically relevant trace gases N_2O and NO_x which do, however, not compensate for the benefit of reduced fossil CO_2 emissions although their specific global warming potentials exceed the one of CO_2.
- The life cycles of whole cereal crops show significant release of volatile nitrogen compounds. The gases have particular main sources. Nitrous oxide originates dominantly from microbial processes in soil, NH_3 from the production and application of synthetic fertiliser, and NO_x from the combustion of the biofuels. Therefore, the heat provision by whole cereal crops substituting light heating oil yields higher potential environmental impacts in the categories stratospheric ozone depletion, eutrophication and acidification. Despite the fact that the emissions of N_2O and NO_x from soils are prone to considerable uncertainty, these statements remain true if the input parameters in the life cycle comparison vary within the presently known band widths.
- In the life cycle comparison of the extensive options to produce whole cereal crops by using biological nitrogen fixation instead of synthetic fertilisation, N_2O emissions are reduced by up to 50 % and NH_3 emissions by close to 100 % as opposed to the conventional production options. However, due to the lower yields per hectare and the additional demand for land to grow legumes for nitrogen fixation as a pioneer crop to the energy crops, the provision of the same useful energy at the gate of the heating plant requires a larger area of agricultural land. Hence, the extensive options show clear advantages against the conventional ones with regard to the impact categories of

stratospheric ozone depletion, eutrophication and acidification, which have to be realised by a greater demand for agricultural area, which exceeds the one in the conventional option by factor two (Triticale) and three (winter rye), respectively, depending on yields.

To summarise, the use of whole cereal crops substituting fossil fuels for heat provision leads to a reduction of the anthropogenic contribution to global warming, but to a higher release of trace gases with negative impact in terms of statospheric ozone depletion, eutrophication and acidification. It must be kept in mind, however, that the technical development of heating plants fired with whole cereal crops has only recently started, so the plants have not yet been optimised with regard to the emissions of volatile nitrogen compounds. Furthermore, there is a number of opportunities in crop production to reduce nitrogen losses. In case the emissions from these two major sources will be reduced in the future it is to be expected that the provision of energy from whole cereal crops will also yield better scores in the latter impact categories. Therefore, identifying appropriate measures and developing implementation strategies should be seen as priority targets if biomass from annual crops shall make a significantly greater contribution to a future climate and environmentally friendly energy provision in Germany.

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4 Controls and Models for Estimating Direct Nitrous Oxide Emissions from Temperate and Sub-Boreal Agricultural Mineral Soils in Europe[2]

Abstract

Based on a review of N_2O *field studies in Europe, major soil, climate and management controls of* N_2O *release from agricultural mineral soils in the European Union have been identified. Data for these* N_2O *emission drivers can easily be gathered from statistical services. Using stepwise multivariate linear regression analysis, empirical first order models of* N_2O *emissions have been established which allow – in contrast to existing large-scale approaches – a regionally disaggregated estimation of* N_2O *emissions at sub-national, national and continental level in the temperate and boreal climate regions of Europe. Arable soils showed lower mean and maximum emissions in oceanic temperate climate ("Temperate West") than in pre-alpine temperate and sub-boreal climate ("Sub-boreal Europe"). Therefore, two separate regression models were developed. Nitrous oxide emissions from arable soils the Temperate West amount to an average flux rate below 2 kg* N_2O*-N* ha^{-1} yr^{-1} *and rarely exceed 5 kg* N_2O*-N* ha^{-1} yr^{-1}*. They are modelled by the parameters fertiliser, topsoil organic carbon and sand content. In Sub-boreal European arable soils,* N_2O *emissions vary in a much wider range between 0 and 27 kg* N_2O*-N* ha^{-1} yr^{-1} *in dependence of available nitrogen, represented in the model by fertiliser and topsoil nitrogen content. Compared to existing methods for large scale inventories, the regression models allow a better regional fit to measured values since they integrate additional driving forces for* N_2O *emissions. For grasslands, a fertiliser-based model was established which yields higher emission estimates than existing ones. Due to an extreme variability, no climate, soil nor management parameters could be included in the empirical grasslands model.*

4.1 Introduction

Nitrous oxide is a trace gas emitted along with chemical and biological processes. It participates in stratospheric ozone depletion as well as in the greenhouse effect and has a 296

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times higher global warming potential than CO_2 /Houghton et al., 2001/. Therefore it belongs to the basket of trace gases addressed within the Kyoto Protocol /UNFCCC 1997/. First incomplete investigations showed that in 1999, agriculture accounted for approximately half of the anthropogenic N_2O emissions in the European Union, with agricultural soils dominating the sources /EEA 2001/.

4.1.1 Controls of soilborne N_2O emissions

Denitrification and nitrification have been identified as the principal processes of N_2O production in soils (e. g. /Bremner 1997/). Whilst these microbial processes and their controls are well understood the tremendous temporal and spatial variability of N_2O emission rates poses an unresolved challenge to modelling, monitoring and prediction. Nitrous oxide emissions from soils arise from low, relatively constant, continuous emissions and – generally more important – from short emission peaks commonly associated with denitrification /Firestone and Davidson 1989/ or both denitrification and nitrification. The ultimate drivers of N_2O emissions act at a proximal scale but are highly interlinked with soil conditions and management expanding over local to regional dimensions.

Proximal soil factors drive the microbial processes of N_2O production and consumption at the micro scale (10^{-3} m). Substrate availability, oxygen availability, soil moisture and soil temperature directly control the hourly and diurnal variation of N_2O fluxes (Figure 8). In fertilised soils, the denitrification rate is most commonly limited by oxygen supply and secondly by the availability of organic carbon as a reductant and by nitrate availability /Tiedje 1988/. Furthermore the N_2O/N_2 ratio produced during denitrification increases with an excess of oxidants (nitrate, nitrite) over reductants, at low temperatures, and whenever a factor reduces the rate of overall reduction in soil /Betlach and Tiedje 1981/. These proximal physical and chemical factors are themselves controlled by biological drivers supplying substrate such as readily degradable organic carbon and by the oxygen demand of decomposing organisms and roots. In general, N_2O production increases along with increasing rates of nitrification and denitrification and N_2O/N_2 product ratios. However, the proximal drivers interact with each other and also depend on larger scale factors. Oxygen availability, for instance, is driven by O_2 consumption through microbial and root activity as well as by diffusion constraints through soil structure and soil water content. The latter is again determined by the water balance as a function of local factors such as precipitation, interflow, drainage rate and evapotranspiration, which again depend on climate, position in the landscape, soil texture, crop type, season etc. (Figure 8).

Local factors (1-10^3 m) govern N_2O emission rates on a daily or weekly time scale. Farm management contributes to the emissions, but the driving forces of the microbial activity intimately link also to climate, weather, site properties /Smith et al. 1998; Skiba and Smith 2000/ and land use history /Mosier et al. 1998b/.

Soil. Many studies have documented the particular importance for N_2O emissions of elevated levels of soil moisture and soil nitrate concentration and hence of conditions favourable for denitrification /Yamulki et al. 1995, Ambus and Christensen 1995, Clayton et al. 1997, Skiba et al. 1994/. Among the soil chemical characteristics affect N_2O formation, transport, and emission /Hutchinson and Davidson 1993, Granli and Bøckman 1994/, nitrogen availability measured as ammonium and nitrate concentrations in topsoil were shown to correlate with N_2O emissions /Kaiser et al. 1996, Skiba et al. 1998, Smith et al. 1998/. Soil texture and clay content /Kaiser et al., 1996/, respectively, as well as drainage status have been proven as useful proxies for O_2 availability /Hutchinson and Davidson 1993, Granli and Bøckman 1994/. The effect of soil moisture changes on N_2O depends on the state of water-filled pore space (WFPS). In dry soils, a rising WFPS will lead to an increase in N_2O emission by enhanced denitrification, but above 80-90 % WFPS, N_2O release declines due to a sharp decrease of the N_2O/N_2 ratio /Linn and Doran 1984/. In the case of WFPS, positive as well as negative interactions with N_2O emissions may occur. Such ambiguity greatly restricts a simple generalisation of the influence of many parameters on N_2O release (Figure 8). Both the direction of the influence as well as the importance of a parameter vary in a complex pattern in space and time.

Management. The N_2O production conditions differ in arable land from those in grassland. Perennials have a longer growing season and therefore a prolonged nitrogen uptake as compared to annual summer crops and no period of bare soil without N uptake by crops. This helps to avoid the accumulation of mineral nitrogen in soil as it may happen after harvest of annual cultures in autumn. Grassland soils tend to accumulate more available carbon in the topsoil layer than arable soils /WBGU 1998/, which in turn tend to have smaller C/N ratios as a result of carbon depletion and intensive fertilisation /Tisdall and Oades 1982/. Furthermore, in the temperate and boreal regions of Europe, grasslands tend to cover the temporarily wet or poorer soils or higher elevations /EEA 1995/. To account for these different patterns, arable lands and grasslands need to be addressed separately. Culture-dependent differences in N_2O emissions from arable soils arise from the temporal and spatial adaptation of nitrogen fertilisation to crop demands, i.e. preferential application in spring or summer, soil compaction by tractor traffic, and the amount, the C/N ratio and timing of crop residues /Flessa et al. 1998/. The decay of nitrogen-rich crop residues (like from rapeseed or potato) may lead to high post-harvest emissions /Flessa et al. 1998, Flessa and Beese 1995, Smith et al. 1998/. Similarly, in root and tuber cultures, soil compaction by frequent traffic and fertiliser spread between the crop rows may increase the N_2O release during the cropping season /Kaiser et al. 1998, Ruser et al. 1996/. Hence, in accordance with /Smith et al. 1998/, lower annual N_2O emissions are expected for soils cultivated with cereals than with oilseeds or root and tuber crops.

Nitrogen input is a principal control of N_2O emissions. The effect of nitrogen fertiliser application has been excessively reviewed in the past /Bouwman 1996, Eichner 1990/. The annual amount of nitrogen input to a field through fertilisation, nitrogen fixation and crop residues is being recommended world-wide to estimate direct N_2O emissions from agricultural soils in national inventories /IPCC 1997/. In general, an increase of N input will increase both the nitrification and denitrification rates as well as the N_2O/N_2 ratio.

Manure as well as combined manure and synthetic fertilisers may lead to higher N_2O emissions directly after application than synthetic fertilisers alone /Bouwman 1996, Clayton et al. 1997, Kaiser et al. 1996/. In contrast to /Eichner 1990/, in recent studies, the various forms of synthetic nitrogen fertiliser commonly applied in Europe resulted in similar emission factors /Bouwman 1996, Hénault et al. 1998, Eichner 1990, Michel and Wozniak 1998/. In the latter studies, emission factors depend more on the soil /Hénault et al. 1998/ and weather conditions /Flessa et al. 1995/ during and after application or both /Clayton et al. 1997, Smith et al. 1998/ rather than on fertiliser type alone.

Climatic features affect N_2O emissions at the **regional to continental scale** by setting a general framework for average, maximum and minimum precipitation, air temperatures and temperature changes (Figure 8).

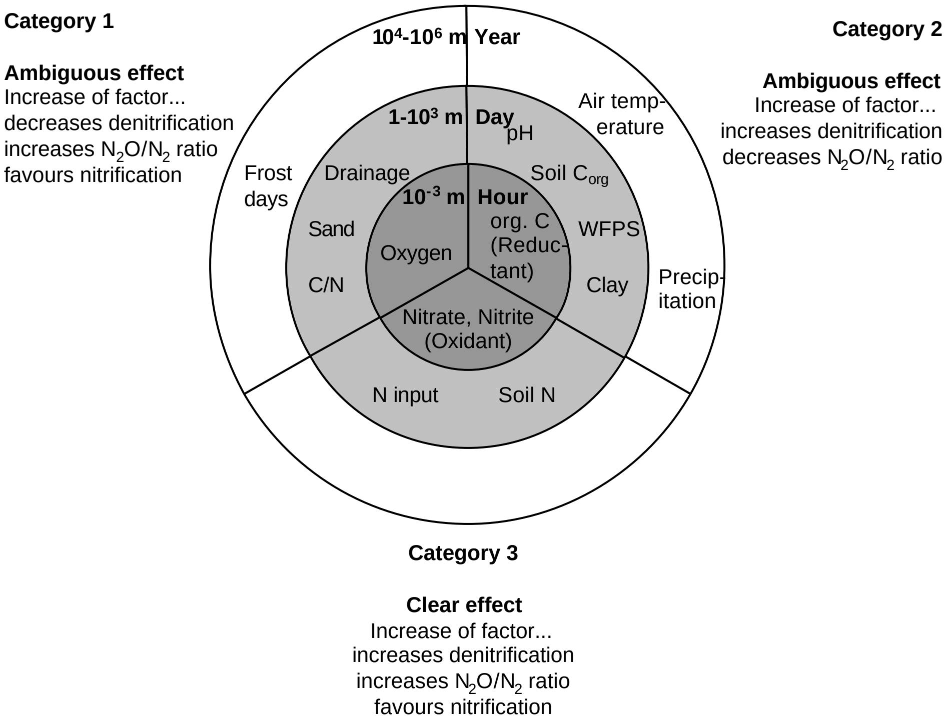

Figure 8 Scheme of factors regulating denitrification in agricultural soil; WFPS: water-filled pore space

Extreme events produce large portions of the annual N_2O emissions within a few days only. They typically occur either in winter during freeze-thaw cycles or in summer through rewetting of dry soil, driven by climate and weather parameters as well as in situations with elevated mineral nitrogen concentrations in moist soils after fertilisation or during the decomposition of crop residues, controlled by management and its interaction with climate, soil and site parameters. The debate about the physical and biochemical mechanisms of freeze-thaw cycles is still ongoing. Field experiments suggest that the N_2O is of microbial origin rather than from chemodenitrification /Röver et al. 1998/. Extreme N_2O emission events occurring after a period of significant soil freezing and/or snow cover have been documented for a variety of climatic conditions throughout Europe (Denmark: /Christensen and Tiedje 1990/, France: /Hénault et al. 1998/, Northern Germany: /Ernst 1997, Heinemeyer et al. 1996, Kaiser and Heinemeyer 1996, Röver et al. 1998/ and Southern Germany: /Flessa et al. 1995, Ruser 1999/). In contrast, /Armstrong 1983/ and /Yamulki et al. 1995/ reported no emission peak after light freezing of a well-drained surface soil for a week.

Emissions during the 3 to 4 winter months have been reported in the range of 7 to 92 % of the annual N_2O release with a mean of 48 % (± 19 %), exceeding in average the emissions during the growing season. In dry continental conditions (USA and Canada: /Chang et al. 1998, Kessavalou et al. 1998/) or in well-drained soils under climate without severe frost (UK: /Armstrong 1983, Yamulki et al. 1995/), monthly N_2O emissions in winter do not exceed 30 % of the average monthly emissions in the rest of the year. According to the studies cited above, elsewhere in Europe, monthly N_2O emissions in winter are similar to those in other months (mean N_2O emission in winter months is 120 % the one in other months, median: 86 %, range 50-312 %).

4.1.2 Quantification of soilborne N_2O emissions

World wide activities are ongoing to quantify the various sources of N_2O. Available inventories of direct N_2O release from agricultural soils were so far based upon the amount of nitrogen added to soil via fertilisation, atmospheric deposition and crop residues or upon land use classes (e.g. /Kroeze 1994, Bouwman 1996, Mosier et al. 1996, IPCC 1997, Mosier et al. 1998a/). National inventories of direct N_2O emissions from agricultural mineral soils most commonly apply the approach (1) by /Bouwman 1996/, which aims to capture the entire N_2O flux, or (2) by /IPCC 1997/, which addresses the human-induced portion of the N_2O flux only:

$$E_{N2O} = 1 + 0.0125 \cdot N_{fert} \tag{1}$$

$$E_{N2O} = 0.0125 \cdot (N_{SN} + N_{AW} + N_{BN} + N_{CR}) \tag{2}$$

With E_{N2O} Emission of N_2O [kg N_2O-N ha^{-1} yr^{-1}]

N_{fert} N input by synthetic fertiliser and manure [kg N ha^{-1} yr^{-1}]

N_{SN} N input by synthetic fertiliser [kg N ha^{-1} yr^{-1}]

N_{AW} N input by manure [kg N ha^{-1} yr^{-1}]

N_{BN} N input by biological N fixation [kg N ha^{-1} yr^{-1}]
N_{CR} N input by crop residues [kg N ha^{-1} yr^{-1}]

However, according to the current state of knowledge outlined above, various additional controlling factors should be considered to improve the accuracy of such inventories (Mosier, et al., 1996). Also, in order to reduce the heterogeneity of the N_2O data, CORINAIR (EEA, 2000) recommends to define different agro-ecological zones taking account of the varying climatic conditions within a country. Agriculture releases N_2O directly from soils and manure but also indirectly through nitrogen losses as ammonia or nitrate, producing N_2O elsewhere in affected ecosystems (Mosier et al., 1998a, Groffmann et al., 1998, Nevison et al., 2000). The lack of data restricts the reliability of estimates of indirect agricultural N_2O emissions (Nevison et al., 2000), but a wealth of European measurement data allows to improve regional estimates of direct agricultural soilborne N_2O emissions.

Consequently, this paper aims to analyse the influence of climatic, soil and management parameters on direct annual N_2O emissions from agricultural soils in the European Union and to develop a detailed methodology for inventories of direct N_2O emissions from agricultural soils applicable at sub-national, national and continental levels.

Since the proximal controls of N_2O vary too much for large scale flux estimates by inventories, the use of proxy variables acting at the local, regional or continental scale is more practicable (Figure 8). Here we quantify the influence of various proxy climate, soil and management parameters on the magnitude of annual N_2O emissions from agricultural soils in the European Union. These proxies shall be commonly reported in the literature and be easily accessible at statistical services or derived from soil maps in order to serve as a feasible and transparent basis for inventory calculations.

4.2 Material and Methods

The literature was reviewed for European field data of N_2O fluxes and additional site information. The study focuses on cultivated mineral soils in order to reduce the heterogeneity of N_2O flux controls. This restriction to mineral soils is justified since N_2O fluxes from cultivated organic soils are driven by “fossil” carbon and nitrogen from peat oxidation after drainage and mechanical soil disturbance rather than by recently added substrates /Klemedtsson et al. 1997/. The investigation extends over the European temperate and boreal regions only because the two available studies in Mediterranean climate /Arcara et al. 1990, Teira-Esmatges et al. 1998/ span over less than five months only.

4.2.1 Nitrous oxide and site data

Data from field experiments based on in-situ measurements with micrometeorological and chamber techniques were admitted since these methods agree reasonably in field intercomparisons /Christensen et al. 1996, Smith et al. 1994/. Actually, most of the used data have been measured discontinuously by closed chamber methods with GC-ECD analysis.

Temporal representation

The discontinuity and limited duration of the field studies produces an intrinsic uncertainty in the data through the interpolation and extrapolation of the measured emission rates. At a Southern German site, for instance, /Ruser 1999/ calculated an average overestimation of the N_2O flux by 26 % by weekly measurements at noon in relation to continuous sampling. The N_2O emissions underlying this study were typically measured in weekly intervals. The uncertainty in the literature data through interpolation can *post hoc* not be quantified, but needs to be kept in mind during modelling and the interpretation of models. We identified 163 (52 sites) annual records in Europe which in many cases, however, sample winter data at intervals of several weeks only and incompletely report site and management conditions. These investigations thus provide only rough estimates of the total annual N_2O emissions as do extrapolations from shorter, but more intensively sampled measurement periods. Therefore, field studies with a measurement period of more than five months were considered. The number of data sets expanded to 256 (69 sites) records with an average measurement period of 10 to 11 months, greatly improving the information about soil properties and management. Emissions reported for periods shorter than one year were proportionally extrapolated to annual estimates, applying the findings above that average monthly N_2O release in winter compare to those in other months in most of Europe. This strategy minimises the inevitable error connected with the temporal extrapolation of the reported data.

Further sources of uncertainty in the temporal dimension lie inherently in the reported data, like measurement errors, basically errors due to sampling design or leakiness of a chamber. The available N_2O data always represent a logistically constrained compromise between addressing temporal and spatial variability. Given the uncertainty in annual N_2O emission rates an accurate assessment of the magnitude of emissions on large scale remains difficult and demands for a careful interpretation of the gathered results.

Spatial representation

The spatial variability of the measurements indicated in the literature ranges from 15 to 350 %, averaging around 70 % for 3 to 8 replicates. Evidently, micro scale variability in N_2O fluxes and their controls generates a wide scatter in the data which models based on local and regional site characteristics cannot explain. The database is biased in space since studies from Germany are over-represented. The number of annual N_2O data per site also varies between 1 and 15. Furthermore, measurements are concentrated around research centres in Europe so their spatial distribution is uneven and not random. This underlines the necessity to include site parameters in a sound generalisation towards regions not yet covered by measurements.

The distribution of arable crops in the studies analysed here corresponds well to the distribution of their cultivated areas in the EU 15 except for leguminous crops being undersampled.

Occasionally, studies do not give full information on climate, soil and management. In particular, soil chemical and soil physical parameters are reported in about half of the studies only. Some gaps could be filled by personal communications with the authors as well as by soil and climate maps.

4.2.2 Statistical analyses

In order to derive empirical models of soilborne N_2O emissions in the European Union, the collected data sets were statistically analysed as follows:

1) Homogeneous climate and land use groups were formed by a hierarchical cluster analysis with N_2O emissions, land use type (arable land or grassland), and frost class (Table 11) as sorting variables, taking account of the recommendation by /EEA 2000/ to separate various agroecological zones if necessary.

2) A set of potential controlling factors was selected based on the analysis above (Table 11) for which data are frequently reported in the literature and, for the purpose of inventories, information can be commonly gathered from national statistics and maps. The correlation with N_2O emissions and intercorrelations among the variables were tested with a bivariate non-parametric rank correlation analysis, using Spearman´s Rho.

3) Then the quantitative relation between annual N_2O emissions and each of the variables was determined by univariate linear regression analysis.

4) Finally, the dependence of annual N_2O emissions from combinations of variables was tested by multiple linear regression analysis with stepwise selection of independent variables /Norusis 1993/. Here the analysis was restricted to complete data sets. The remaining data sets with information gaps in some of the controlling factors were used for validation of the multivariate models.

All steps were applied to all data sets as well as separately to each homogeneous group of data sets. The regression analyses were performed with weighted data sets in order to assure an equal representation of all sites. A site was defined by the same climatic and soil properties, but eventually differing measurement years and fertilisation rates. Weighing the data sets by the inverse number of data sets per site avoided to optimise the models towards the conditions at the few excessively studied sites (Braunschweig/Germany, Edinburgh/UK, Göttingen/Germany, Longchamp/France, and Munich/Germany) but rather optimised the models to fit best to many sites. Tests with randomly chosen subsets of sites proved that the weighted regression analyses produced stable correlations and coefficients in contrast to nonweighted regression runs where in turn the regression coefficients depended on the choice of sites in the test regression runs.

The coefficient of determination R^2 of the regression model and the regression coefficients r^2 of the respective variables describe the degree of linear association /Neter et al. 1996/. The reliability of a model or parameter is given as its significance, tested by either F- or t-tests (* = $p < 0.05$; ** = $p < 0.01$; *** = $p < 0.001$). Correlation and regression analysis filters out those parameters interacting with N_2O emissions in a relatively uniform and homogeneous pattern over a wide range of environmental conditions. Parameters of which the impact and importance varies in space and time cannot be considered due to non-linear behaviour. The linear approach used in this study drastically simplifies the real situation but allows to estimate the annual N_2O flux when detailed local site data are unavailable. All statistical tests were performed with the software SPSS 9.0 for Windows.

Table 11 Potential controlling factors to describe N_2O emissions

Category*	Parameter	Unit
Climate		
2	Annual precipitation	mm
2	Mean air temperature	°C
1	Frost class	classes: 0 = no significant frost; 1 = periodical frost with shallow snow cover; 2 = regular frost with high snow cover
Soil		
2	Drainage class	classes: 0 = well drained; 1 = moderately; 2 = poorly; 3 = very poorly
2	Texture	% clay
1		% sand
?		% silt
2	Carbon content of topsoil	%
3	Nitrogen content of topsoil	%
1	C/N ratio	1
2	pH	1
Management		
?	Crop type[a]	classes: 0 = fallow; 1 = cereals; 2 = oilseeds; 3 = roots/tubers; 4 = legumes
3	Total N-fertiliser applied	kg N ha^{-1} yr^{-1}
?	Fertiliser type	classes: 1 = synthetic; 2 = organic; 3 = synthetic + organic 11 = synthetic nitrate; 12 = synthetic ammonium
?	Application mode	classes: 1 = split application; 2 = all at once

* parameter category in Figure 8
[a] applies to arable soils only

5) The regression models were validated with independent test sites. The uncertainty in the predicted N_2O emissions exceeds the mean standard error of the regression /Neter et al. 1996/. The prediction uncertainty for the validation test sites was quantified by Bonferroni simultaneous prediction limits /Neter et al. 1996/ using the software Mathematica 3.0. In analogy to the procedure described by /Fonseca and Parresol 2001/, the prediction interval was obtained as

$$N_2O_{pred} \pm se_{pred} \cdot B \qquad (3)$$

where N_2O_{pred} is the predicted emission in kg N_2O-N ha^{-1} yr^{-1}, se_{pred} the standard error of the prediction /Neter et al. 1996, Fonseca and Parresol 2001/ and B the Bonferroni value for simultaneous prediction limits:

$$B = t(1\text{-}p/2;\ n\text{-}a) \qquad (4)$$

was derived from tables of the *t*-distribution (Neter et al., 1996) for a given Type I error p, for n regression observations and for a parameters in the regression equation.

4.3 Results

4.3.1 Homogeneous climatic groups

The cluster analysis sorted the data sets into classes according basically to annual N_2O emission levels. When attributing sites to the classes, the first class contained most arable temperate sites with 90 % of annual mean emissions below 3 kg N_2O-N ha^{-1} yr^{-1} and maximum emission rates below 10 kg N_2O-N ha^{-1} yr^{-1}. Some Southern German sites bearing extended snow cover in winter fell into joint classes with the sub-boreal sites achieving in general mean annual emissions above 3 kg N_2O-N ha^{-1} yr^{-1} and maximum rates above 10 kg N_2O-N ha^{-1} yr^{-1}. Therefore, the arable sites were separated into two distinct climatic groups: 1) Temperate Western Europe (all temperate EU except alpine and extensively snow-covered pre-alpine regions, “Temperate West”) and 2) Temperate-moist-subcontinental (e.g. Germany South of 49°N), sub-alpine and sub-boreal regions (“Sub-boreal”). In contrast, the classification of grassland sites did not follow a spatial pattern but rather soil and management properties. The grassland sites all joined low to moderate emission classes with few exceptional data sets on poorly drained or excessively fertilised (>500 kg N input per year) sites. This allows to choose a uniform climatic group for European grassland soils (Table 12).

Table 12 Annual N_2O emissions in European homogeneous climatic groups

Region	Number of data sets (sites)	Mean	Median	Range
		---------[kg N_2O -N ha^{-1} yr^{-1}]----------		
Arable soils				
Temperate West	91 (27)	1.8	1.5	0.0 – 8.0
Sub-boreal	67 (13)	6.5	5.3	0.0 – 27
Total	158 (40)	3.6	2.0	0.0 – 27
Grassland soils				
Temperate and Sub-boreal Total	64 (29)	3.6	2.3	0.0 - 21

4.3.2 Importance of controlling factors

Univariate regression analyses identified a set of factors out of those of Table 11 with a significant quantitative linear statistical relationship with annual N_2O flux rates (Table 13). The other parameters did not reveal statistically significant correlations.

Many of the parameters tested (Table 11) are intercorrelated. So the emission factors given in Table 13 should be interpreted with some precaution since the statistical significance does not indicate any causal relationship. A given parameter represents a set of typical site conditions which must always be kept in mind. In general, the parameters displayed in Table 13 show highly significant, but relative low regression coefficients due to the facts that the data sets are widely scattered and some important sources of uncertainty like the interannual variability and microscale spatial variability in N_2O flux rates have been disregarded as separate factors in the analyses.

Table 13 Significant emission factors based on univariate regression analysis

Parameter	Emission factors	Descriptives (N° of data sets)	Significant ($p < 0.05$) intercorrelations with
All arable and grassland soils			
Drainage class	1.7 ± 0.5 kg N_2O-N (class number)$^{-1}$	$r^2 = 0.05$*** (212)	Soil C, soil N, fertiliser, clay, silt, sand
Clay	6.2 ± 3.2 kg N_2O-N %$^{-1}$ clay	$r^2 = 0.05$** (130)	Drainage class, silt, sand, soil N, soil C/N, soil pH
Soil C	0.64 ± 0.25 kg N_2O-N %$^{-1}$ soil C	$r^2 = 0.03$* (219)	Drainage class, silt, sand, soil N, soil C/N, soil pH, fertiliser
Fertiliser	0.014 ± 0.002 kg N_2O-N kg^{-1} N-input	$r^2 = 0.15$*** (242)	Drainage class, soil C, fertiliser type, crop type
All arable soils			
Clay	0.17 ± 0.5 kg N_2O-N %$^{-1}$ clay	$r^2 = 0.08$** (107)	Drainage class, silt, sand, soil N, soil C/N, soil pH
Silt	0.065 ± 0.025 kg N_2O-N %$^{-1}$ silt	$r^2 = 0.06$** (107)	Drainage class, clay, sand, soil C, soil C/N, soil pH, fertiliser type
Sand	-0.065 ± 0.019 kg N_2O-N %$^{-1}$ sand	$r^2 = 0.10$*** (107)	Drainage class, clay, silt, soil C, soil C/N, soil pH
Soil N	27 ± 10 kg N_2O-N %$^{-1}$ soil N	$r^2 = 0.06$** (132)	Clay, soil C, C/N, soil pH, crop type
Soil C/N	-0.38 ± 0.17 kg N_2O-N (C/N)$^{-1}$	$r^2 = 0.04$* (130)	Clay, silt, sand, soil C, soil N, soil pH
Fertiliser	0.013 ± 0.006 kg N_2O-N kg^{-1} N-input	$r^2 = 0.03$* (169)	Fertiliser type, crop type
Fertiliser type	3.5 ± 0.7 kg N_2O-N (fertiliser class)$^{-1}$	$r^2 = 0.15$*** (130)	Drainage class, silt, fertiliser
Arable soils Temperate West			
Clay	0.075 ± 0.019 kg N_2O-N %$^{-1}$ clay	$r^2 = 0.20$*** (65)	Drainage class, silt, sand, soil N, soil C/N, soil pH
Silt	0.019 ± 0.010 kg N_2O-N %$^{-1}$ silt	$r^2 = 0.05$* (65)	Drainage class, clay, sand, soil C, soil C/N, fertiliser type
Sand	-0. 025 ± 0.007 kg N_2O-N %$^{-1}$ sand	$r^2 = 0.17$*** (65)	Drainage class, clay, silt, soil N, soil C/N, soil pH
Soil C	0.65 ± 0.13 kg N_2O-N %$^{-1}$ soil C	$r^2 = 0.21$*** (100)	Silt, soil N, soil C/N, soil pH
Soil N	17 ± 3 kg N_2O-N %$^{-1}$ soil N	$r^2 = 0.25$*** (84)	Clay, sand, soil C, soil C/N, soil pH
Soil C/N	-0.32 ± 0.03 kg N_2O-N (C/N)$^{-1}$	$r^2 = 0.07$* (106)	Clay, sand, soil C, soil N, soil pH
Soil pH	0.47 ± 0.16 kg N_2O-N (pH)$^{-1}$	$r^2 = 0.07$** (118)	Clay, sand, soil C, soil N, soil C/N
Fertiliser	0.003 ± 0.002 kg N_2O-N kg^{-1} N-input	$r^2 = 0.01$ (106)	Fertiliser type, crop type
Crop type	0.63 ± 0.23 kg N_2O-N (crop class)$^{-1}$	$r^2 = 0.07$** (102)	Fertiliser type
Arable soils Sub-boreal Climate			
Fertiliser	0.039 ± 0.008 kg N_2O-N kg^{-1} N-input	$r^2 = 0.30$*** (62)	Fertiliser type
Grassland soils			
Drainage class	3.2 ± 0.7 kg N_2O-N (drainage class)$^{-1}$	$r^2 = 0.32$*** (42)	Clay, sand, soil N, fertiliser
Soil pH	2.3 ± 0.6 kg N_2O-N (pH)$^{-1}$	$r^2 = 0.20$*** (52)	Clay, sand, soil C/N, fertiliser type
Fertiliser	0.015 ± 0.003 kg N_2O-N kg^{-1} N-input	$r^2 = 0.23$*** (72)	Air temperature, drainage class, fertiliser type

Climate

Climate plays a key role in determining the maximum measured annual N_2O release from arable sites upon which the distinction of homogeneous climatic site groups (cf. above) was based. Within the homogeneous climatic groups, no significant quantitative relationship was found between annual precipitation, air temperature or frost and N_2O emissions from arable nor from grassland soils. Evidently, the scale and the general nature of the tested climate factors is too unspecific for a significant linear interaction with local soil processes.

Soil

In accordance with /Kaiser et al. 1996/, among the soil physical characteristics, the clay fraction best explains annual N_2O emission rates. The sand content correlates negatively with N_2O release in Temperate Western arable soils. These parameters can be interpreted as indicators of oxygen availability /Hutchinson and Davidson 1993/. Among the soil chemical characteristics, the topsoil nitrogen content best predicts N_2O emissions, which is a proxy for substrate limitation. Topsoil carbon content and pH also link with N_2O emissions, but at lower r^2. Unfortunately, precise information about the site properties has been documented in about half of the grassland studies only, restricting the scope of the analyses. The negative correlation of the silt content in grassland soils might be attributed to the interrelation with fertiliser amount (Table 13).

Management

Fertilisation affects N_2O emissions through 1) adding nitrogen and 2) the form of the nitrogen and possible carbon source as a function of the fertiliser type.

Nitrogen input turned out as the most important control of N_2O emissions of agricultural soils in general. The amount of fertiliser yields highly significant and well correlated emissions factors for grassland soils but less so for arable soils. If arable soils are analysed separately, the emission factor for fertiliser is not significant if all arable soils are analysed jointly and even less in Temperate Western arable soils. In the latter region, statistically, the soil physical and chemical features become the dominant control of the annual N_2O emissions. They promote nitrification and denitrification as well as the partitioning between the two processes. This finding opposes the presently used default methods to estimate N_2O release from agricultural soils /IPCC 1997/. However, nitrogen input by fertiliser explains a great portion of the variability in annual N_2O fluxes from Sub-boreal arable soils.

The mean fertiliser emission factors reported in the literature for European conditions are 0.013, 0.022, and 0.012 kg N_2O-N kg^{-1} nitrogen fertiliser in Temperate West arable, Sub-boreal arable, and grassland soils, respectively. They do not entirely agree with the calculated emission factors (Table 13) and the slope of the regressions (Table 14). Table 14 compares the regression equations for N input obtained in this study with the one of /Bouwman 1996/.

Interestingly, our emission factor for combined arable and grassland soils comes close to the one derived from annual studies world wide /Bouwman 1996/, which also stem mainly from temperate climate regions. Hence, the emission factor of 0.0125 kg N_2O-N kg^{-1} nitrogen fertiliser /Bouwman 1996/ as recommended for national greenhouse gas inventories in the IPCC methodology /IPCC 1997/, seems appropriate for rough conservative emission estimates on a continental scale.

Measured N_2O emissions from unfertilised treatments show average fluxes of 0.7 (0 to 1.7) kg N_2O-N ha^{-1} a^{-1} for arable soils in the Temperate West, 2.3 (0 to 6.1) kg N_2O-N ha^{-1} a^{-1} in the Sub- Boreal region, and 1.2 (0 to 5.0) kg N_2O-N ha^{-1} a^{-1} in grassland soils. They follow the average magnitude of annual N_2O emissions as displayed in Table 12 but are not reflected by the intercept of the regressions in Table 14. The disaggregation of the data into arable lands in two regions and grassland, respectively, helps to understand regional variations in fertiliser-derived emissions. The lower R² highlights the importance of controls other than N input.

Table 14 Fertiliser-derived N_2O emission estimates [kg N_2O-N ha^{-1} yr^{-1}]

Area	**Model**	**R²**	**n**
World (Bouwman, 1996)	N_2O = 1 (-0.6 to 3.2) + 0.0125 (0.0025 to 0.0225) · fert	0.8***	20
Arable + grassland soils, Europe	N_2O = 1.87*** (± 0.38) + 0.0138*** (± 0.002) · fert	0.15***	242
Arable soils, Temperate West	N_2O = 1.84*** (± 0.37) + 0.0003 (± 0.002) · fert	n.s.	106
Arable soils, Sub-boreal	N_2O = 1.71 (± 0.93) + 0.0388*** (± 0.008) · fert	0.30***	62
Grasslands, Temperate and Sub-boreal Europe	N_2O = 2.34*** (± 0.70) + 0.0152*** (± 0.003) · fert	0.23***	72

N_2O	annual emission in kg N_2O-N ha^{-1} yr^{-1}
fert	annual fertiliser input in kg N ha^{-1} yr^{-1}
n	number of data sets
n.s.	not significant

Fertiliser type. In our data base, the fertiliser type (mineral – organic – mineral plus organic) and the various forms of mineral fertilisers are highly intercorrelated with the amount of fertiliser added, so no meaningful analysis of the impact of fertiliser type on N_2O emissions was possible. In addition, most of the studies applied calcium-ammonium-nitrate fertiliser, so the analysis is also biased due to limited data for other fertiliser types. No significant correlation was found between the emission factors reported in the literature and the fertiliser types nor the mineral fertiliser form. As a result, in parallel with Bouwman (1996) and IPCC (1997), a uniform emission factor is used for all types of synthetic fertilisers as well as for manure.

In terms of crop type, in arable soils of the Temperate West, annual N_2O emissions from non-cereals (2.0 (0 – 8.7) kg N_2O-N ha^{-1} a^{-1}) exceed in general those of cereal fields (1.0 (0 – 5.2) kg N_2O-N ha^{-1} a^{-1}). However, the effect of crop type on N_2O emissions is masked by an intercorrelation with fertiliser amounts. Short-term and annual measurements on pea cultures /Duyzer 1996, Goossens et al. 2001/ and grass-clover mixtures /Allen et al. 1996, Heinemeyer et al. 1996/ do not exceed the N_2O release expected in non-leguminous crops. However, observations suggest extraordinary N_2O release in the season after the legumes /Goossens et al. 2001/. These emissions have been attributed to the succeeding crop, masking the effect for our analysis.

4.3.3 Combined effects

In the previous sections, major controls of annual N_2O emissions from European agricultural soils have been identified. Using all parameters that were correlated with annual N_2O emissions of the respective homogeneous climate class, we deduced models of N_2O emissions by stepwise multivariate linear regression analysis (Table 15). Nitrogen input was treated as obligatory parameter in order to ensure comparability with the methods of /Bouwman 1996/ and /IPCC 1997/. The emission factors in the multivariate regression models match well with those in the univariate models (Table 13) indicating that the subsample of data sets in the combined models is representative for the whole set of available data and that the explanatory variables in the models are uncorrelated /Neter et al. 1996/.

Table 15 Regression models of annual N_2O emissions from agricultural mineral soils in Europe

Model [kg N_2O-N ha^{-1} yr^{-1}]	**R^2**	**n**
Arable soils, Temperate Western Europe		
N_2O (± 0.7) = 0.6 (± 0.5) + 0.002 (± 0.002) · fert + 1.27*** (± 0.28) · soil C - - 0.024*** (± 0.005) · sand	0.38***	61
Arable soils, Sub-boreal Europe		
N_2O (± 1.6) = -1.3 (± 2.1) + 0.033*** (± 0.008) · fert + 28* (± 13) · soil N	0.31***	46
Grassland soils, Temperate and Sub-boreal Europe		
N_2O (± 2.6) = 2.4*** (± 0.7) + 0.015***(± 0.003) · fert	0.23***	72

N_2O	annual emission in kg N_2O-N ha^{-1} yr^{-1}
fert	annual fertiliser input in kg N ha^{-1} yr^{-1}
soil C	soil organic carbon content in topsoil in % of soil weight
soil N	total soil nitrogen content in topsoil in % of soil weight
sand	sand content in topsoil in % of soil weight
n	number of data sets
(± x)	standard error of predictor variables and mean standard error of models

Compared to the fertiliser-based models in Table 14, the consideration of the combined effects of fertiliser, climate and soil clearly improves the model fit (Table 15) for N_2O emissions from arable soils. In consistence with the findings in section 4.3.3, in Temperate Western arable soils, soil parameters exhibit a stronger relationship with annual N_2O emissions than fertiliser input (Table 16). This agrees with the findings of /Kaiser et al. 1996/ and /Hénault et al. 1998/. In Sub-boreal arable soils and grasslands, fertiliser remains the dominant controlling parameter (Table 16).

Table 16 Standardised coefficients in the regression models of Table 15 and the effect of their variation on annual N_2O emission estimates

Region	**Standardised coefficients**				**Effect on N_2O flux estimate in kg N_2O-N ha^{-1} yr^{-1}**			
	Fertiliser	Soil C	Soil N	Sand	Fertiliser	Soil C	Soil N	Sand
Arable soils, Temperate West	0.09	0.48		-0.47	0 - 1	0 - 4		0 - 2.4
Arable soils, Sub-boreal Europe	0.51		0.28		0 - 16		0 - 7	
Grassland soils	0.48				0 - 8			

4.4 Discussion

4.4.1 Validation

The regression models of Table 15 may serve for estimating the average order of magnitude of annual N_2O emissions from agricultural mineral soils in the temperate and sub-boreal regions of Western Europe but are not designed for predicting fluxes from a given site. Nevertheless, they are validated against measured data from independent data sets (Figure 9). Overall, the range of the measured data is readily predicted by the models without, however, matching properly the observed annual N_2O fluxes on the test sites. Given the scatter in the data for model development, the prediction intervals $N_2O_{pred} \pm se_{pred} \cdot B$ are relatively large. The standard errors of the predictions se_{pred} slightly exceed those of the regression models, B is about 1.68 for $p < 0.1$. Consequently, the average prediction uncertainty is ± 1.2, ± 2.9, and ± 4.4 kg N_2O-N ha^{-1} yr^{-1} for arable soils in the Temperate West, arable Sub-boreal soils, and grasslands, respectively. For $p < 0.05$, B and consequently the prediction uncertainty both increase by ± 20 %. The standard error of the grasslands model might decline in the future by accounting for site characteristics such as drainage status and management (pastures versus meadows). Obviously, the models allow to capture the correct order of magnitude of annual N_2O emissions when extrapolated in space and time over climate regions of Europe. Given the wide scatter, they suit less for predicting N_2O emissions at local scale.

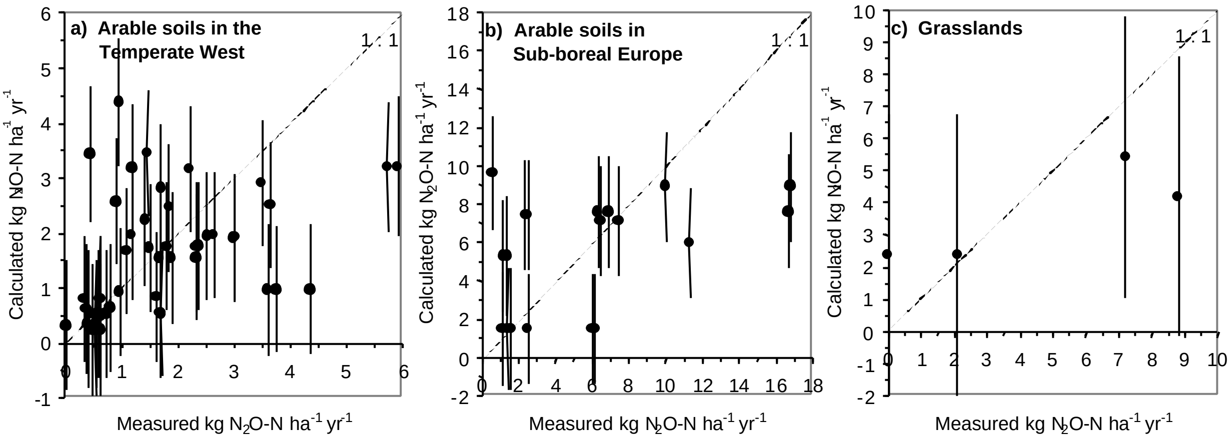

Figure 9 Validation of the multivariate regression models for a) arable soils in the Temperate West, b) arable soils in Sub-boreal Europe, and c) grasslands. Error bars indicate the prediction uncertainty for the validation test sites given by Bonferroni simultaneous prediction limits at $p < 0.1$

4.4.2 Comparison with existing approaches

Existing approaches /IPCC 1997, Bouwman 1996/ and the regression models developed in this study rely on relatively simple, linear statistical relations between annual N_2O emissions and some controls rather than on mechanistic process understanding. They should be able to generate the correct order of magnitude of measured values and the general patterns of low versus high annual gas fluxes. This hypothesis is tested for the three approaches mentioned, based on all available European data (Figure 10).

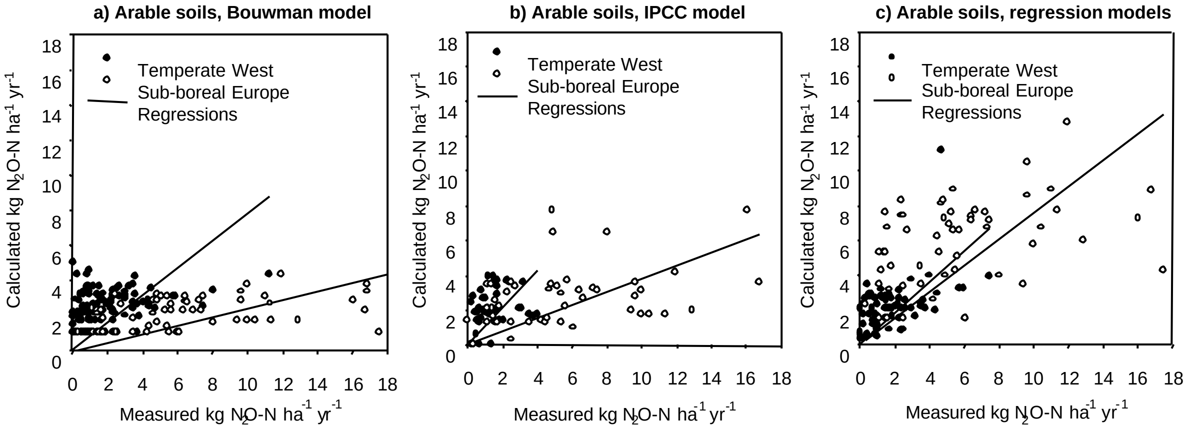

Figure 10 Comparison of measured N_2O emissions with estimates by existing models /Bouwman 1996 and IPCC 1997/ and by the new regression models

All models produce widely scattered results as compared to measured data (Figure 10). The /Bouwman 1996/ model (Figure 10a) estimates fluxes generally below 5 kg N_2O-N ha^{-1} a^{-1} but tends to overestimate small and underestimate higher fluxes, even in the Temperate West region. Both regression lines stay below the 1:1 line of measured versus modelled data. In contrast, the /IPCC 1997/ model (Figure 10b) captures at some extent the regional pattern of the measured data. It predicts the data in the Temperate West at an adequate magnitude while in Sub-boreal Europe, it underestimates the measured data exceeding 8 kg N_2O-N ha^{-1} a^{-1}. The new regression models (Figure 10c) follow most closely the 1:1 line of measured versus modelled data in both climate regions. The models fit local measurements in the Temperate West with an average deviation of 1-2 kg N_2O-N ha^{-1} yr^{-1}. In the Sub-Boreal region, the new regression model is clearly superior to the other approaches although still substantial errors occur. It estimates local N_2O emissions with an average deviation of <4 kg N_2O-N ha^{-1} yr^{-1}.

For grasslands, the three approaches use N-input as control parameter only, with (/Bouwman 1996/ and the model developed here) or without /IPCC 1997/ background emissions, so their general behaviour is similar. The new grassland model developed in this study estimates N_2O emissions at 1 to 2 kg N_2O-N ha^{-1} yr^{-1} above the model by /Bouwman 1996/, which in turn includes 1 kg N_2O-N ha^{-1} yr^{-1} background emissions not considered by the /IPCC 1997/ models (Figure 11). The regression model overestimates small annual N_2O fluxes while it better fits the average expected emission rates than /Bouwman 1996/ and /IPCC 1997/. The average deviation of estimates at local scale can exceed 4 kg N_2O-N ha^{-1} yr^{-1} in all models (Figure 11).

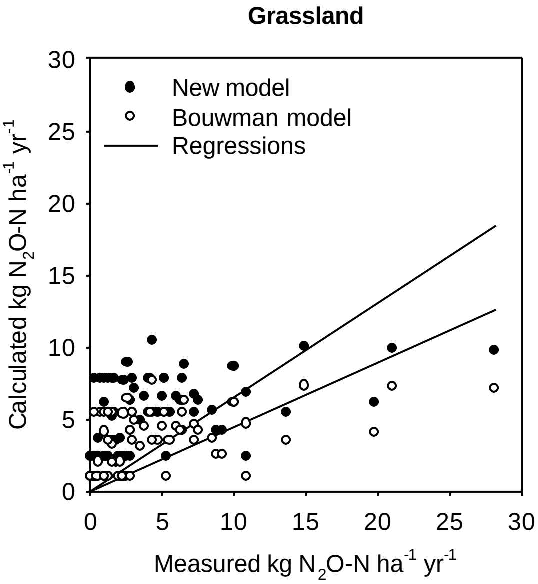

Figure 11 Comparison of measured N_2O emissions with modelled estimates by Bouwman (1996) and the new grassland model

4.5 Summary and conclusions

Based on a review of N_2O field studies in Europe, the influence of soil, climate and management on the annual N_2O release from agricultural mineral soils in Europe was assessed. Using stepwise multivariate linear regression analysis, simple first order models of N_2O emissions were established suitable for the calculation of inventories from sub-national to continental scale in the European temperate and sub-boreal climatic regions. This method identifies statistical relations between annual N_2O emissions and some important controls and allows – in contrast to the approaches of /Bouwman 1996/ and /IPCC 1997/ – to quantify N_2O emissions from arable mineral soils with a variable background emission in dependence on site conditions and climate. We have shown that for improving estimates of N_2O emissions it is necessary to stratify the agricultural soils of Europe on the basis of environmental and management characteristics. Climate proved important for N_2O emissions from arable fields but not for grasslands.

For modelling annual N_2O emissions on arable soils in Europe, two climate regions were distinguished, the “Temperate West” (UK, France, Belgium, The Netherlands, Denmark and Northern Germany) and the “Sub-boreal” region (Finland, Sweden, South Germany; no data available for Austria and Switzerland, which would be expected to fall in this group). Nitrogen input by fertiliser and soil characteristics (texture, soil organic carbon and soil nitrogen content) quantifiably determine the annual N_2O release rate. Each of these parameters allows to explain a small part of the variability in the N_2O data in at least one climate region, so in a whole, the multivariate models of Table 15 represent a clear improvement for the estimation of N_2O emissions from arable mineral soils in Europe in comparison to the currently used approaches based on N-input /Bouwman 1996, IPCC 1997/. In contrast to existing large scale approaches, both the order of magnitude of fluxes and the reaction of flux rates to changes in controlling factors is roughly explained by the new models. For grasslands, the /Bouwman 1996/ model was modified towards higher emission estimates. The large variability in the database prevented from an incorporation of site and climate parameters, which would, however, be highly desirable for future improvement.

The relatively low values for R^2 suggest that some important controlling factors could not yet been integrated in the models, which were either not quantifiable due to their ambiguous interaction with N_2O production or not sufficiently documented in the literature. This restriction applies particularly to soil physical and chemical properties and organic carbon input by crop residues and manure. The lack of detailed site and management descriptions in the literature makes the interpretation and generalisation of local measurements difficult. In order to facilitate future synthesis, the following frame data should be given as a minimum in future studies: precipitation, position in the landscape (plane, top, slope, depression), soil type, detailed texture data, soil organic carbon and nitrogen in the topsoil, soil pH, drainage and soil moisture changes, N input, crop type, yields or N removed. Very few data are available from the Mediterranean region. Therefore, more long-term studies are urgently needed in Southern Europe for all typical crops before N_2O emissions can be generalised for this climate region. Also greater emphasis on leguminous crops is desired in order to understand the role of nitrogen fixation during an entire crop rotation.

Furthermore, the temporal variability of site conditions is not considered in these regression models, so changes of the importance of controls and the complex interactions among them are disregarded here. The mean standard error of the models is of the same order of magnitude as the temporal and spatial uncertainty in the underlying observations.

Despite of these restrictions to be overcome in future, major soil, climate and management controls of N_2O release from agricultural mineral soils in the European Union were identified, which can be easily gathered from statistical services, and empirical statistical models were established which allow – in contrast to existing large-scale approaches –a regionally disaggregated assessment of N_2O emissions from sub-national to continental scale.

We applied the regression equations in a GIS-based inventory of N_2O emissions from European agriculture /Freibauer, accepted/. As a result, in EU-15 in 1995, agricultural soils emitted 419 Gg N_2O-N, which compares well to 380 Gg N_2O-N reported in official inventories based on the /IPCC 1997/ methodology /Ritter 1999/. In contrast, as expected, results for national inventories differ more /Freibauer, accepted/.

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5 Regionalised Inventory of Biogenic Greenhouse Gas Emissions from European Agriculture[3]

Abstract

This study develops a detailed methodology compatible to the Guidelines of the Intergovernmental Panel on Climate Change (IPCC) to assess the annual direct biogenic emissions of greenhouse gases released from European agriculture. This approach relies on emission factors and regression equations derived from all long-term measurements in Europe available by the end of 2001. Applying the methodology, the biogenic greenhouse gas emissions from agriculture within the European Union (EU) and within its Member States are calculated for the period from 1975 to 1997 at a spatial resolution of regions or federal states (NUTS 1-2 level). As a result, in 1995, European agriculture emitted 0.84 ± 0.19 Tg N_2O*, 8.1 ± 2.0 Tg* CH_4 *and 39 Tg ± 25* CO_2*, which adds up to 470 ± 80 Tg* CO_2*-equivalents or 11% of the overall greenhouse gas emissions. At the EU level, these numbers are surprisingly close to the official inventory submitted under the United Nations Framework Convention on Climate Change (UNFCCC). But compared to the latter, the approach taken here leads to higher agricultural* CH_4 *emissions in Austria and the Netherlands, at least 20 % lower* CH_4 *emissions in Denmark, Germany, Greece, Spain, and Sweden, and higher* N_2O *emissions in most EU Member States. In countries with – even small – areas of farmed organic soils,* CO_2 *emitted from peat oxidation can significantly contribute to the overall emissions. Hence, only the detailed approach adequately resolves regional and national specifics of agricultural conditions. It furthermore reduces the uncertainty in the emissions estimates to half of the one in inventories based on the IPCC Guidelines. Fair agreement with inverse atmospheric models was achieved. These results suggest that the methodology developed and applied here could serve as a significantly improved standard for official inventories of biogenic greenhouse gas emissions from EU Member States.*

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5.1 Introduction

Inventories of methane (CH_4) emissions from ruminants have been published since the 1970s /Ehhalt 1974/. With increasing awareness of the anthropogenic contribution to climate change, the inventories were stepwise extended to more agricultural sources and sinks as well as to other trace gases like carbon dioxide (CO_2) and nitrous oxide (N_2O). The complexity of the applied methodology varies with the aim and scope of the inventory and with knowledge available about controlling factors and emission rates. Naturally, this knowledge increases with the period and intensity of research and is hence greatest for CH_4 from enteric fermentation and ammonia (NH_3), but limited for N_2O, nitric oxide (NO), volatile organic carbon (VOC) and carbon sequestration. The most widely used approach to greenhouse gas (GHG) inventories for agriculture at the national to continental scale relies on emission factors, which reduce the complex features of trace gas production, transport and consumption to a simplified standard situation that can be described by statistical data (e.g. /Heyer 1994; Chadwick et al. 1999/).

Countries having ratified the United Nations Framework Convention on Climate Change /UNFCCC 1997/ have committed themselves to annually report their GHG emissions, preferably using the standard methodology of the /IPCC 1997/ Guidelines. Since the latter was designed for global application, the default emission factors for agricultural sources characterise broadly averaged conditions, and the default data for the characteristics of animal husbandry systems may impose considerable bias on national inventories. It still has to be proven that the emission factors adequately fit to agricultural practice and conditions in EU Member States. At present, we are not aware of any detailed model specifically designed for European conditions that has a spatial resolution as high as district or federal states level. Such features would significantly improve the accuracy and precision of the national inventories submitted under the UNFCCC.

Therefore, this study aims at developing a methodology with special reference to agricultural conditions in EU Member States, which is applicable on regional, national and continental scale. In order to make this approach feasible for all EU Member States and expandable to Applicant States, it relies on emission factors and linear regressions multiplied by activity data which can be easily derived from census and soil maps. The methodology uses Europe-specific emission factors and regressions and refines the /IPCC 1997/ approach where sources and sinks are missing (animal houses, CH_4 sink in soils, CO_2 emissions from continuously farmed peatlands), where emissions can be further disaggregated (manure management) and where local conditions interfere with emission factors (N_2O emissions from agricultural land). It is applied to quantify the biogenic direct greenhouse gas sources and sinks in the agriculture of EU Member States at a subnational resolution (NUTS 1 to NUTS 2 – federal states to sub-regions; /EUROSTAT 1999/). The results are analysed for statistical uncertainties, used to test the reliability of the national reports submitted by EU Member States under the UNFCCC /Ritter, 1999/ and verified against inverse atmospheric models.

5.2 Emissions calculations

In the following, the calculation procedures, the sources of activity data and the statistical methods for performing uncertainty analyses are described. The methodology of this study supplements the /IPCC 1997/ method as well as the related approach of CORINAIR /EEA, 2000/.

All literature available by the end of 2001 about European long-term measurements of soil emissions and experimental data on emissions from animal houses and manure management at real scale conditions is reviewed in order to derive the emission factors and regression equations for the quantification of carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) emissions from agriculture in Europe. All emission factors are given with either the standard error whenever possible or with the range of underlying data.

5.2.1 Agricultural soils

Nitrous oxide

Based on a review of N_2O field studies in Europe, major soil, climate and management controls of the annual N_2O release from agricultural mineral soils are identified, which can be easily gathered from statistical services and soil maps in EU Member States. By means of stepwise multivariate linear regression analysis, empirical models of N_2O emissions are established which allow – in contrast to existing large-scale approaches – a regionally disaggregated estimate of N_2O emissions at the subnational, national and continental scale in the major climate regions of Europe /Freibauer and Kaltschmitt, accepted/. Arable soils in oceanic and low subcontinental temperate climate emit less N_2O than those in temperate pre-alpine, alpine and sub-boreal climate with severe winter frost where high freeze-thaw emissions occur during spring. Therefore, two separate regression models are deployed /Freibauer and Kaltschmitt, accepted/. In arable soils of the temperate oceanic climate zone, the emission of N_2O from agricultural soils E_{N2O} [kg ha^{-1} yr^{-1} N_2O-N] is calculated as a function of the nitrogen input by mineral and organic fertiliser *fertiliser* [kg ha^{-1} yr^{-1} N], the organic carbon content in topsoil *soil C* [g kg^{-1} of soil weight in A horizon], and the sand content in topsoil *sand* [g kg^{-1} of soil weight in A horizon] (1). The model for arable soils in climate with severe winter frost has the parameters nitrogen input by mineral and organic fertiliser *fertiliser* [kg ha^{-1} yr^{-1} N] and the total nitrogen content in topsoil *soil N* [g kg^{-1} of soil weight in A horizon] (2). Compared to existing methods for large scale inventories, the regression models allow a better regional fit to measured values since they integrate additional driving forces of N_2O emissions such as climate, substrate availability represented by soil carbon and nitrogen, and oxygen availability, represented at some extent by sand content /Freibauer and Kaltschmitt, accepted/. Nitrous oxide has rarely been measured in Mediterranean soils, but results from short-term studies in Italy and Spain /Arcara et al. 1990; Teira-Esmatges et al. 1998/ suggest low to moderate emission rates in analogy to soils in oceanic temperate climate. For grasslands, E_{N2O} is modelled by a fertiliser-based approach similar to the one of /Bouwman 1996/, but with slightly higher annual emissions (3). Due to an extreme variability, climate, soil and management parameters cannot yet be included in the empirical grasslands model. Equations (4) to (5) address farmed organic soils. The brackets in the equations indicate the respective standard errors of the parameters.

Mineral arable soils in the temperate oceanic and Mediterranean climate (East and South Austria, Belgium, Denmark, France, Germany North of 49°N, Greece, Luxembourg, Ireland, Italy, The Netherlands, Portugal, Spain, South Sweden, UK):

$$E_{N2O} = 0.6(\pm 0.5) + 0.002(\pm 0.002) \cdot \mathit{fertiliser} + 12.7(\pm 2.8) \cdot \mathit{soil\ C} - 0.24(\pm 0.05) \cdot \mathit{sand} \qquad (1)$$

Mineral arable soils in the pre-alpine, alpine and sub-boreal climate regions (North and West Austria, Finland, Germany South of 49°N, Sweden except South, Switzerland):

$$E_{N2O}\ -1.3(\pm 2.1) + 0.03(\pm 0.008) \cdot \mathit{fertiliser} + 280(\pm 130) \cdot \mathit{soil\ N} \qquad (2)$$

Mineral grassland soils in the temperate and sub-boreal climate regions:

$$E_{N2O} = 2.4(\pm 0.7) + 0.015(\pm 0.003) \cdot \mathit{fertiliser} \qquad (3)$$

Farmed organic soils:

$$E_{N2O} = 7(6\text{-}9)\ kg\ ha^{-1}\ a^{-1}\ N_2O\text{-}N \text{ for grasslands and cereal crops} \qquad (4)$$

$$E_{N2O} = 20(10\text{-}30)\ kg\ ha^{-1}\ a^{-1}\ N_2O\text{-}N \text{ for vegetables and root crops} \qquad (5)$$

Given the range of the data underlying the models, equations (1) to (3) can be applied for the following range in parameters: (1) to (3) for fertiliser input from 0 to 500 kg ha^{-1} yr^{-1} N; (1) is further restricted to topsoil organic carbon contents from 5 to 82 g kg^{-1} and topsoil sand contents from 15 to 857 g kg^{-1}, and (2) to topsoil nitrogen contents between 0.7 to 2.5 g kg^{-1}. The emission factors for N_2O from organic agricultural soils (equations (4) and (5)) are derived from unpublished results (L. Klemedtsson, in /Freibauer and Kaltschmitt 2001/).

For typical soil conditions and fertilisation rates under common agricultural practice in Europe and mean flux rates, the regression models have a mean standard error of 40 to 50 % (equation (1)), 30 % (2) and 70 to 100 % (3). This considerably improves the /IPCC 1997/ approach with an original estimated uncertainty of 70 to 80 % /Lim et al. 1999/ for arable soils. Our approach for grasslands is similar to /IPCC 1997/. The large uncertainty in our case results from widely scattered measured data underlying the regression model.

In inventories of N_2O emissions from agricultural soils, the description of soil characteristics and land use introduces additional uncertainty which is quantified by error propagation. As a result, inventories using equation (1) are associated with an uncertainty of 41 % a quarter of which pertains to uncertainty in the constant of the equation and to the uncertainty in the topsoil carbon contents, respectively (c.f. section 3.2). The uncertainty in inventories based on equation (2) amounts to 64 %, more than half of which is explained by the uncertainty in the emission factor for soil nitrogen and in the topsoil nitrogen contents.

Grassland inventories have an estimated uncertainty of 29 %, most of which stems from the constant of the equation. Given the relatively bad representation of measured N_2O emissions by equation (3) indicated by the high mean standard error, the actual accuracy of the grassland emissions might be overestimated.

Methane

Methanogenesis is restricted to completely anaerobic soil conditions as occur in rice paddies when the soils are flooded. These CH_4 emissions are calculated by the /IPCC 1997/ methodology. All other agricultural soils take up CH_4 through oxidation by methanotrophic bacteria except for poorly drained sites. Fertilised and cultivated soils consume CH_4 at rates of 40 % or less of those in undisturbed forest soils as a result of complex short-term and long-term adverse effects of soil disturbance, shift in microbial species diversity and ammonium inhibition of methanotrophic activity /Hütsch 2001/. This sink of CH_4 E_{CH4} [$kg\ ha^{-1}\ yr^{-1}\ CH_4$] is quantified as in equation (6).

$$E_{CH4} = -0.5(\pm 0.5)\ kg\ ha^{-1}\ yr^{-1}\ CH_4 \qquad (6)$$

The emission factor relies on a set of 66 long-term measurements in Europe analysed by /Freibauer and Kaltschmitt 2001/ and agrees well with the log-mean oxidation rate of -0.65 $kg\ ha^{-1}\ a^{-1}\ CH_4$ of agricultural soils in Northern Europe given by /Smith et al. 2000/.

Carbon dioxide

Carbon dioxide emissions considered here result from the mineralisation of peat through drainage and soil disturbance and hence reflect the background emissions in farmed organic soils without fertilisation. Equations (7) and (8) show the emission factors E_{CO2} [$Mg\ ha^{-1}\ yr^{-1}\ CO_2$] for grasslands and arable land. Again, the emission factors rely on unpublished results (L. Klemedtsson, in /Freibauer and Kaltschmitt 2001/).

$$E_{CO2} = 10 \pm 5\ Mg\ ha^{-1}\ a^{-1}\ CO_2 \text{ for grassland} \qquad (7)$$

$$E_{CO2} = 15 \pm 5\ Mg\ ha^{-1}\ a^{-1}\ CO_2 \text{ for arable crops} \qquad (8)$$

Land use change and associated changes in the carbon cycle are not considered here since these areas represent a minor fraction of the European agricultural surface /Rabbinge and van Diepen, 2000/.

5.2.2 Animal husbandry

Nitrous oxide

The emission factors for N_2O from animal excreta in animal houses, manure storage and dropped on pasture are derived as the median of all available European measurements (Table 17). The fertiliser-dependent emissions from agricultural soils include N_2O emissions from manure spreading. The small number and the wide range of the measured data renders a reliable distinction between farmyard manure (FYM) and slurry-based management systems and by animal types difficult. Indeed, the difference between N_2O emissions from FYM and slurry-based management is smaller than in the /IPCC 1997/ Guidelines because the emission factors for FYM systems are lower and those for slurry systems are comparable or higher. Given the frequency distribution and range of measured data, the average uncertainty in the emission estimates is 80 %, equivalent to the one in the /IPCC 1997/ methodology /Lim et al. 1999/, but the approach taken here is more detailed.

Methane

Methane is released from ruminants as well as from manure management. Both sources are considered here. Due to intensive research efforts during the last decades, the quantification of CH_4 release from enteric fermentation by the IPCC Tier 1 methodology is already detailed and well established /IPCC, 1997; EEA, 2000/. /EEA 2000/ estimates the uncertainty at 30 %.

In the field of manure management, the approach taken here introduces the animal house as an additional source of CH_4, which has been neglected in existing default methodologies. This accounts for the differences in temperature, aeration and physical disturbance to which the excreta are exposed in the house and in the subsequent separate store (Table 18). Methane fluxes from manure inside the animal houses distinguish between animal types and manure types. Measurements in animal houses generally refer to the sum of CH_4 emissions from the animals and the manure in the house. The contribution of the manure to the total measured CH_4 emissions in animal houses with a respective manure management system is estimated as follows: cattle – slurry-based systems 15-20 % /Amon et al. 1998/ (*i.e.* 80-85 % of the measured CH_4 is attributed to enteric fermentation), cattle – farmyard manure (FYM)-based systems 5 to 10 % /Kinsman et al. 1995; Amon et al. 1998/, swine – slurry-based systems 70 %, swine – FYM-based systems 50 %, poultry 100 %. The emission factors for CH_4 emissions from animal houses indicate the median of the manure part in all available European measurements (Table 18).

For CH_4 emissions from manure storage, the /IPCC 1997/ Tier 2 approach has been widely adopted in national inventories of EU Member States. It is also used here because the few available real scale measurements of CH_4 emissions from stored manure as recently reviewed by /Jungbluth et al. 2001/ and /Freibauer and Kaltschmitt 2000/ do not allow to establish new emission factors. The wide range in the results and uncertainties introduced through the conversion of the reported units to a standard unit allow a coarse validation of the /IPCC 1997/ Tier 2 approach only. The real scale data suggest higher or similar emissions from stored cattle and swine FYM and lower or similar emissions from stored slurry, which justify the application of the /IPCC 1997/ Tier 2 approach for European conditions under present knowledge.

Given the uncertainties in the emission factors and those in the distribution of manure storage systems the overall uncertainty in the annual emission is in the order of 40 to 50 %, as is state of the art /Lim et al. 1999/.

Table 17 Nitrous oxide emission factors for livestock

Class of livestock	**Farmyard manure**			**Slurry**		
	Emission in g N_2O-N (kg N excreted)$^{-1}$	**Range g N_2O-N (kg N excreted)$^{-1}$**	**Number of studied systems**	**Emission in g N_2O-N (kg N excreted)$^{-1}$**	**Range g N_2O-N (kg N excreted)$^{-1}$**	**Number of studied systems**
Animal houses						
Cattle	2	0.4-34.6	7[a]	2	0.2-3.2	2[g]
Sheep, goats	1		(Sneath et al. 1997b)	1		(Sneath et al. 1997b)
Swine	14	0-46	17[b]	2	0.3-16	12[h]
Poultry	12	0-37.6	16[c]			
Manure store						
Cattle	5	0-30	25[d]	2	0-11.5	12[i]
Sheep, goats	5			2		
Swine	2	0-13.2	8[e]	1	0-19	2[k]
Poultry	0.1		2[f]			
Animal droppings on pasture						
all	20	5-30	(IPCC 1997)			

[a] Amon et al. (1998); unpublished data of D.R. Chadwick; Sneath et al. (1997b)

[b] Ahlgrimm and Breford (1998); unpublished data of H.-J. Ahlgrimm; Heinemeyer et al. (1997); Hörnig et al. (1996); Hoy et al. (1996); Kaiser and van den Weghe (1997); references cited in Jungbluth et al. (2001)

[c] Neser et al. (1997); Sneath et al. (1997b); Wathes et al. (1997); references cited in Jungbluth et al. (2001)

[d] Ahlgrimm et al. (1996); Amon et al. (1998); unpublished data of D.R. Chadwick; Schuchardt and Hüther (1996); Sneath et al. (1997b)

[e] Ahlgrimm et al. (1996); Ahlgrimm and Breford (1998); unpublished data of H.-J. Ahlgrimm; Petersen et al. (1998); Schuchardt and Hüther (1996); Sibbesen and Lind (1993)

[f] Sneath et al. (1997b)

[g] Amon et al.(1998); Sneath et al. (1997b)

[h] unpublished data of H.-J. Ahlgrimm; Hahne et al (1999); Heinemeyer et al. (1997); Hoy et al. (1996); Sneath et al. (1997b); references cited in Jungbluth et al. (2001)

[i] Hüther et al. (1997); Phillips et al. (1997); Sneath et al. (1997b)

[k] Ahlgrimm et al. (1996); Hüther et al. (1997)

Table 18 Methane emission factors from livestock

Class of livestock	Farmyard manure			Slurry		
	Emission in $g\ CH_4\ head^{-1}\ d^{-1}$	Uncertainty % of median	Number of studied systems	Emission in $g\ CH_4\ head^{-1}\ d^{-1}$	Uncertainty % of median	Number of studied systems
Animal houses						
Dairy cattle	36	+/- 50 %	3-5[a]	45	+/- 50 %	1-3[e]
Other cattle	12	+/- 50 %	2[b]	20	+/- 50 %	0
Sheep, goats	7	+/- 50 %	0	10	+/- 50 %	0
Swine	3	+/- 50 %	9[c]	9	+/- 50 %	10[f]
Poultry	0.004 broilers 0.12 laying hens	+/- 50 %	8[d]			
Manure store						
all	(IPCC 1997) Tier 2 approach					
Manure spreading						
Dairy cattle	0.17			0.077		
Other cattle	0.17			0.077		
Sheep, goats	0.08			0.039		
Swine	0.020			0.022		
Poultry	0.00004					
	EFs adapted from (Sneath et al. 1997a), (Chadwick et al. 2000)			EFs adapted from (Sneath et al. 1997a), (Chadwick et al. 2000)		
Animal droppings on pasture						
All	0					

A Amon et al. (1998); Brunsch et al. (1993); Kinsman et al. (1995)

B Amon et al. (1998)

C Ahlgrimm and Breford (1998); unpublished data of H.-J. Ahlgrimm; Hörnig et al. (1996); Sneath (1996)

D Wathes et al. (1997); Neser et al. (1997); Sneath et al. (1997a); references cited in Jungbluth et al. (2001)

E Amon et al. (1998)

F Ahlgrimm and Breford (1998); Brunsch et al. (1993); Sneath (1996); Sneath et al. (1997a); references cited in Jungbluth et al. (2001)

Methane emissions from manure spreading are calculated here in addition to the sources covered by /IPCC 1997/ although the rates are small /Sneath et al. 1997a; Chadwick et al. 2000/ (Table 18).

Methane is emitted from animal droppings on pasture at even lower rates than from manure spreading /Sneath et al. 1997a; Chadwick et al. 2000/ and is hence set zero.

5.3 Activity data

Data needed to calculate regional, national and EU-wide inventories encompass the area of arable land, grassland, farmed organic soils, fertiliser rates, livestock numbers and characteristics, manure management systems, as well as soil and climatic properties. The sources of the data are given below.

5.3.1 Statistical data

The agricultural land area and the livestock data are taken from the EUROSTAT New Cronos Database at a spatial resolution of NUTS 1 to NUTS 2 Level (regions or federal states, about 130 regions in EU-15; /EUROSTAT 1999/). Occasional gaps are closed by national data from various national statistical services. Such gaps occur particularly when the European Union comprised a smaller number of Member States and for the former German Democratic Republic (GDR). The amount of fertiliser spread is estimated from national and subnational sales statistics.

5.3.2 Soil characteristics

The Soil Geographical Data Base of Europe /JRC-SAI 2000/ provides a harmonised map on European level at a maximum resolution of 9 ha for soil types, some soil properties and land use. Spatial information on organic soils can be directly gathered from the soil type Histosols. However, small patches of organic soils, which typically occur in hilly Pleistocene landscapes, cannot be detected. Furthermore, the spatial distribution of agricultural land use areas inferred in the database is not consistent with the CORINE Land Cover Data Base /EEA 1995/. Therefore, the resulting farmed organic soil areas are compared with an independent survey of peat resources /Lappalainen 1996/, for which, in turn, the distribution of arable lands and grasslands is determined by expert judgement only.

Relevant information on topsoil organic carbon contents can be derived by so-called pedotransfer functions /Daroussin and King 1996/ applied to the Soil Geographical Data Base. The topsoil organic nitrogen contents are derived from the topsoil organic carbon contents by applying a C/N ratio of 11, the average ratio reported in European studies of soil N_2O

emissions /Freibauer and Kaltschmitt, accepted/. The sand content is estimated from the topsoil textural class. Since all soil characteristics extracted from the Soil Geographical Data Base exist as classified values only, we calculated the respective area-weighted mean, minimum and maximum values of the parameters for arable soils at a spatial resolution according to NUTS 3 (county subdivision) level using ArcView GIS 3.1 and statistical programmes. This introduces an uncertainty of 40 % to the mean values for soil properties used in the analyses.

5.3.3 Livestock management systems

At present, the lack of reliable information upon the distribution of animal house types among the livestock categories and regions represents one of the greatest challenges for assessing GHG emissions from manure. Here, the distribution of excreta between outdoor grazing and indoor farmyard manure (FYM) and slurry systems and the amount of nitrogen excreted are estimated by expert judgements on a national to subnational (France) basis. The uncertainty in the judgement is estimated at 20 %. Ammonia volatilisation losses are subtracted in order to quantify the amount of animal-derived nitrogen in animal houses, manure storage, manure spreading, and pasture, respectively, which is available for nitrification and denitrification. The NH_3 loss rates are determined by CORINAIR emission factors /EEA 2000/ and national inventories /ECETOC 1994; Menzi et al. 1997/.

5.4 Results

The GHG emissions from European agriculture are quantified by a spreadsheet model with a spatial resolution at NUTS 1 to NUTS 2 level although the soil data have a higher resolution. Given the lack of adequately resolved data on livestock management systems and the inherent uncertainty in the methodology and in underlying measurements, a higher spatial resolution would not increase the information content. Emissions are given in detail for 1995 and trends cover the period from 1975 to 1997. The emissions are expressed per gas species (*i.e.* Gg CH_4 yr^{-1}, Gg N_2O yr^{-1}, and Tg CO_2 yr^{-1}). They are subsequently converted to CO_2-equivalents by their respective global warming potentials of 21 for CH_4 and 310 for N_2O, referring to CO_2 as the reference substance /IPCC 1996/ when set against a time horizon of 100 years.

5.4.1 Agricultural soils

Regional distribution of emissions

Soil and climatic conditions as well as land management control the GHG emissions from agricultural soils. As a result, GHG emissions vary, in average, from 0.7 Mg ha^{-1} yr^{-1} CO_2-equivalents on sandy arable soils to 25 Mg on organic soils cropped with vegetables (Figure 12), with average flux rates around 2 Mg ha^{-1} yr^{-1} CO_2-equivalents. When compared to the size of the areas, mineral arable lands and grasslands emit GHGs in an underproportionate way. In contrast, the relatively small area of farmed organic soils emits more than a fifth of the GHGs from agricultural soils (c.f. Table 19, Figure 12).

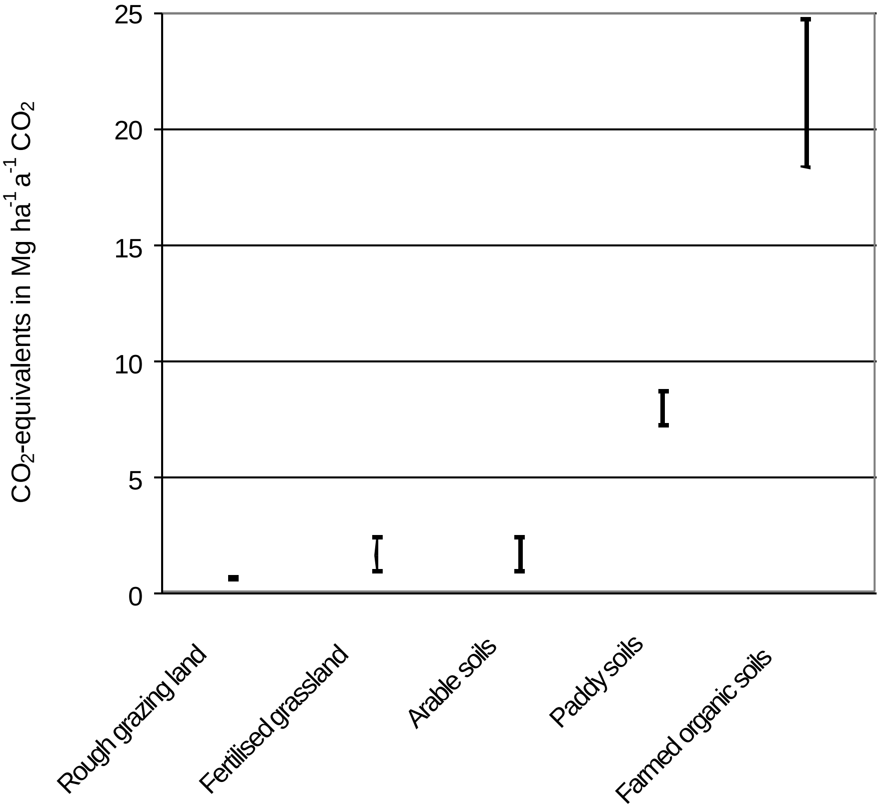

Figure 12 Average emissions per hectare of CO_2-equivalents from agricultural ecosystems in Europe

Table 19 Greenhouse gas emissions from agricultural soils in the European Union (EU-15)

Land use	**1995 CH_4 Gg yr^{-1}**	**1995 N_2O Gg yr^{-1}**	**1995 CO_2 Tg yr^{-1}**	**1995 CO_2-equiv. Tg yr^{-1}**	**% of soil emission CO_2-equiv.**	**% of agric. area**	**Change 1975-1990 CO_2-equiv.**	**Change 1990-1997 CO_2-equiv.**
Arable soils	110 ± 84	391 ±164	-	126 ±46	49 %	61 %	-4 %	-2 %[a]
Grassland soils	-22[b] ± 22	259 ±75	-	80 ±26	31 %	36 %	-5 %	-9 %
Farmed org. soils	-	42 ± 25	39 ± 25	52 ± 33	20 %	3 %	0 %	0 %
Total soils	**88 ± 67**	**694 ±182**	**39 ± 25**	**256 ±61**	**100 %**	**100 %**	**-4 %**	**-4 %**

No significant temporal changes in emissions of CO2-equivalents.

a significant increase of CH4 emissions from rice paddies

b methane sink

Owing to the detailed methodology, the N_2O emissions from agricultural soils reflect the land use intensity and agricultural focal areas at a subnational level (Figure 13, Figure 14). In general, elevated N_2O fluxes occur where the land receives high doses of nitrogen, where soils are moist, rich in humus and experience freeze-thaw cycles. In arable soils (Figure 13), the highest N_2O fluxes occur in situations of excessive nitrogen fertilisation combined with elevated soil carbon contents (Netherlands) and in the intensively managed pre-alpine regions where high spring emissions produce annual N_2O fluxes above the European average. In Ireland and parts of Italy, high average N_2O fluxes from arable soil are related to high topsoil carbon contents in fine-textured soils. In contrast, the other Mediterranean and extensively used regions show lower average emissions. Grassland emissions (Figure 14) reflect the average fertilisation. Assuming that all animal excreta are spread in the same NUTS region where the animals are registered, high livestock densities also produce high average N_2O fluxes from arable and grassland soils (Belgium, Denmark, parts of Germany, Netherlands, alpine areas; Figure 13, Figure 14).

Methane and CO_2 emissions indicate the regional distribution of rice paddies, mineral agricultural soils and farmed organic soils, respectively. As a regional average, agricultural soils in Europe emit 2.1 Mg ha^{-1} yr^{-1} CO_2-equivalents with a range from 0.8 to 8.9 Mg ha^{-1} yr^{-1} CO_2-equivalents.

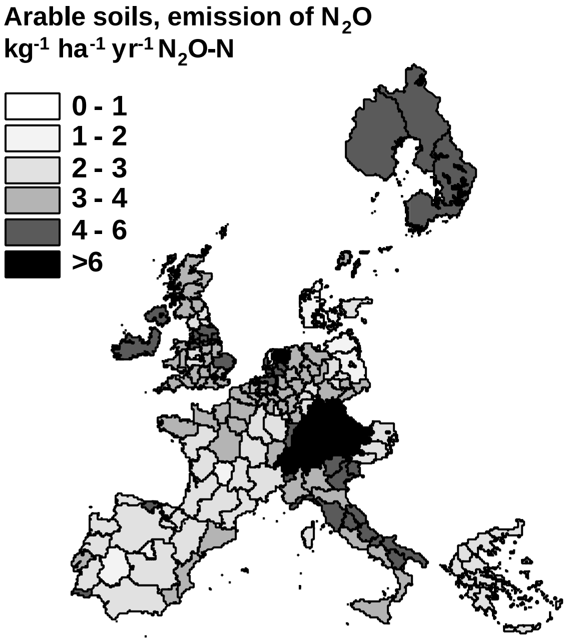

Figure 13 Regional distribution of average emissions of N_2O per hectare mineral arable soil in 1995 [kg ha^{-1} yr^{-1} N_2O-N]

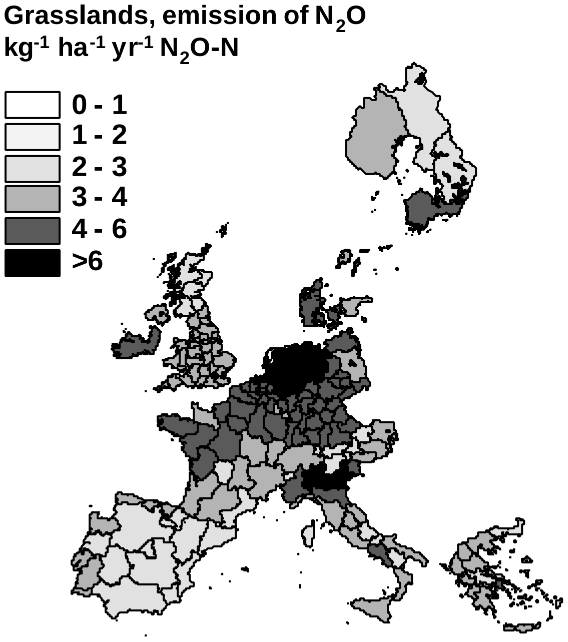

Figure 14 Regional distribution of average emissions of N_2O per hectare mineral grassland in 1995 [kg ha^{-1} yr^{-1} N_2O-N]

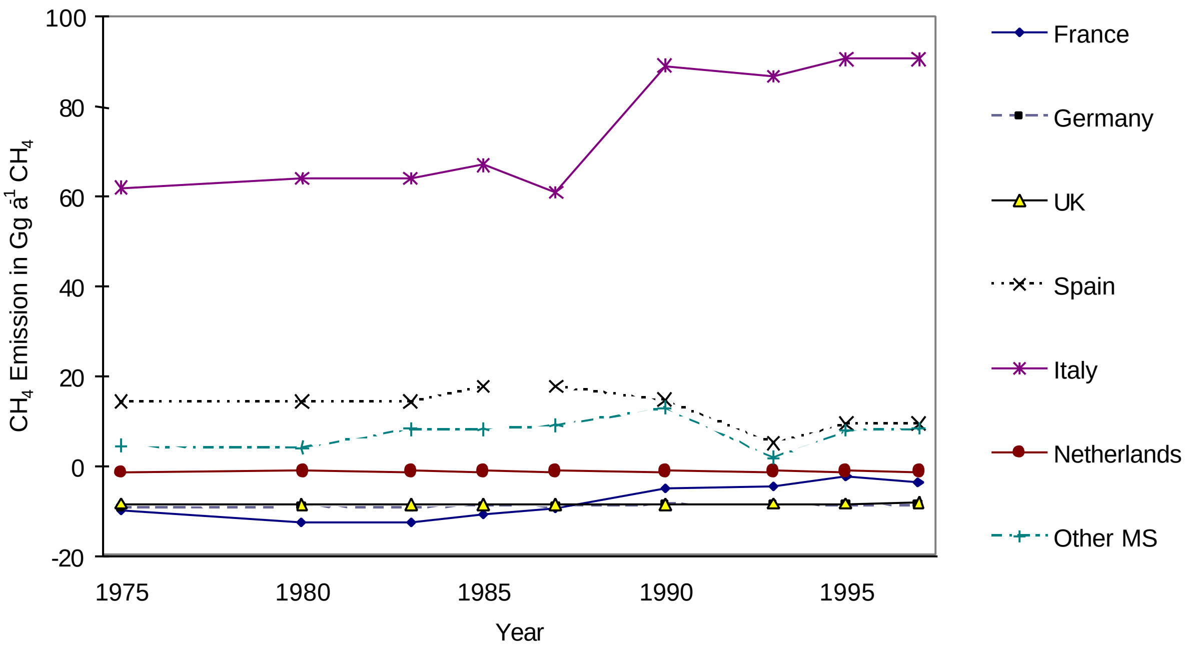

Figure 15 Methane fluxes on agricultural soils (Average uncertainty: France 600 %; Germany 100 %; UK 100 %; Spain 120 %; Italy 28 %; Netherlands 100 %; EU average 76 %); MS Member States

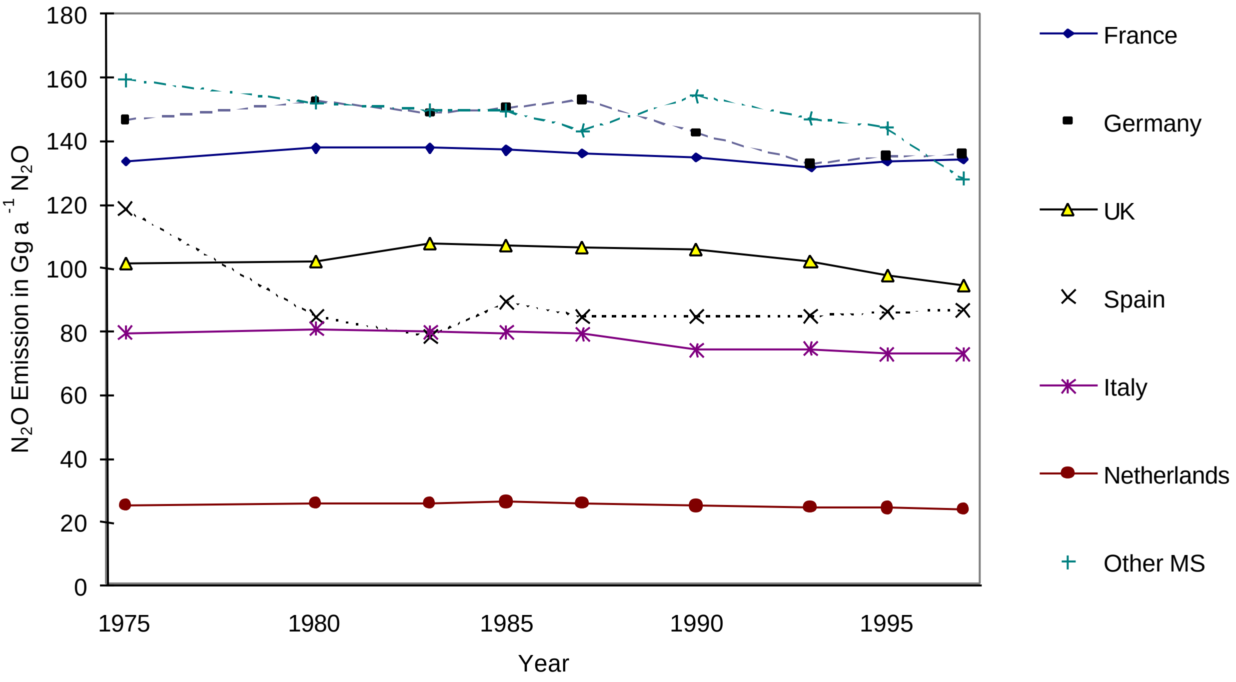

Figure 16 Nitrous oxide fluxes on agricultural soils (Average uncertainty: France 28 %; Germany 26 %; UK 25 %; Spain 36 %; Italy 37 %; Netherlands 25 %; EU average 28 %); MS Member States

National greenhouse gas emissions and temporal trends

In terms of CO_2-equivalents, the European GHG emissions from agricultural soils (Table 19) are composed of 1% CH_4, 11 % CO_2, and 89 % N_2O.

Average emissions per hectare and national emissions vary widely among EU Member States (Figure 13, Figure 14, Figure 15, Figure 16). On a national basis, agricultural soils emit CH_4 in Italy, Spain, Portugal and Greece, where rice paddy fields cover 0.3 to 3 % of the arable land surface. All other EU Member States sink CH_4 in agricultural soils at low rates (Figure 15).

France, Germany and the UK – the countries with the largest agricultural area within the EU – account for more than half of the N_2O emissions from agricultural soils (). While France and UK emit close to the European average, Germany is the only large EU Member State in which the N_2O emissions from agricultural soils exceed the European average by 34 %. Dutch agricultural soils show by far the highest land use intensities, which exceed the European average N_2O fluxes more than twice.

Estimates of GHG emissions from farmed organic soils involve more uncertainty than those from mineral soils due to both the small experimental basis of emission factors and vague estimates of the area and land use type of farmed organic soils. Therefore, the numbers in Table 20 give the order of magnitude of national emissions only. The results based on the two different base data sets agree well for EU Member States with small areas of farmed organic soils. But there is a discrepancy of more than factor 2 in the GHG emissions from farmed organic soils in Denmark, Germany, Ireland, the Netherlands and UK. Probably /JRC-SAI 2000/ underestimates the areas in the patchy Pleistocene landscapes of Northern Germany. In contrast, it can be doubted whether all of the rough grazing land in Scotland and Ireland included in the /JRC-SAI 2000/ based estimate is actually drained. Although difficult to verify, the /Lappalainen 1996/ area basis seems to yield more realistic results than the /JRC-SAI 2000/ data base and will hence be pursued in this study.

Overall, the GHG emissions from agricultural soils were increasing in EU Member States and Europe as a whole from 1975 during the 1980s and peaked in 1987 (Table 19, Figure 16). In recent years, the GHG emissions declined by an average rate of 0.7 % yr^{-1}, which is less than the trend uncertainty.

Methane emissions from agricultural soils increased due to the extension of the rice paddy area in Italy and France during the late 1980´s (Table 19, Figure 16). Nitrous oxide emissions from arable soils do not exhibit a pronounced trend in most of the countries during the last 25 years. In Germany and the UK, a reduction in fertiliser use and in livestock has inferred lower N_2O emissions from arable land during the 1990´s. This trend is likely to continue. In contrast, in France, Spain and the Netherlands, N_2O emissions from arable land have been continuously increasing. In contrast, N_2O emissions from grasslands have been decreasing through extensivation and a small reduction in grassland area in all European countries by 4 to 14 % since 1990, which is a statistically significant national and European trend.

Table 20 Greenhouse gas emissions from farmed organic soils

	Areas from /JRC-SAI 2000/ Tg yr^{-1} CO_2-equivalents	Areas from /Lappalainen 1996/ Tg yr^{-1} CO_2-equivalents
Austria	0 ± 0	0 ± 0
Belgium	0 ± 0	0.4 ± 0.2
Denmark	10 ± 7	3.3 ± 2.1
Finland	not reported	5.6 ± 3.3
France	0.9 ± 0.6	0.8 ± 0.5
Germany	4.2 ± 2.7	17 ± 11
Greece	0 ± 0	0 ± 0
Ireland	12 ± 8	1.6 ± 1.0
Italy	0.4 ± 0.3	0.5 ± 0.3
Luxembourg	0 ± 0	0 ± 0
Netherlands	12 ± 7	3.5 ± 2.3
Portugal	0 ± 0	0 ± 0
Spain	0 ± 0	0.3 ± 0.2
Sweden	not reported	5.1 ± 3.2
UK	25 ± 16	11 ± 7
EU-15	**73 ± 47**	**52 ± 33**

5.4.2 Animal husbandry

Regional distribution

In animal husbandry, the animal type determines the average rates of CO_2-equivalents emissions rather than the management system when the emissions are normalised by livestock units (lu) (Figure 17). Ruminants emit about 3 Mg lu^{-1} yr^{-1} CO_2-equivalents, swine 1.5 Mg lu^{-1} yr^{-1} and poultry 0.5 Mg lu^{-1} yr^{-1}. Ruminants and swine emit 4 to 5 times more CO_2-equivalents as CH_4 than as N_2O while the ratio is balanced for poultry. Slurry-based systems tend to emit more GHG than FYM-based and pasture systems. This trend is caused by a clear increase in CH_4 emissions from manure in the order pasture < FYM < slurry. The opposite order is found for N_2O, but with less pronounced differences between the manure management systems.

The CH_4 emissions strongly correlate with cattle numbers. The N_2O and CH_4 emissions also reflect the animal management systems (Figure 17), which differ considerably among EU Member States (Figure 18). In France, for instance, it is estimated that between 54 and 72 % of the cattle excreta are dropped during grazing depending on the region, while the average rate for Germany is 8 % only. Due to the differing management and to the varying importance of ruminants versus non-ruminants, animal husbandry emits 8 times more CH_4 than N_2O in Denmark, but only 1.2 times more in Greece. The EU average ratio is 3.6.

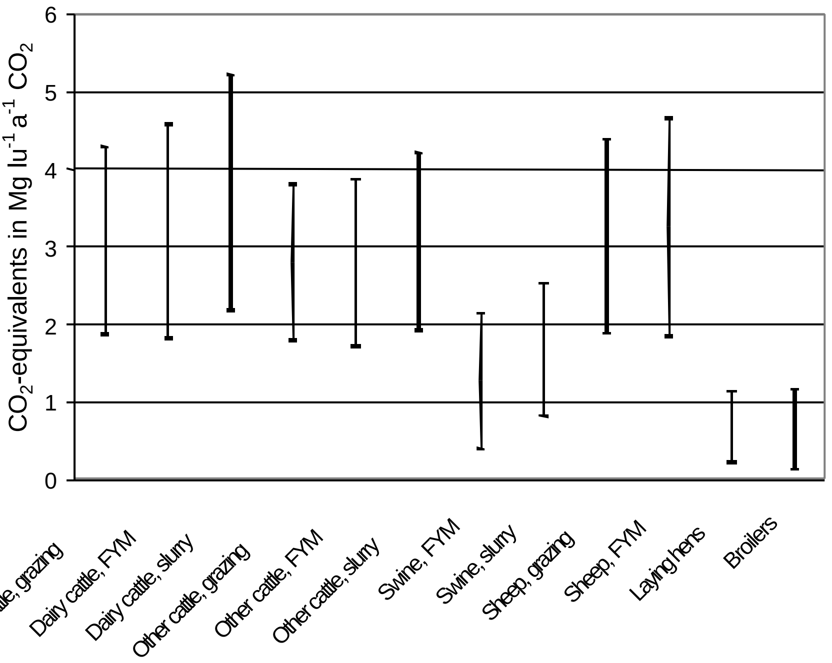

Figure 17 Average emissions of CO_2-equivalents emissions per livestock unit in Europe (Conversion factors from animal place to livestock unit (lu): Dairy cattle = 1 lu $head^{-1}$; other cattle = 0.8 lu $head^{-1}$; swine = 0.16 lu $(animal\ place)^{-1}$; sheep = 0.08 lu $(animal\ place)^{-1}$; laying hens = 0.02 lu $(animal\ place)^{-1}$; broilers = 0.01 lu $(animal\ place)^{-1}$)

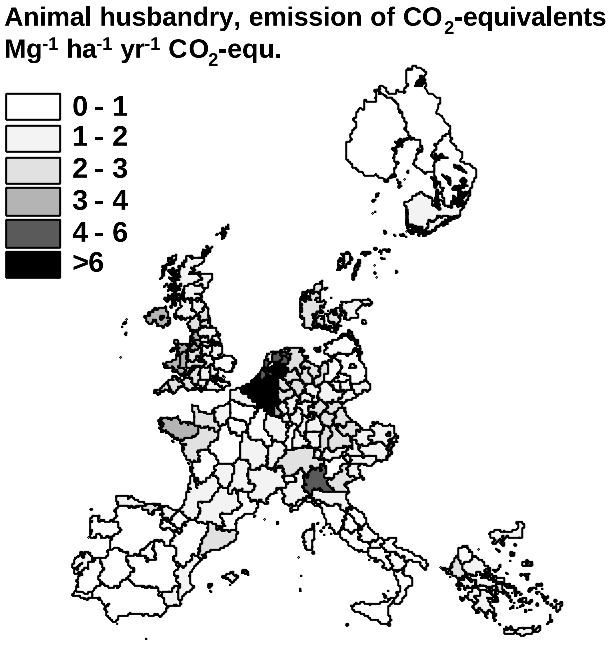

Figure 18 Regional distribution of average emissions of CO_2-equivalents from animal husbandry, normalized by the area of agricultural land in the NUTS 3 regions [Mg ha^{-1} yr^{-1} CO_2-equivalents]

Animal husbandry has the highest emission densities (>10 Mg ha^{-1} yr^{-1} CO_2-equivalents) in South Netherlands and the Flemish part of Belgium. Emission densities around 6 Mg ha^{-1} yr^{-1} CO_2-equivalents characterise regions of concentrated grazing cattle or sheep such as in Ireland, Scotland, Lower Saxony, Bavaria and the Lumbardy. Emission densities from animal husbandry are low in Boreal and Mediterranean climate regions. In average, related to the agricultural area, animal husbandry in Europe emits 2.3 Mg ha^{-1} yr^{-1} CO_2-equivalents with a wide range from 0.15 to 12.1 Mg ha^{-1} yr^{-1} CO_2-equivalents.

National greenhouse gas emissions and temporal trends

In terms of CO_2-equivalents, the biogenic GHG emissions from animal husbandry in the EU are composed of 79 % CH_4 and 21 % N_2O. Methane is released from enteric fermentation and manure decomposition in animal houses, manure storage and to small amounts during manure spreading. Nitrous oxide is emitted from manure decomposition in animal houses and manure storage and from animal droppings on pasture (Table 21). Manure spreading is included in the N_2O emissions from agricultural soils, since the manure is used as a fertiliser.

Again, France, Germany and the UK together share half of the CH_4 and N_2O emissions from animal husbandry in Europe (Figure 19, Figure 20).

70 % of the CH_4 emissions originate from the cattle sector, and 60 % from enteric fermentation. Hence, enteric fermentation is by far the largest source of CH_4 (Table 21), which is an order of magnitude larger than animal houses and manure storage, which are together twice as high as grazing, which in turn exceeds manure spreading by two orders of magnitude. The national and European emissions directly relate to the herd size and herd management. The emissions and their trends reflect the number of animals and the distribution of age classes among the national herd, mainly of cattle for enteric fermentation, and of cattle, swine and poultry for animal housing, manure storage and spreading. In Europe, N_2O emissions from grazing exceed those from managed manure since grazing ruminants dominate in the Atlantic temperate and Mediterranean Member States. About twice as much N_2O originates from animal droppings on pasture than from manure management of housed animals, but in total, in terms of CO_2-equivalents, N_2O from manure is less important than CH_4.

Methane emissions from animal husbandry decreased significantly in EU15 by 11 % between 1975 and 1997 (Table 21, Figure 19). This reflects a general slight decrease in CH_4 emissions in all EU Member States, but particularly the decline in the dairy sector in the late 1980s and early 1990s. In the same period, N_2O emissions in Europe were constant since emission reductions in the cattle sector were compensated by increased emissions from swine and poultry housing and sheep/goats grazing. It can be assumed that animal performance and animal size have increased during the last 25 years. This pattern has been excluded here, so the reduction in CH_4 emissions as a consequence of lower animal numbers will be smaller in reality.

Emissions from sheep and goats are of minor importance in most European countries since 40 % of the EU´s sheep are kept in the UK. Trends vary among Member States, with increases in Italy and a reduction in Germany.

Swine numbers, and hence GHG emissions, in the European Community have been on a more or less constant level slightly above 110 millions of swine since 1980. Rapid increases in the Belgium, Danish and French herd between 1987 and 1993 were compensated by a reduction of the German numbers.

Poultry is the smallest source of GHG in animal husbandry (Table 22). Poultry numbers were first constant, but have then increased since the early 1990s in many EU countries.

The decrease in CH_4 emissions from animal husbandry between 1990 and 1997 is significant in Germany and the EU, as is the N_2O decrease in Germany.

Table 21 Greenhouse gas emissions from animal husbandry (AH) in the European Union (EU-15).

Source	**1995 CH_4 Gg yr^{-1}**	**1995 N_2O Gg yr^{-1}**	**1995 CO_2-equiv. Tg yr^{-1}**	**% of emission from AH**	**Change 1975-1990**	**Change 1990-1997**
				CO_2-equiv.	**CO_2-equiv.**	**CO_2-equiv.**
Enteric fermentation	6048 ± 1839		127 ±39	59 %	-9 %*	-7 %*
Animal houses	669 ± 341	35.9 ±29.0	25 ±12	12 %	+11 %*	-2 %
Manure storage	1158 ± 537	13.4 ±11.2	28 ±13	13 %	+10 %	-5 %
Manure spreading	3 ± 1	(soils)	0.06 ±0.02	0 %		
Grazing	0 ± 0	100.2 ±50.7	31 ±16	15 %	+4 %	0 %
Total AH	**7979 ± 1944**	**150 ± 60**	**214 ±51**	**100 %**	**-4 %**	**-6 %***

* significant trend

Table 22 Biogenic GHG emissions from agriculture by source category in 1995

Emission [Tg CO2-equivalents]	**470**
Contribution	**100 %**
Agricultural soils	55 %
Mineral soils	43 %
Organic soils	11 %
Rice paddies	1 %
Animal husbandry	45 %
Dairy cattle	14 %
Other cattle	18 %
Sheep/goats	7 %
Swine	5 %
Poultry	2 %

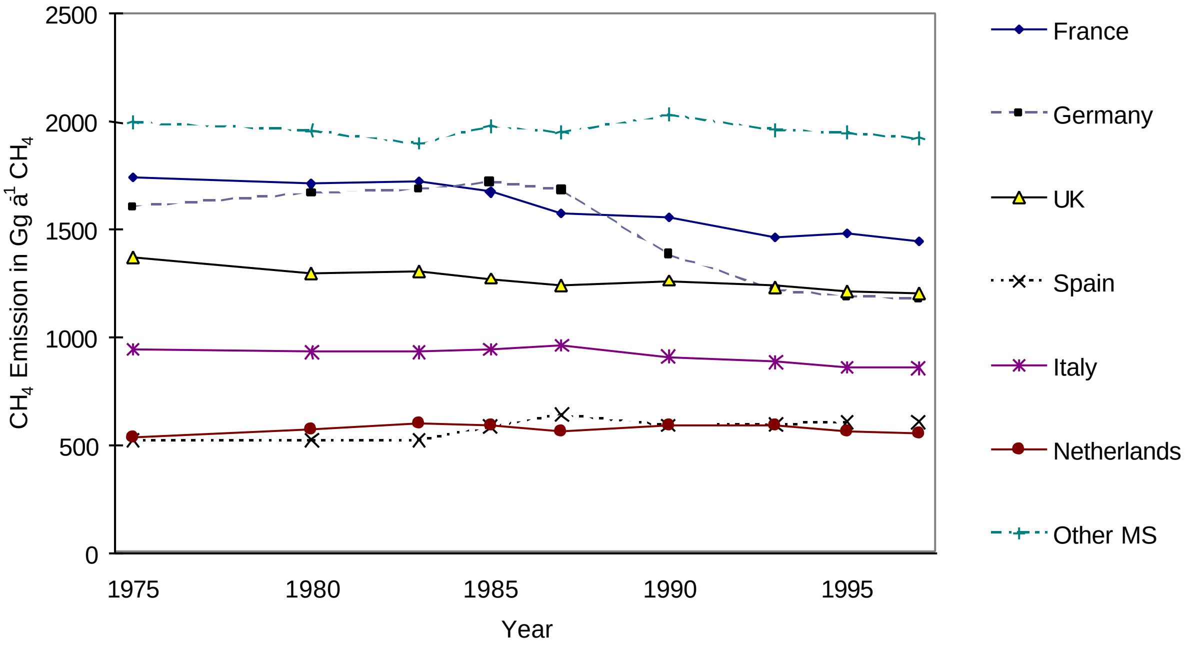

Figure 19 Methane emissions from animal husbandry (Average uncertainty: France 26 %; Germany 24 %; UK 27 %; Spain 24 %; Italy 25 %; Netherlands 24 %; EU average 25 %); MS Member States

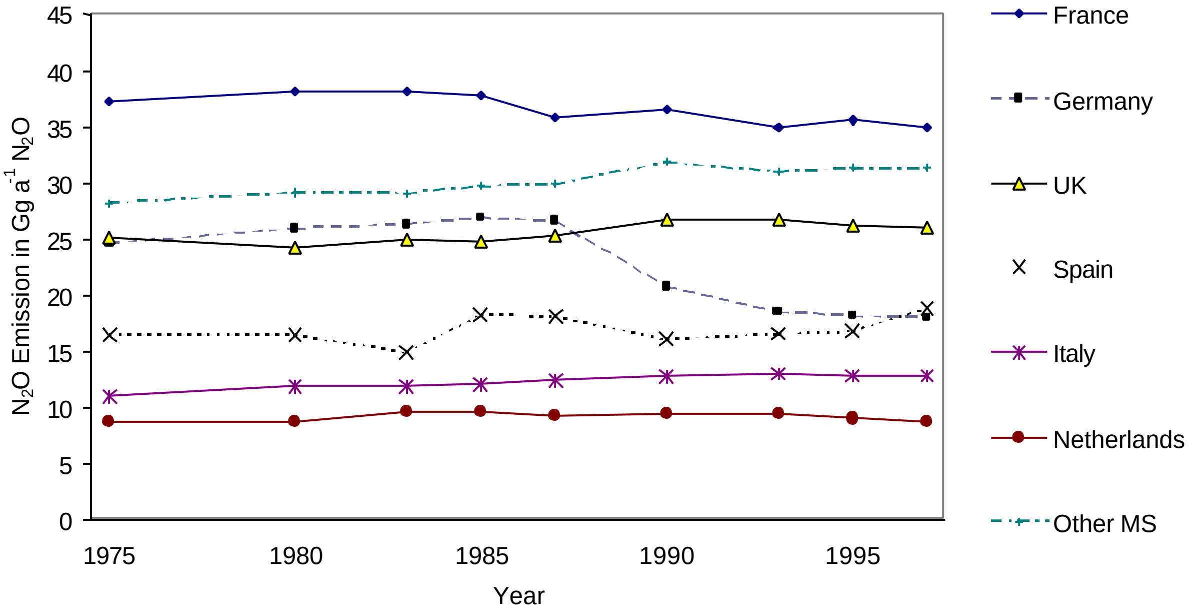

Figure 20 Nitrous oxide emissions from animal husbandry (Average uncertainty: France 43 %; Germany 43 %; UK 41 %; Spain 41 %; Italy 40 %; Netherlands 48 %; EU average 40 %); MS Member States

5.4.3 Entire European agriculture and contribution to anthropogenic emissions

Figure 18 and Figure 21 display the regional distribution of average GHG emissions from animal husbandry and agricultural soils, normalised by the area of agricultural land in the NUTS regions. Evidently, the fluxes from both agricultural sectors go in parallel and add up to an average flux of 4.4 Mg ha^{-1} yr^{-1} CO_2-equivalents. The highest emission densities arise from the Netherlands and Belgium due to intensive animal husbandry. Medium fluxes (3 to 5 Mg ha^{-1} yr^{-1} CO_2-equivalents) characterise most of the relatively intensive agricultural areas of Europe, such as North-western France, Denmark, North-western and Southern Germany, Northern Italy, Northern Spain, Switzerland, and the animal regions of the UK. Moderately low average emissions arise from the Mediterranean, and most of France, Sweden and Finland.

In terms of CO_2-equivalents, the biogenic GHG emissions from agriculture in the EU consist of 49 % N_2O, 41 % CH_4 and 10 % CO_2. Mineral agricultural soils emit the largest share of GHG, cattle represent the second largest source of biogenic GHG emissions in agriculture (Table 22). A significant proportion also originates from farmed organic soils. The trace gas species show typical patterns: On agricultural soils, N_2O dominates, except for rice paddies. For animals, CH_4 dominates the trace gases, especially for ruminants (Table 22).

The official national inventories report N_2O emissions, CH_4 emissions, CO_2 emissions and CO_2 removals by land use change and forestry (Ritter, 1999). In the following, the net CO_2 emissions (emissions minus removals) will be used as reference.

Agriculture contributes to the anthropogenic CH_4 emissions between 20 and close to 100 % of the total anthropogenic GHG emissions in EU Member States (Figure 22; EU15: 42 to 47 %). The share of agriculture in anthropogenic N_2O is generally larger (EU15: 70 %). The numbers calculated here even exceed the numbers reported as the overall emissions in some countries (Figure 22). This may be partly explained by the fact that N_2O emissions from manure management have not been included in the respective national inventories (Ritter, 1999). This finding clearly demands an improvement of the national official reports. The anthropogenic CO_2 emissions are mainly driven by fossil fuel consumption, so agriculture is a relatively small source in countries with significant areas of farmed organic soils and negligible in other countries (EU15: 1 %). In terms of CO_2-equivalents, agriculture emits between 7 and 50 % of the anthropogenic GHGs in EU Member States (EU15: 11 %).

Agriculture had a share of 13 % of the overall reduction in GHGs between 1990 and 1997, which is more or less equivalent to its share in total emissions.

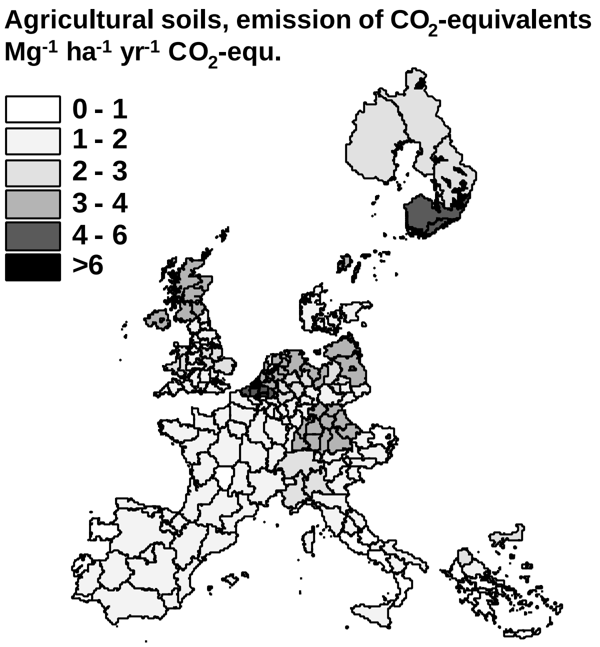

Figure 21 Regional distribution of average emissions of CO_2-equivalents from soils, normalized by the area of agricultural land in the NUTS 3 regions [Mg ha^{-1} yr^{-1} CO_2-equivalents]

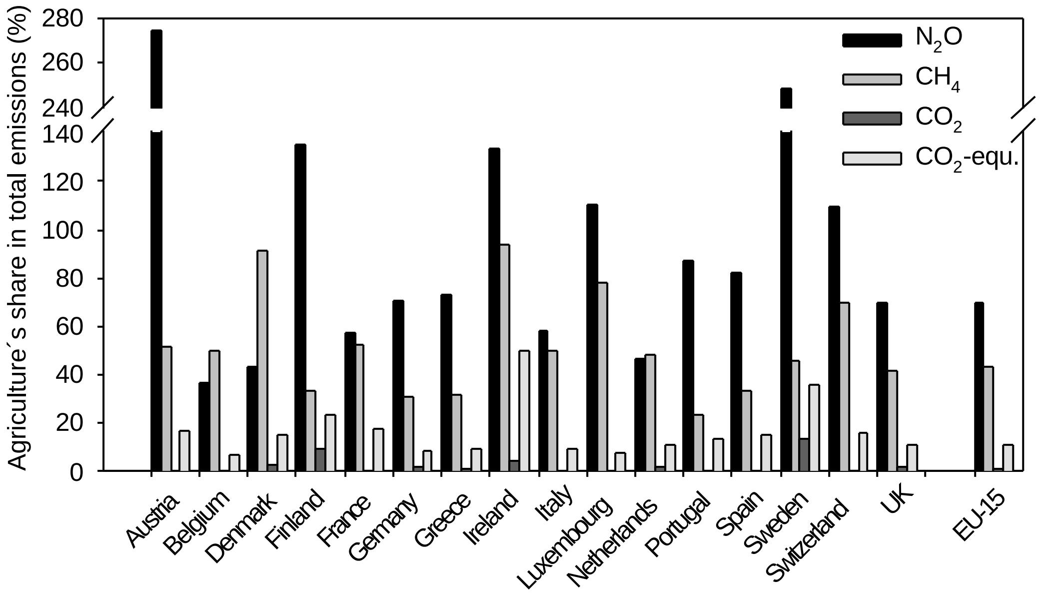

Figure 22 Contribution of agriculture to total anthropogenic emissions in 1995 (Values above 100 % identify underestimates in the national submissions to UNFCCC /Ritter 1999/)

5.5 Discussion

The following analyses quantify the uncertainties associated with the agricultural emission inventories at EU and Member States level. The comparison of the results gathered with the Europe-specific approach pursued here with more general inventories gives insight in how much different assumptions and methods alter the national agricultural emission inventories. In order to have an independent quality check, the results are verified against large-scale atmospheric observations and inverse models. The atmosphere records the net exchange of sinks and sources on a large spatial scale, so this verification pertains to the overall anthropogenic emissions, including non-agricultural sources as reported in /Ritter 1999/. However, the isotopic composition of carbon and oxygen in trace gases of the atmosphere allows to some extent a discrimination of sources /Levin et al. 1999; Perez et al. 2001/.

5.5.1 Uncertainties in the results

Not all sources of uncertainty can be quantified. Uncertainty due to lack of adequate knowledge of processes and dependencies between emissions and their natural and human controls inherently limits the adequacy and accuracy of any model approach. However, the magnitude of this source of uncertainty is unknown. The verification of the inventory results against results obtained by independent measurements and methods such as inverse atmospheric models (Section 5.5.3) at least rules out that the inventory results are systematically biased by limited process understanding.

Another source of uncertainty is directly associated with the emission models, emission factors and data. Inherent uncertainty in the emission models and emission factors results from the interannual variability of weather, unresolved small-scale spatial variability of biological processes and soil conditions, unaccounted diversity in animal housing and manure management, and unconsidered driving forces of emissions. These unaccounted explanatory variables produce a wide scatter of measured versus modelled trace gas fluxes from agricultural sources, which translates in relatively large standard errors. In comparison, uncertainties associated with the precision and accuracy of measurement methods and with data from agricultural census are generally small and well known. The uncertainties in emission models, emission factors and resulting inventories of emissions from source categories, of national inventories, of the European inventory and of time trends are analysed by error propagation according to the IPCC Tier 1 methodology of the IPCC Good Practice Guidelines for Greenhouse Gas inventories /Penman et al., 2000/.

Table 23 Sources of uncertainty in national inventories for 1995 according to Tier 1 of IPCC Good Practice Guidance (Penman et al., 2000)

Source	**U(IPCC)**	**U(EF)**	**U(Data)**	**A**	**BE**	**D**	**DK**	**E**	**FIN**	**F**	**GR**	**IE**	**IT**	**NL**	**P**	**S**	**UK**	**EU**	**EU IPCC**
	%[a]	**%**[b]	**%**[c]	**%**[c]	**%**[c]	**%**[c]	**%**[c]	**%**[c]	**%**[c]	**%**[c]	**%**[c]	**%**[c]	**%**[c]	**%**[c]	**%**[c]	**%**[c]	**%**[c]	**%**[c]	**%**[c]
CH_4 Soil sink	n.c.	100	5	0	0	0	0	1	0	0	0	0	0	0	0	0	0	**0**	**0**
CH_4 Paddy soils	21	21	5	0	0	0	0	0	0	0	0	0	1	0	1	0	0	**0**	**0**
N_2O Arable soils	75[d]	41-64	5	30	6	15	7	22	31	14	14	5	18	4	18	18	8	**12**	**35**[d]
N_2O Grassland soils		29	5	8	4	5	2	11	1	6	3	8	5	5	7	2	6	**6**	
N_2O Organic soils	100	43	40	0	1	3	2	0	6	0	5	1	0	2	1	6	3	**2**	**1**
CO_2 Organic soils	n.c.	50	40	0	2	11	8	0	23	0	8	3	1	7	3	19	10	**6**	
SUM Soils				**30**	**7**	**20**	**11**	**25**	**39**	**16**	**18**	**10**	**19**	**10**	**20**	**27**	**14**	**15**	**35**
CH_4 Enteric fermentation	30	30	5	12	12	7	7	10	4	11	9	14	11	8	11	5	10	**10**	**10**
CH_4 Animal housing		50	21	3	4	2	4	2	1	1	1	1	2	3	3	1	1	**2**	
CH_4 Manure storage	50	45	21	2	6	3	6	3	1	2	3	2	4	7	3	1	2	**3**	**5**
CH_4 Manure spreading	n.c.	20	21	0	0	0	0	0	0	0	0	0	0	0	0	0	0	**0**	
N_2O Animal housing		80	21	2	3	2	2	3	1	1	2	1	3	4	4	1	2	**2**	
N_2O Manure storage	50	80	21	2	1	1	1	0	0	1	0	1	1	1	1	1	0	**1**	**5**
N_2O Grazing	80	50	21	0	3	1	1	7	1	7	15	4	3	3	6	2	5	**4**	
SUM Animal husbandry				**13**	**15**	**8**	**10**	**13**	**4**	**13**	**18**	**15**	**13**	**12**	**14**	**6**	**11**	**11**	**13**
TOTAL AGRICULTURE				**33**	**17**	**21**	**15**	**28**	**39**	**20**	**25**	**18**	**23**	**15**	**24**	**28**	**18**	**19**	**38**

[a] Uncertainty in emission factors of IPCC Guidelines (IPCC, 1997) in % of emission factor value derived from Lim et al. (1999)

[b] Uncertainty in emission factors and data in this study in % of emission factor value or activity data values

[c] Uncertainty in inventory as % of total agricultural greenhouse gas emissions in 1995

[d] includes grasslands

n.b. not considered

The emission factors bear larger uncertainty *U* (Table 23) than the data derived from agricultural census and statistics. However, the uncertainty of the emission factors developed here is lower than in the IPCC (1997) methodology except for N_2O emissions from manure storage where, however, more source categories than in IPCC (1997) are distinguished and the uncertainty refers to the scatter in real measurements. As a consequence, while considering more sources and source categories than IPCC (1997), the methodology developed here reduces the uncertainty in most source categories and in the national and European inventories to half of the uncertainty of inventories based on the IPCC (1997) methodology.

As shown in Table 19 and Table 21, the GHG inventories for agricultural soils tend to have larger uncertainties than those for animal husbandry because enteric fermentation, the largest GHG source from animals, is relatively well constrained. The inventories of agricultural greenhouse gas emissions in EU Member States are still associated with an uncertainty of 15 to 39 % (Table 23), *i.e.* 7 to 39 % from soil emissions and 4 to 17 % from animal husbandry, depending on the relative importance of the source categories in the national context. Methane emissions from enteric fermentation, N_2O emissions from mineral arable and grassland soils and N_2O and CO_2 emissions from farmed organic soils – the largest national sources – contribute most to the overall uncertainty. The following main sources of uncertainty in national inventories are identified:

- Emission models for N_2O from mineral agricultural soils;
- Emission factors for N_2O and CO_2 from farmed organic soils;
- Soil properties in soil maps since topsoil carbon, nitrogen and sand contents are obtained from the Soil Geographical Database as classified values only;
- Areas of farmed organic soils and their respective use;
- Allocation of animal manure to management categories. Here, the data is improving in the context of the reduction of ammonia emissions from animal husbandry in Europe.

The magnitude of the uncertainty in the inventories is larger than the one in the trends of annual emissions if assumed that the uncertainties in emission factors are temporally correlated because they remain constant while the uncertainties in the activity data are not /Penman et al. 2000/. Consequently, the uncertainties in the trends depend more on those of the activity data than of the emission factors. The trends from 1975 to 1990 and from 1990 to 1997 in national inventories (Table 19, Table 21) are associated with uncertainties of 7%, 5 % and 6 % for emissions from soils, animal husbandry and total agriculture, respectively.

Further improvement must reduce 1) uncertainties in the emission calculations by additional long-term measurements under full scale and field conditions, 2) statistical uncertainties due to the natural variability in the emissions through use of process-based models, and 3) uncertainties in the emission generating activities, especially in the distribution of animal housing and manure storage types and in the area and use of farmed organic soils. These constraints all result from lack of adequate data, which should be provided in the near future by intensified long-term observations and detailed farm surveys.

5.5.2 Comparison with other agricultural inventories of GHG emissions

Greenhouse gas emissions from agriculture are calculated here with a detailed, Europe-specific and regionally disaggregated approach. Hence a comparison between the results of this study and other inventories based on more general approaches such as /IPCC 1997/ identifies the implications of taking different assumptions, emission factors and scope and detects regional specificities unresolved by the more general approaches.

At the European level, the N_2O emission estimates for agriculture in this study exceed those in the official national inventories submitted under the UNFCCC /Ritter 1999/ by 37 % (Table 24). The differences on national level generally are less than 40 %, which still ranges in the uncertainty band of the national inventories. However, the N_2O emissions calculated here for 1995 exceed those given by /Ritter 1999/ for Austria, Finland, Ireland, Portugal and Sweden by a factor of 2 or more. The extreme outlier of Sweden (factor 114) is caused by an unrealistically small number given by /Ritter 1999/, which has been corrected in the 1997 inventory to fit our results. In contrast, the estimate in this study shows lower N_2O emissions in Denmark. Evidently, the data reported by /Ritter 1999/ based on IPCC (1997) cannot resolve the regional soilborne N_2O emissions found in this study. In addition, N_2O release from manure management are underestimated or not reported (/Ritter 1999/ and CORINAIR (http://www.aeat.co.uk/netcen/corinair/) (Table 24) and farmed organic soils presumably are not considered as a source of N_2O.

The agreement in the CH_4 emission estimates for agriculture in this study and the study of /Ritter 1999/ is excellent on the EU level (-9 %) and generally within 20 % for most EU Member States, and hence lies within the range of uncertainty of both inventories. Our results for Greece, Spain and Sweden are 30 to 50 % lower than those of /Ritter 1999/.

Carbon dioxide emissions from farmed organic soils are so far missing in official national inventories.

To conclude, inventories using the IPCC default methodology (IPCC, 1997) reveal adequate results at the European scale, but N_2O and CO_2 data loose some credibility at the national scale.

National total anthropogenic CH_4 emissions from selected EU Member States (Table 25) show in general a good agreement among a set of inventories. Significant differences found for Germany and the Netherlands in the databases reported by /Nisbet et al. 1998/ cannot be explained because the source categories have not been specified in the latter study. A clearer insight in the data would be needed in order to identify what source category produces the difference. However, the agricultural CH_4 emissions seem to be reasonably well quantified, and inconsistencies in national total anthropogenic emissions are dominantly located in non-agricultural sources /Nisbet et al. 1998; Janssen et al. 1999; Levin et al. 1999; Lim et al. 1999/.

Table 24 Comparison of national agricultural N_2O emissions in 1995 with other inventories of selected EU Member States

	This study	**CORINAIR**[a]	**National communications** /Ritter 1999/
	Tg yr^{-1} N_2O	**Tg yr^{-1} N_2O**	**Tg yr^{-1} N_2O**
Belgium	0.012 ± 0.003	0.012	0.010
Denmark	0.018 ± 0.006	0.009[b]	0.030[c] *
France	0.14 ± 0.04	0.08[b]	0.17
Germany	0.13 ± 0.04	0.09	0.08
Ireland	0.035 ± 0.011	0.020[b]	0.019[c]
Netherlands	0.033 ± 0.010	0.025[b]	0.026
UK	0.11 ± 0.02	0.01*	0.098
EU-15	0.65 ± 0.16	0.44	0.61

[a] available at http://www.aeat.co.uk/netcen/corinair/

[b] including managed forests, but manure management not reported

[c] agricultural soil only

* significantly different value

Table 25 Comparison of national total CH_4 emissions with other inventories of selected EU Member States

	This study[a]	**CORINAIR**[b]	**National communications** /Ritter 1999/	**COMET database** /Nisbet et al. 1998/	**LOTOS database** /Nisbet et al. 1998/	**U.C.G.** /Nisbet et al. 1998/
	Tg yr^{-1} CH_4	**Tg yr^{-1} CH_4**	**Tg yr^{-1} CH_4**	**Tg yr^{-1} CH_4**	**Tg yr^{-1} CH_4**	**Tg yr^{-1} CH_4**
Year	**1995**	**1994**	**1995**	**n. r.**	**n. r.**	**1993**
Belgium	0.54 ± 0.16	0.42	0.59	0.59	0.63	0.37
Denmark	0.36 ± 0.11	0.43	0.43	0.37	0.31	0.31
France	2.70 ± 0.81	2.84	2.79	3.10	3.30	2.20
Germany	3.53 ± 1.06	4.85	3.90	7.00*	3.80	3.40
Ireland	0.77 ± 0.23	0.81	0.81	0.65	0.58	1.00
Netherlands	1.27 ± 0.28	1.08	1.17	1.10	0.80*	0.63*
UK	3.91 ± 1.18	3.85	3.75	3.80	2.30	2.90

[a] anthropogenic emissions, calculated as the sum of agricultural emissions in this study and the non-agricultural emissions as given in /Ritter 1999/

[b] available at http://www.aeat.co.uk/netcen/corinair/

n. r. not reported

* significantly different from results in this study

Verification against atmospheric measurements and models

The inversion of atmospheric transport models (Table 26) offers an independent top-down validation of the bottom-up inventories of total anthropogenic emissions. /Hensen et al. 1999/ estimated the N_2O emissions from the Netherlands by measurements of atmospheric N_2O concentrations at a 200 m tower at Cabaauw, the Netherlands, in nights with inversion, through backward trajectory calculation at 31±13 Gg N_2O-N yr^{-1}. This is lower but not significantly different from the overall emission of 49±11 Gg N_2O-N yr^{-1} calculated in this study.

Based on atmospheric concentration measurements at Mace Head, Ireland, and a climatological model, /Derwent et al. 1998/ estimated the average source strength of GHG emissions in the UK between 1987 and 1996. The results agree well with the average calculated overall anthropogenic emissions in this study for both CH_4 (this study vs. /Derwent et al. 1998/: 77±39 vs. 64±16 kg CH_4 (population head)$^{-1}$ yr^{-1} and 3.8±1.4 vs. 3.6 kg N_2O (population head)$^{-1}$ yr^{-1}).

/Levin et al. 1999/ validated the CH_4 emissions by source categories for a region around Heidelberg, Germany, by continuous atmospheric CH_4 concentration measurements combined with $\delta^{13}C$-CH_4 and atmospheric ^{222}Rn observation with an uncertainty range of 20 to 30 %. The validation of CH_4 emissions from cattle, however, was not entirely independent from the CORINAIR1990 data. The regional results of the approach in this study agrees with the emission budget reported in Levin et al. (1999) within less than 10 % and also follows the reported decrease of CH_4 emissions by about 15 % between 1990 and 1997.

For North-western Europe, a set of regional inverse atmospheric models is available to validate the national CH_4 inventories (Table 26). Each of the models applies to a limited area only where adequate atmospheric observations can be obtained. The COMET inverse model is based on atmospheric CH_4 records at Cabaauw tower, the Netherlands. Obviously, various inversions of the same model yield source estimates scatter by factor 2 to 12. Well constrained estimates are achieved for the Netherlands and Belgium only where the inversion is based on a high number of trajectories. There is also a wide range in the results of different inverse models (Table 26). In general, the CH_4 inventory in this study agrees fairly well with the inverse models for Belgium – although relatively low -, Germany, Netherlands and the UK while the large scatter in the inverse models prevent a meaningful validation of the CH_4 inventories for Denmark, France and Ireland. No significant bias occurs between the CH_4 inventory and the regional atmospheric inverse models once the inverse models agree with each other.

Table 26 Verification of total national anthropogenic emissions in Northwestern Europe by inverse atmospheric models

	This study	**COMET inverse**	**COMET inverse**	**COMET inverse**	**LOTOS inverse**	**EUROS Kalman**	**UCG/RHUL inverse**
		/Veltkamp et al. 1995/	**/Janssen et al. 1999/**	**/Nisbet et al. 1998/**	**/Janssen et al. 1999/**	**/Janssen et al. 1999/**	**/Nisbet et al. 1998/**
	Tg yr^{-1} CH_4	**Tg yr^{-1} CH_4**	**Tg yr^{-1} CH_4**	**Tg yr^{-1} CH_4**	**Tg yr^{-1} CH_4**	**Tg yr^{-1} CH_4**	**Tg yr^{-1} CH_4**
Year	**1995**	**1993**	**1993-1995**	**1993-1995**	**1994**	**1995**	**1995-1996**
Belgium	0.54 ± 0.16	0.91- 1.07	0.6	1.2 ± 0.6	0.63	0.70	1.0 ± 0.2
Denmark	0.36 ± 0.11	0.38 - 0. 50		1.3 ± 0.5	0.35	0.45	0.7 ± 0.2
France	2.70 ± 0.81	1.98 - 2.41	0.8	7.0 ± 3.5	0.98	1.00	7.3 ± 1.7
Germany	3.53 ± 1.06	10.4 - 11.8[a]	3.6	10.2 ± 7.0	5.73	4.50	
Ireland	0.77 ± 0.23	0.47 - 0.73		0.13 ± 0.06			0.9 ± 0.2
Netherlands	1.27 ± 0.28	0.85 - 1.01		1.5 ± 0.6			2.0 ± 0.5
UK	3.91 ± 1.18	2.21 - 3.23	4.8	5.2 ± 2.3	4.90	5.00	4.5 ± 1.0

[a] poorly constrained for South Germany

To conclude, there is fair agreement between the N_2O and CH_4 inventories on national and regional level in this study with the available atmospheric inverse models. The validation is, however, spatially constrained to North-western Europe, limited by the accuracy of the inventories and inverse atmospheric models and complicated by the fact that in most cases the gas fluxes cannot be attributed to agricultural versus non-agricultural sources and sinks.

5.6 Summary and outlook

This study has developed a detailed methodology to quantify the biogenic emissions of GHGs from European agriculture, which is based on emission factors and regional regression equations derived from all available measurements in Europe. Several innovations as compared to /IPCC 1997/ have been achieved in the sectors of both soil-borne and manure-borne emissions. The methodology hence allows a more accurate assessment of the emissions under European conditions than global default methodologies such as /IPCC 1997/, reducing the uncertainty to half of the /IPCC 1997/ approach. The overall uncertainty in the GHG inventories of agriculture in the EU Members States and the EU as a whole sums up to around 20 %. Fair agreement with the inverse atmospheric models supports the validity of the inventory. Still, further improvement is desirable, demanding for intense long-term field observations of soil-borne CO_2 and N_2O fluxes and of CH_4 and N_2O emissions from animal houses and manure management. Data on emission generating activities can be directly obtained from official census and soil maps, so this approach offers a transparent and straightforward calculation of present and future national and European agricultural GHG inventories. National bodies in Europe could adopt this detailed methodology in order to improve the national submissions under the UNFCCC and to better determine the source strengths of the various GHG sources in the agricultural production system as a basis for cost-effective and efficient GHG mitigation measures.

Agricultural N_2O and CH_4 emissions in the European Union of this study agree well with the official submission to UNFCCC /Ritter 1999/. But clearly shortcomings and gaps in the official inventories at regional and, in some cases, national scale are identified in terms of omissions and over- or underestimated soilborne emissions. For instance, in countries with – even small – areas of farmed organic soils, CO_2 emitted from peat oxidation can significantly contribute to the overall emissions, which has so far widely been neglected.

Annual emissions can be readily determined by the method applied here. However, annual estimates and average emission factors used in default methods and in this study do not offer the opportunity to monitor and evaluate the efficiency of GHG mitigation measures in agriculture since the daily variation in the driving forces of biogenic emissions is complex and demands for a sophisticated process-based modelling approach.

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6 Mitigation of greenhouse gas emissions in European agriculture

Abstract

In this study, a regionalized cause-oriented methodology was developed and applied to quantify greenhouse gas (GHG) sources and sinks in European agriculture. It reduces the uncertainty in national inventories to 20 %, i.e. halves the original value of the IPCC default methodology. The fact that the regionalized approach better fitted to measured data than the universal one, in particular with regard to N_2O *emissions from soils, suggests that efficient GHG mitigation in agriculture requests locally adjusted and site-specific complex strategies rather than uniform management recommendations. Agriculture in EU-15 produces annually 0.4 Pg* CO_2*-equivalents, i.e. 10 % of the anthropogenic GHG emissions. The assessment of technical, socio-economic and political mitigation measures clearly allocates the highest mitigation potential to biomass from perennial, best woody, plants produced for bioenergy. The substitution of fossil carbon emissions by bioenergy can potentially compensate for 0.4 to 0.8 Pg* a^{-1} CO_2*-equivalents. The potential for technical GHG reduction measures in EU agriculture does not exceed 0.1 Pg* a^{-1} CO_2*-equivalents. Promising measures promote the extensivation of arable cropping by reduced synthetic and overall nitrogen inputs and technological innovation in animal husbandry, which is best accompanied by a further decline in animal numbers. Carbon sequestration through change in land use and land management does not mitigate climate change from a scientific perspective since the carbon sink in the biosphere is highly vulnerable to any human-induced or natural forcing and can only alleviate the increase in atmospheric* CO_2 *concentration for decades rather than permanently. The carbon sequestration in agriculture is probably small until 2010 as compared to the potential of real GHG mitigation options. Efficient GHG mitigation in agriculture needs strong financial, political and educational support.*

6.1 Introduction

The implementation of the Kyoto Protocol requires adequate methods and instruments to quantify, monitor and verify greenhouse gas (GHG) emissions and their reduction /UNFCCC 1997/. Science has to provide relevant information and to develop adequate tools to quantify emissions since 1990, to quantify, monitor, and verify emission reductions during the first commitment period from 2008 to 2012 and to guarantee adequate statistical certainty to proof the fulfillment of the Kyoto commitments on project scale, national, and European scale. Furthermore, options and potential for GHG mitigation in all sectors of human society have to be identified.

Three complementary strategies can be pursued to stabilise the atmospheric GHG concentrations, which differ widely in their technical feasibility, costs, ease of monitoring and verification, ancillary effects and duration of effects in the atmosphere. These are /IPCC 1996/:

1. Reduction of anthropogenic GHG emissions
2. Carbon substitution (use of renewables to substitute fossil energy sources and materials)
3. Carbon sequestration by enhancing sinks and reservoirs

The implementation of GHG mitigation strategies again operates at three different levels:

1. Technical measures addressing individual sources
2. Systemic measures operating at industrial or farm entity scale
3. Socio-economic and political measures aimed at changing human behaviour, altering the driving forces of emissions.

Measures at the various levels imply side-effects and synergies. A successful, efficient GHG mitigation at the national or international scale will combine all sorts of strategies and implementation levels. This is true in principle for all economic sectors. However, in the following, the agricultural sector will be discussed only because there are some extra complications in terms of quantification of GHG emissions and baselines, and in terms of the potential, efficiency and permanence of GHG mitigation measures.

Agriculture represents the major source of N_2O and CH_4 in Europe and is a fundamental component of all of the three GHG mitigation strategies mentioned above. In recent years, a suite of synthesis papers has assessed the potential for mitigation of individual greenhouse gas species in agriculture at global (N_2O: /Mosier et al. 1998b/, CH_4: /Mosier et al. 1998a; Neue 1997/, CO_2: /Sauerbeck 1993; Paustian et al. 1998/) and European or national scale (N_2O: /Armstrong-Braun et al. 1996; Smith et al. 1997a; Velthof et al. 1998/, CH_4: /Lusk 1997/, CO_2: /Smith et al. 1997b, 2000/). But so far, synergies or trade-offs between different greenhouse gas species and side-effects with regard to other environmental and socio-economic goals have not been adequately addressed /Storey and McKenzie-Hedger 1997/.

Greenhouse gas mitigation in agriculture has some complications. The biogenic agricultural GHG emissions emerge from microbial processes involved in enteric fermentation in ruminants and in the degradation of organic matter in manure and in soil. The flux rates vary widely in space and time. Farm management contributes to the emissions, but the ultimate driving forces of the microbial processes – substrate and oxygen availability, moisture and temperature – are intimately linked to climate, weather, site properties /Smith et al. 1998b; Skiba and Smith 2000/ and land use history /Mosier et al. 1998b/, many factors which can be hardly controlled by the farmer. This reduces the scope for GHG mitigation in agriculture and also makes a reliable prediction of both the annual baseline emissions and the reduction efficiency difficult.

Therefore, this chapter aims to analyse essential components of an efficient and effective GHG mitigation strategy in European agriculture. As a basis, this chapter compiles all findings of the preceding chapters in order to discuss comprehensively their implications for agriculture in the European Union (EU). Based upon the detailed assessment of agricultural GHG emissions, promising mitigation options are identified, discussed with regard to the uncertainty associated with them and assessed in terms of their environmental impact and socio-economic feedback as far as information can be drawn from literature. Finally, the mitigation of climate change is discussed in the direct context of climate and agricultural policy.

6.2 Present greenhouse gas emissions from European agriculture

The adequate quantification of GHG sources and sinks is an indispensable prerequisite to any mitigation effort. The exact location of sources and sinks must be adequately characterised and underlying processes, and emission producing mechanisms must be understood so that the effects of mitigation measures can be predicted. Both location and processes are readily known for most industrial sources, but agriculture still poses a challenge to those who wish to identify, quantify and reduce GHG emissions and increase sinks. This is because agricultural trace gas fluxes have a biogenic origin, so the actors and driving forces producing and consuming trace gases act at a microscale, depend highly on local climatic, edaphic and biological conditions and are widely dispersed in space. The present national GHG inventories rely mostly on the IPCC-Guidelines /IPCC 1997/ and involve a methodological uncertainty of close to 40 % in the agricultural emission estimates. Therefore, before the mitigation of GHG emissions in agriculture becomes a feasible option, the uncertainties in the quantification of GHG fluxes need to be significantly reduced. This section synthesizes the Chapters 4 and 5, which provide a clear improvement in terms of location of sources and accuracy of national and European GHG inventories.

6.2.1 Driving forces and regional patterns of soilborne emissions

As proven in Chapter 4, the uncertainty in the quantification of N_2O emissions from agricultural soils on national and European scale is significantly reduced by improving the methodology for the inventory calculation. It is important to determine at sub-national scale the climatological, pedological, and management factors controlling the annual N_2O release.

As a methodology, statistical models were set up on the basis of a stepwise multiple linear regression analysis of annual N_2O emissions and combinations of various soil, climate, and management variables. The data were derived from measurements across the EU reported in literature. Each of the biogeographic regions of Europe was analysed separately (Chapter 4).

The mean annual N_2O emissions from arable soils increase from South to North-East according to the biogeographic regions of Europe. This points to differences in background emissions as well as to potential differences in emission controls. In arable soils, the magnitude of annual N_2O emissions depends rather on the climate and site conditions than on recent fertilisation. Soil moisture, topsoil nitrogen content, and annual fertiliser application turned out as the best descriptors of N_2O emissions from arable soils at the European scale. Climate and soil effects are less clear in grasslands (Figure 13).

The methodology accounts for climate and soil controls of N_2O release in addition to fertilisation and can relatively easily be integrated in a standard methodology for national greenhouse gas inventories. It represents a further step towards the description of N_2O emissions on a more detailed regional scale although the uncertainty in the models still range between 30 and 40 %.

The fact that the regionalized approach to an inventory of N_2O emissions from agricultural mineral soils in Europe was more successful than the universal one of /IPCC 1997/ points to the direction which N_2O mitigation strategies must head for. The complex interaction of climate, soil and management variables controlling the N_2O exchange between the agroecosystem and the atmosphere limit the farmer´s ability to reduce N_2O emissions from the largest source in Europe. Evidently, an efficient N_2O mitigation requests for regional and site-specific complex strategies rather than for uniform management recommendations.

6.2.2 Greenhouse gas inventory for entire agriculture

Agriculture is both, a source and sink of greenhouse gases (GHGs). The emissions originate mainly from microbial processes in the rumen of animals, in manure and soil. Also the production of farm inputs, mechanical work on the farm and food and feed processing release GHGs. However, IPCC /IPCC 1997/ assigns only the biogenic sources to the agricultural sector. For comparability reasons, the same definition is applied here. In terms of CO_2-equivalents, agricultural biogenic sources contribute between 7 and 50 % to the anthropogenic GHG emissions in EU Member States (EU-15: 11 %) /Freibauer, accepted; Chapter 5/.

The cause-oriented methodology developed and applied in Chapter 5 allows a detailed quantification of GHG sources in the various steps of the agricultural production chain and a more management- and animal-specific estimation of the share of the agricultural sub-sectors than /IPCC 1997/. Specifically, as innovation, manure management is separated into the CH_4 and N_2O sources “animal houses”, “manure storage”, and “manure spreading”. For soils, the CH_4 sink is included and N_2O emissions are calculated on a regional basis with respect to climate, site conditions and management. As far as possible, indirect GHGs such as NO and NH_3 are included as well. The uncertainty in the national and European inventory is significantly reduced for GHG emissions from agricultural soils. In the animal husbandry sector, more source categories and animal types are separated, but lacking measured data prevent a significant improvement of the precision in the estimate.

The resulting inventory of European GHG emissions from agriculture is shown in Table 27. In terms of CO_2-equivalents, the biogenic GHG emissions from agriculture in the EU consist of 56 % N_2O, 36 % CH_4 and 8 % CO_2. Cattle and mineral arable soils represent the largest sources of biogenic GHG emissions in agriculture. A significant portion also originates from mineral grasslands and farmed organic soils. The trace gas species show typical patterns. On agricultural soils, N_2O dominates, except for rice paddies. For animals, CH_4 dominates the trace gases, especially for ruminants.

The inventory identifies regions in which agricultural GHG emissions are high and hence, where GHG mitigation is most promising: The highest emission densities are located in parts of the Netherlands, Belgium, South Germany, in the Po Valley, and the moist agricultural and sheep grazing areas of Ireland. Medium to high flux densities characterise most of the relatively intensive agricultural areas of Europe, such as North Western France, Denmark, Northern Germany, Northern Italy, and the moist agricultural and sheep grazing areas of Scotland, Wales and Southern Finland.

Assuming that GHG mitigation will work most efficiently if it addresses the largest sources, then the following priorities evolve from the inventory (Table 27):

- N_2O: Agricultural soils, grazing animals
- CH_4: Enteric fermentation of ruminants, manure storage
- CO_2: Farmed organic soils

Table 27 Biogenic greenhouse gas emissions from agriculture in the European Union (EU-15) in 1995 /Freibauer, accepted; Chapter 5/

EU15	**CH_4 Gg a^{-1}**	**N_2O Gg a^{-1}**	**CO_2 Tg a^{-1}**	**CO_2-equiv. Tg a^{-1}**	**% of agric. emission**
Arable soils	110 ± 84	391 ± 164		126 ±46	27 ±%
Grassland soils	-22 ± 22	259 ± 75		80 ±26	17 ±%
Farmed organic soils		42 ± 25	39 ± 25	52 ± 33	11 ±%
Total Soils	**88 ± 67**	**694 ±182**	**39 ± 25**	**256 ±61**	**55 %**
Enteric fermentation	6048 ± 1839			127 ±39	27 %
Animal houses	669 ± 341	36 ± 29		25 ±12	5 %
Manure storage	1158 ± 537	13 ±11		28 ±13	6 %
Manure spreading	3 ± 1	(soils)		0.06 ±0.02	0 %
Grazing	0 ± 0	100 ± 51		31 ±16	7 %
Total Animal Husbandry	**7979 ± 1944**	**150 ± 60**	**0**	**214 ±51**	**45 %**
Total Agriculture	**8067 ± 1945**	**844 ± 192**	**39 ± 25**	**470 ± 80**	**100 %**

The detailed methodology developed here (Chapter 5) allows to reduce the uncertainty in the national and European inventories to less than 20 %, which is only half of the uncertainty of /IPCC 1997/ based inventories. However, even with the detailed methodology, GHG emissions reductions must be substantial in order to be statistically significant and verifiable in the context of the Kyoto Protocol. No single feasible mitigation measure may achieve this goal in the short run. Alternatively, rigorous case-specific measurements need to be performed.

6.3 Greenhouse gas reduction

The reduction of anthropogenic GHG emissions represents the most straightforward strategy to mitigate human induced global warming. The agricultural sector offers a wide range of technically feasible /Minami 1997; OECD 1997; Kroeze 1998; Mosier et al. 1998a; Velthof et al. 1998; Löthe 1999/ and economically viable /Bates 2001/ options to reduce the emissions of CH_4, N_2O and to a minor extent of CO_2.

6.3.1 Description of measures

A large number of possible measures to mitigate agricultural GHG emissions have been proposed, many of which have a technical nature or address farm management. Without a clear stimulation by agricultural and environmental policy, however, the adoption of such measures in farming practice will be slow and inefficient. Therefore, the technical feasibility alone does not give any indication about a potential realisation of the measures. In order to demonstrate the range of possible activities, some of the promising measures are listed and discussed below.

Agricultural soils

- Using nitrogenous fertiliser more efficiently leads to reduced N_2O emissions /Storey and McKenzie-Hedger 1997; Mosier et al. 1998b/. Matching N supply with crop demand can be realised by soil/plant testing to determine fertiliser N needs, minimising fallow periods to limit mineral N accumulation, optimising split application systems, and match N application to reduced production goals in regions of crop overproduction. Fertiliser use efficiency is also enhanced by the application of advanced fertilisation techniques such as controlled release fertiliser, placing fertilisers below the soil surface, foliar application of fertilisers as well as enhanced spreader maintenance /Bates 2001/. Some studies favour also the use of nitrification inhibitors /Delgado and Mosier 1996; Mosier et al. 1996/ but it has not been fully proven yet that the microbial fauna in soil is not negatively affected in long term and whether or not there are no environmental side effects. Furthermore, the use of nitrification inhibitors does not effectively reduce N_2O emissions in all situations, in particular not in loamy and clayey soils where the nitrification inhibitor is blocked by adsorption on mineral surfaces /Barth et al. 2001/ and when microbially available soil N contributes significantly to N_2O release /Linzmeier et al. 2001/. While in a 3-year field experiment, CH_4 oxidation was not negatively affected, soil respiration was significantly reduced by nitrification inhibitors /Weiske et al. 2001/. This suggests major negative effects on a wider range of microorganisms than those involved in nitirifcation. In a laboratory experiment with pig slurry N_2O emissions could not be reduced by the addition of Dicyandiamide (DCD, /Dendooven et al. 1998/). Nitrification inhibitors are still at an experimental rather than operational stage of development. A more elegant solution is to match fertiliser type to seasonal precipitation /Smith et al. 1998b/, for instance using ammonium-based fertiliser when the soil is wet but nitrate when the soil is dry. This option has been successfully demonstrated on Scottish soils only, so further testing in a wider range of climates is necessary.

- Tighten N flow cycles in the agricultural farm system reduces losses of all nitrogen compounds, including N_2O /Mosier et al. 1998b; Bates 2001/. This means to integrate animal and crop production systems in terms of manure re-use in plant production and maintain plant residue N on the production site (and account for it as fertiliser!).
- Optimise tillage, irrigation and drainage /Mosier et al. 1998b/. This may have beneficial effects on carbon sequestration and losses of other nitrogen compounds as well. However, some precaution seems necessary. /Mummey et al. 1998/ found in the Great Plains of the USA that the initial conversion of agricultural land to no-till showed greater N_2O emissions per hectare than conventional tillage. Differences between the two tillage scenarios were strongly regional and suggested that conversion of conventionally tilled soil to no-till may have a greater effect on N_2O emissions in drier regions. It still needs to be proven whether and how this approach can be transferred to European conditions.
- Letting agricultural land revert to forest and wooded land can help increase carbon sequestration (Chapter 2).
- Also grazed pastures have high emission rates, which should be reduced by management options. For instance, /Kammann et al. 1998/ found that an increase in the number of cuttings per year reduces N_2O emissions. The results suggest that the ability of plant roots to take up NO_3^- increases with increased cutting frequencies throughout the vegetation period, therefore reducing the amount of NO_3^- available for soil denitrifying microorganisms. These results need to be confirmed for a wider range of grasslands. In general, grasslands have been so far underlooked with regard to GHG mitigation.
- Riparian and wetland restoration enhances the terrestrial carbon sink, reduces nitrate transport to surface waters and coastal areas and may reduce regional N_2O emissions if the N_2O-to-N_2 ratio during denitrification is altered towards N_2 /Groffman et al. 2000/.
- Although farmed organic soils cover a relatively small fraction of the agricultural surface, they contribute to the biogenic GHG emissions from European agriculture almost as much as dairy cattle (Chapter 5). Therefore, the abandonment of farming on fens and former peatlands and complete rewetting of the areas has a locally high potential for reduction of CO_2 and N_2O emissions which is only slightly counteracted by possibly higher CH_4 production. The conversion of arable land on organic soils to grassland, combined with the abandonment of tillage or mechanical soil disturbance will give the first significant

relief of GHG emissions, which can probably be doubled by full abandonment and rewetting.

Animal husbandry

- Fewer livestock leads to reduced CH_4 emissions at the national level. Trends in cattle numbers already show a decline as a consequence of milk quota and the development of beef markets. However, a reduction of domestic livestock production below consumption levels would stimulate imports and hence translocate the GHG emissions outside the EU borders only.
- Methane emissions from enteric fermentation can be theoretically reduced at some extent through improving feed conversion efficiency by optimised diets /Jarvis and Pain 1994/ – an option almost exhaustively applied in intensive European livestock breeding systems, by increasing animal productivity through feed additives or breeding – an option which is greatly limited by animal health and consumers´ demands and current EU legislation – and by breeding potential, or increasing rumen conversion efficiency through feed additives – an option with similar constraints and need for more research prior to application /Finger 1999; Bates 2001/. Some natural tannins are capable to increase rumen conversion efficiency and reduce methanogenesis in the rumen of sheep, but adaptation of the ruminal microflora and complex interactions with other feed compounds demand for further research /Hayler 1999/. Also the supplementation of fats of sunflower seeds to the diet of cows and sheep reduces significantly methanogenesis /Finger 1999/. The replacement of roughage in cattle feed through concentrates provokes increased imports of feed at the cost of the maintenance of pastures and grasslands in Europe. /Bates 2001/ estimates a realisable mitigation potential in the EU by 2010 of 5 % of the cattle CH_4 emissions.
- A shift of aerobic to anaerobic storage of manure can reduce the N_2O emissions by a factor of 10 but will concomitantly increase CH_4 emissions /Bates 2001/, so the overall effect might cause similar or higher release of CO_2-equivalents. The opposite direction from anaerobic storage to aerobic one would in turn decrease CH_4 at the cost of more N_2O and stands against the current dominance of slurry-based systems. However, straw-based animal management is implemented in some cases for animal health reasons and as part of organic farming systems.
- Optimising the climatic conditions in animal houses /Gallmann et al. 2001/ and especially storing the manure under cool temperatures will significantly reduce the microbial activity in manure and hence reduce both CH_4 and N_2O production. It is equally important to remove the manure frequently and completely from the animal house because residues of old manure can act as microbial inoculants to stimulate the formation of CH_4 and N_2O in fresh manure. In general, many measures to reduce NH_3 emissions will also favour the reduction of GHG (e.g. /Müller 2001/) if the saved N is accounted for as fertiliser /Bates 2001/.

- Composting of manure, a controlled aerobic digestion, implies significant trade-offs of higher N_2O release against the avoidance of CH_4. Since the results in literature show contrasting effects in terms of budgets of CO_2-equivalents, composting is not recommended as mitigation option /Bates 2001/.
- Controlled anaerobic digestion of manure and waste represents the most promising option for GHG mitigation in manure management if the produced CH_4 is used for substitution of fossil fuels /Jarvis and Pain 1994; Bates 2001/. For instance, biogas could realistically mitigate 1.4 Tg CO_2-equivalents in Germany by the year 2010 /BMU 2000/. Biogas production from animal manure and slurry has additional benefits with regard to health and emissions of CO_2-equivalents after the application to soil /Wulf et al. 2000/.
- All nitrogen emissions (including N_2O) may be reduced by adapted feeding. The current N in feed can be reduced e. g. by 20 % in German pig fattening /Berg 1997/. Reducing N in feed is the most efficient and cheapest mitigation option to reduce losses of all N species including N_2O, NH_3 and nitrate leaching. This option, operating at farm level, shows a promising reduction potential for N emissions.

Whole farm approaches

Among the farm-based approaches, most studies have focused on cattle production /Jarvis et al. 1996; Löthe et al. 1997; Subak 1997; Haas et al. 2000; Müller 2001/.

/Subak 1997/ compares the full cycle emissions from extensive and intensive beef production in Europe in order to quantify the trade-off of an extensive, land-demanding beef production strategy against an intensive, energy demanding one. She concludes that intensive systems involving higher fossil-fuel inputs represent a better GHG reduction strategy if the alternative involves carbon storage opportunities on land. Competing demands such as landscape protection, ecological suitability of grazing land for intensivation or land use change etc. have been disregarded. However, her analysis relies on the hypothesis that carbon sequestration was equivalent to the reduction of GHG emissions, which opposes the present scientific understanding of non-permanence in terrestrial carbon sinks (Chapter 2). Therefore, her conclusions should be reverted to propose extensive systems rather than intensive ones.

Decreasing the intensity of grassland use under German farming conditions /Löthe et al. 1997/ reduces the emission of CO_2-equivalents through lower energy input and N_2O emissions. The effect of a switch from high intensity (0.73 kg CO_2-equivalents $(\text{kg milk})^{-1}$) to intermediate (0.62 kg CO_2-equivalents $(\text{kg milk})^{-1}$) is more pronounced than for a further extensivation. The results are consistent with options discussed previously and with the new interpretation of /Subak 1997/. Along the same line, according to a life cycle assessment of dairy farms in the South German Allgäu region /Haas et al. 2000/, extensivation of dairy production reduces the emission of GHGs with respect to all of the criteria per farm, per area, per livestock unit and per product (milk) while in contrast, organic dairy production scores even better with respect to emissions per farm, per area and per livestock unit, but GHG emissions per product (milk) are similar to intensive farming systems. Consequently, the performance of GHG reduction strategies depends largely on the development of human consumption patterns and of the market for agricultural products. Given the present surplus of milk production in the EU, the promotion of organic dairy farming turns out as the most promising option in the Allgäu.

Whole farm analyses by linear programming models, including emissions from prechains of products entering the farm system, identify a series of farm-based GHG mitigation options for representative farms in Southern Germany with either positive or no significant negative environmental side effects /Löthe et al. 1997; Angenendt et al. 2000; Müller 2001/. Per unit of farm product, best GHG reductions are achieved with

- Extensivation: Significant reduction of synthetic fertiliser use; extend agricultural production to entire land area instead of rotational fallow on 10 % of the land surface.
- Change in feeding strategy: replace hay by silage to reduce energy cost of drying.
- Intensification: Increase milk yield per cow, without, however, reducing the lifetime performance. There is hence an optimum intensity, which may be already achieved in temperate continental and boreal Europe.
- Slurry treatment: Anaerobic digestion of slurry for biogas production or, as minimum solution, slurry cover during storage and spreading with low NH_3 losses.
- Produce biomass for energy. Although at the farm level the effect on GHG emissions is small and eventually negative for the farm N balance, there are great positive carbon substitution effects from biomass use.
- Options involving higher fat content in animal diet may negatively affect other environmental indicators such as farm N balance.

These results have to be seen against their respective reference scenarios. Whether these findings can be generalized for other European regions still remains to be proven.

Whole systems approaches

Individual measures to mitigate agricultural GHG emissions generally bear the risk of sub-optimisation and “leakage” since they tend to address a local source in the overall agricultural production system only. Indirect effects on other sources and sinks in the agricultural system and impacts on other environmental issues have been rarely regarded.

The assessment of synergies and side effects requires the analysis of whole systems in the form of life cycles of products and services (cf. Chapter 3), entire farms or even the human society with regard to the agri-food sector.

Whole systems analysis indicates a number of potential management options to mitigate part of the CO_2-equivalents emitted during food and feed production /Robertson et al. 2000/. Such options ask for (a) tightening the nitrogen cycle (N_2O, NO, NH_3), and (b) substitution of synthetic fertilisers by biological nitrogen fixation (N_2O, NO, NH_3, CO_2 during the production of synthetic fertiliser) and hence support the recommendations obtained in farm-based studies.

Also changing land use from annual crops to perennial species and rewetting of drained organic soils has a significant potential for some specific areas but the potential for abandonment of agricultural production is limited by growing food demand /Rosenberg and Scott 1994/.

Other activities

/Sauerbeck 2001/ estimates the reduction potential of fossil fuel energy in the agriculture of industrialized countries at 0.01 to 0.05 Pg a^{-1}. Assuming that the reduction potential is proportional to the amount of fossil energy consumed, about a quarter of the potential /FNR 1999/ is located in the EU.

6.3.2 Environmental implications

Most of the GHG reduction measures on agricultural soils (Table 28) enhance either the fertiliser use efficiency of crops or more generally, the nitrogen cycling on farms. These measures will have clear additional benefits for the environment. They will reduce losses of reactive nitrogen as NH_3, NO, nitrate, and N_2O from agricultural soils to adjacent ecosystems and hence reduce nutrification and acidification. This will enhance biodiversity and water and soil quality. However, all of the options bear some risk of yield losses. Greenhouse gas mitigation by water management and on grasslands is yet poorly studied, but in theory, the measures could also have more positive than negative effects on the environment as long as water reservoirs are not depleted. Wetland restoration offers the widest scope for ancillary benefits, but requires a significant extensivation or even abandonment of agricultural use. A full analysis of side effects should also consider the effect of loss of agricultural products from these areas, which might be compensated by increased imports or intensification elsewhere unless the food consumption patterns of European citizens change.

Greenhouse gas reduction measures in animal husbandry focus on CH_4 reduction from enteric fermentation and manure management (Table 28). Methane emission from enteric fermentation can be reduced by intensification of cattle production through breeding, genetic improvement, replacement of grazing by concentrates and feed additives. This is based on the fact that the same yield is achieved with fewer animals, reducing the specific energy cost for maintenance and reproduction. Some of these options risk to have adverse effects on animal health or use substances presently not admitted by EC legislation. Replacing roughage by concentrates contradicts the European environmental policy to promote extensive use of grasslands, and to maintain grasslands, which also store significant amounts of carbon in soil. Concentrates will be partly imported to the farm and will hence negatively affect the nutrient balance at the farm gate. Reducing CH_4 release from enteric fermentation by reducing ruminant numbers would severely affect farmers in economic terms, but would be the measure most favourable for the environment for a number or reasons, including nitrogen loading, nutrification and acidification.

Aerobic storage of manure has been proposed as a measure to abate CH_4 emissions from manure storage, but will increase N_2O emissions in parallel at a rate that might overcompensate the reduced anthropogenic greenhouse effect from CH_4 /Bates 2001/. Therefore, this option is not pursued any more. In contrast, anaerobic digestion in conjunction with the production of energy from CH_4 is feasible and offers some environmental benefits with regard to reduced trace gas emissions. The most important effect has not yet been considered in Table 28, which results from the substitution of fossil fuels by renewable energy (cf. Chapter 6.4).

In summary, there are clear ancillary environmental benefits of GHG mitigation measures on agricultural soils, neutral or positive effects for anaerobic digestion, but measures that alter ruminal digestion or increase the cattle production intensity bear some risk of adverse environmental and animal health effects. Extensivation in crop production offers a win-win situation for climate as well as local and regional environment, whilst in animal production, benefits with reference to land area will not necessarily imply benefits per unit of product. In animal production, especially in the dairy sector, the normative framework by society and policy makers whether or not the present production and consumption level will be maintained will decide what GHG mitigation options will be effective.

Table 28 Summary of emissions reduction options in agriculture

Measure	**Emission reduction potential in EU15**		**Confidence**	**Environmental implications**
	Biological potential	Feasible by 2010		
	Tg CO_2-eq.	Tg CO_2-eq.		
AGRICULTURAL SOILS				
1. Fertiliser use efficiency	>10		Low	+A, +B, +C, -G, +H, +J
Enhanced spreader maintenance	4.0	0.4	Extrapolated from /1/	+A, +B, +C, -G, +H, +J
Fertiliser-free zones	0.3	0.03	Extrapolated from /1/	+A, +B, +C, -G, +H, +J
Optimising distribution geometry	0.9	0.1	Extrapolated from /1/	+A, +B, +C, -G, +H, +J
Improvements in fertiliser efficiency through precision farming	2.7	0.3	Extrapolated from /1/	+A, +B, +C, -G, +H, +J
Optimising N application by allowing for manure N and residual N	Up to 10		Low /1/	+A, +B, +C, -G, +H, +J
Continuation of set-aside		6.2 /1/	Medium	+A, +B, +C, -G, +H, +J
Synchronise N supply with crop demand	2.7 /2/		Medium-low	+A, +B, +C, -G, +H, +J
Match N and production to needs	2.7 /2/		Medium-low	+A, +B, +C, -G, +H, +J
Controlled release fertilisers	2.7 /2/		Medium-low	+A, +B, +C, -G, +H, +J
Fertiliser placement	1.8 /2/		Medium-low	+A, +B, +C, -G, +H, +J
Fertiliser form	2.7 /2/		Medium-low	+A, +B, +C, -G, +H, +J
Nitrification inhibitors	Site-dependent		Medium-low	+A, +B, +C, -G, +H, +J
2. Tighten N flows in farm	>2.7 /2/		Medium	+A, +B, +C, +H, +J
3. Optimise water management	?		Low	+A, +B, +D, +F, +G, -I
Irrigation	?		Low	
Drainage	?		Low	
4. Pasture management	1.1 /3/		Low	+A, +B, +C, +E, -G, +H, +J

Table 28 continued

Measure	**Emission reduction potential in EU15**		**Confidence**	**Environmental implications**
	Biological potential Tg CO_2-eq.	Feasible by 2010 Tg CO_2-eq.		
5. Wetland restoration	Up to 30		Low	+A, +B, +C, +D, +E, -G, +H, +J, +K
Riparian areas and wetlands	?		Low	
Abandonment of farmed organic soils	Up to 30		Medium-low	+A, +B, +C, +D, +E, -G, +H, +J, +K
Conversion of cropland on organic soils to grassland	Up tp 9		Low	+A, +B, +C, +D, +E, -G, +H, +J, +K
Maximum GHG Reduction in Agricultural Soils	**34**	**6**	**Low**	
ANIMAL HUSBANDRY				
6. Enteric fermentation	20	7	Medium	
Enhanced feed intake	8.2 /1/	4.1 /1/	High	+G
Replace roughage by concentrates	8 /1/	0.8 /1/	High	+G, -H, -I, -J
Change concentrates (additives)	7 /1/	0.3-0.4 /1/	High	+G, animal health?
Propionate precursors (additives)	12 /1/	0.6 /1/	High	+G, animal health?
7. Manure management				
Improve manure storage (cover, cooling)	2.6	0.7 /1/	Medium-high	+B, +C, better air quality
Composting	Negative?	Negative?		
Anaerobic digestion (heat&power)*	1.5 /1/	1.5 /1/	High	+B, +C, better air quality, provides energy
Anaerobic digestion (heat only)*	4.5 /1/	4.5 /1/	High	+B, +C, better air quality, provides energy
Maximum Reduction GHG in Animal Husbandry	**30**	**13**	**High**	
Maximum Total GHG Reduction	**>60**	**<20**	**Medium**	
Maximum GHG Reduction in % of agricultural emissions	**15%**	**5%**		

/1/	/Bates 2001/		E	Erosion	+ decrease; - increase
/2/	adjusted to EU-15 from /Mosier et al. 1998b/		F	Salinity	+ decrease; - increase
/3/	/Kammann et al. 1998/		G	Productivity	+ increase; - decrease
A	Biodiversity	+ increase; - decrease	H	Fertiliser use	+ decrease; - increase
B	Water quality	+ increase; - decrease	I	Energy use	+ decrease; - increase
C	Soil quality	+ increase; - decrease	J	Nitrate leaching	+ decrease; - increase
D	Flooding	+ decrease; - increase	K	Pesticides	+ decrease; - increase
*	without accounting for substitution of fossil fuel and associated CO_2 savings				

6.3.3 Potential and limitations

Table 28 synthesizes the potential for GHG reduction in agriculture in EU-15 at present knowledge. The biological potential indicates the theoretically possible maximum of GHG reduction without any technical, economic or social constraints, assuming 100 % of implementation. This is opposed to the technically feasible potential by 2010, indicating the possible contribution to meet the commitments under the Kyoto Protocol. Still, the latter potential does not fully account for social and economic constraints, so the actual opportunities might be even lower. The biological potential is calculated by applying average global reduction rates of /Mosier et al. 1998b/ to European emission rates and agricultural area, by extrapolation of the results of /Kammann et al. 1998/ to European grasslands, and extrapolation of the technically feasible potential in /Bates 2001/ to 100 % adoption in Europe.

Increasing nitrogen use efficiency and closing the agricultural nitrogen cycle offers a biological potential of between 3 and more than 10 Tg a^{-1} CO_2-equivalents by reducing N_2O emissions. The measures apply to most of the arable crops. Wetland restoration refer to only 4 % of European agricultural soils, but is highly efficient in reducing N_2O emissions, and in reverting the CO_2 source of drained peatlands to a carbon sink. Concomitant increase of CH_4 release is already considered in the biological potential of GHG reduction of between 9 and 30 Tg a^{-1} CO_2-equivalents. Although focused and associated with a range of environmental benefits, this measure is only available in short term if there are clear incentives to the farmers, who must abandon or extensify production on highly productive land. Without reducing cattle numbers, the CH_4 release from enteric fermentation can be theoretically reduced by up to 15 % below present rates by biochemical and management measures, which might even increase farm income /Bates 2001/. The mitigation of CH_4 emissions from manure storage include anaerobic digestion without accounting for carbon substitution (Table 28). There is hence a significantly higher GHG mitigation effect than indicated here, depending on whether heat and power or heat only are produced, the type and efficiency of the digestion plant and plant management. For the minor sources, no likely reduction measures have been identified by /Bates 2001/.

Measures in the areas of enteric fermentation and manure management will reduce CH_4 emissions with relatively high confidence. In contrast, at local project scale, the efficiency of GHG reduction measures addressing soil-borne N_2O emissions is much more uncertain. In the latter case, success also depends on local soil conditions and, e.g., whether there is heavy rain short after fertiliser application and whether nitrogen availability really matches crop demand. Another important constraint is the relative importance of N_2O from recent farm activities against “background emissions” related to land use history and natural variability. As a consequence, the biological potential for GHG reduction is balanced between agricultural soils and animal husbandry, but in the short term until the first commitment period of the Kyoto Protocol from 2008 to 2012, animal husbandry offers a higher potential for readily feasible and relatively certain measures (Table 28). In total, technical and biochemical measures can reduce agricultural GHG emissions by 5 % until 2010 or by 15 % if fully implemented across Europe. Obviously, none of the likely reductions nor the sum of all exceeds the uncertainty of 20 % in the present national inventories.

All of the options discussed above (Table 28) are available and many being implemented to some degree today. The main barrier to uptake appears to be the risk of a loss in yield, if the reductions in fertiliser application lead to under-fertilisation. Other potential barriers are a lack of information and understanding of potential benefits, although these might be overcome by information and training programmes.

More drastic changes in land use and land management and effective system-based strategies have been proposed /Berg 1997; Peel et al. 1997; Yagi et al. 1997; Kammann et al. 1998; Mosier et al. 1998a; Löthe 1999/. However, the realisation of options that rely on significant changes of agricultural practice, farm structures and agricultural products require a drastic and well-directed support by political and socio-economic actors and may not be readily implementable in the short run. In general, for evaluation of GHG reduction efficiency and in order to develop regional, national or local GHG mitigation strategies, the adequate reference unit (per unit of product, per unit of area, per farm,…) needs to be selected based on political and society-driven decisions. However, this decision is normative and hence beyond the scientific framework.

6.3.4 Greenhouse gas abatement costs

Given the uncertainty in the quantification of GHG emissions and our limited understanding of the causal interactions between agricultural management and changes in gas flux rates, average costs of GHG abatement are subject to considerable uncertainty, too. Moreover, the reduction potential and scope for given measures varies along climatic gradients, farm structures and management patterns. So costs calculated in one study cannot be generalized for other conditions across Europe. Nevertheless, some studies in literature illustrate the order of magnitude of potential abatement costs. Table 29 summarises GHG abatement costs in European agriculture and the scope of their applicability.

Also /Bates 2001/ reports reduction costs for all feasible measures, but the numbers are derived from a limited number of local and regional case studies and are hence highly uncertain. They may vary widely among EU Member States and may, in several cases, even be negative.

Greenhouse gas reduction costs range from negative, implying non-economic barriers, to relatively high levels as compared to 20 € t^{-1} CO_2 expected as the upper limit of CO2 emissions trading costs or 30 to 35 € t^{-1} CO_2 calculated as marginal costs to meet the European Kyoto commitment by domestic action only /Hendriks 2001/. On the other hand, if ancillary environmental benefits are included in a full environmental assessment, many of the options in Table 30 appear much more attractive.

Table 29 Greenhouse gas abatement costs in agriculture

Measure	Emissions included	Scope of study	Specific costs € t CO$_2$-equv.$^{-1}$	Reference
Technical measures				
Improvement in fertiliser efficiency	N_2O	Experimental, Netherlands	-45 to –2[a]	Bates 2001
Making better use of manure-N	N_2O	Theoretical, UK	-37 to 0.8 [a]	Bates 2001
Enteric fermentation: Replace roughage by concentrates	CH_4	Experimental	-212 [a]	Bates 2001
Enteric fermentation: Change composition of concentrates (extra fat, NSC)	CH_4	Experimental	-66 to –12 [a]	Bates 2001
Enteric fermentation: Improved level of feed intake	CH_4	Experimental	-43 [a]	Bates 2001
Enteric fermentation: Propionate precursors	CH_4	Experimental	170 to 340 [a]	Bates 2001
Manure: anaerobic digestion for heat and power[b]	CH_4	Farm scale, cool Europe Farm scale, warm Europe Centralised facility, cool Europe	-46 [a] 23 [a] -6 [a]	Bates 2001
Manure: anaerobic digestion for heat only[b]	CH_4	Farm scale, cool Europe Farm scale, warm Europe	143 [a] 38 [a]	Bates 2001
Continuation of set-aside in CAP	N_2O	Europe	0	Bates 2001
Farm management				
a) Reduced intensity: -40% N-fertiliser	N_2O, CH_4, CO_2 over life cylce per farm unit, including substitution of fossil fuel by biogas in options c) and d)	Forage-growing farm, South Germany (Allgäu)	a) positive[c]	Angenendt et al. 2000, Zeddies et al. 2000, Müller 2001
b) +1000 kg increased milk yield per cow			b) slightly negative[c]	
c) biogas for electricity and heat			c) and d) negative or positive depending on frame conditions and energy yield of cofermentation material [c]	
d) like c) with cofermentation				
a) reduced or no synthetic fertiliser	N_2O, CH_4, CO_2 over life cylce per farm unit	Forage-growing dairy farm, South Germany (Allgäu)	a) 20 to 60	Müller 2001, Zeddies et al. 2000
b) feed hay silage instead of dried hay			b) –1 to –7	
c) +1000 kg increased milk			c) –50	
d) More crude fats in dairy diet			d) 45 to 107	
e) Cover on slurry store			e) –30	
f) NH_3-saving slurry spreading			f) 100 to 160	

Table 29 continued

Measure	**Emissions included**	**Scope of study**	**Specific costs € t CO_2-equv.$^{-1}$**	**Reference**
g) biogas for electricity and heat	N_2O, CH_4, CO_2 over life cylce per farm unit, including substitution of fossil fuel in options g) to i)	Forage-growing dairy farm, South Germany (Allgäu)	g) 1 to 22	Müller 2001, Zeddies et al. 2000
h) like c) with cofermentation			h) –18 to positive depending on gas yield	
i) Energy crop production			i) 76 to 100	
a) Minimum tillage	N_2O, CH_4, CO_2 over life cylce	Crop production farm, South Germany (Kraichgau)	a) 95	Löthe 1999, Zeddies et al. 2000
b) Extensified production		Mixed farm, South Germany	b) 35	
c) Change in processing of grains for storage		Mixed farm with dairy cows	c) 3	
d) Biogas with cofermentation		Mixed farm with dairy cows	d) –0.1	

[a] investment costs, operation and maintenance costs The specific costs are calculated using a real interest rate of 4% and using the technical lifetime of the option, i.e. installation.

[b] without effects of carbon substitution by use of biogas

[c] change in gross margin

NSC: non-structural carbohydrate

6.4 Carbon substitution

Carbon substitution means the replacement of fossil fuels and materials by renewable sources. Here, the major opportunity, the bioenergy option, is considered. Carbon substitution relies on the assumption that CO_2 emitted from burning biofuels does not contribute to global warming as long as the biofuels are produced in a sustainable way and as long as local effects on the carbon balance in the agroecosystem and “leakage” effects through market distortions or failed substitution of fossil energies are subtracted from that benefit. An in-depth discussion is given in Chapter 1.

6.4.1 Description of measures

Biomass for energy occurs in either solid form (agricultural and forest residues, wood, energy crops), liquid form (plant oils, processed biomass such as rape methyl ester and ethanol), or gaseous form in the case of biogas from agricultural waste. It is converted to either heat or power or both. This variety of choices produces varying benefits and environmental side effects. So when assessing the climate benefits of bioenergy, the frame conditions and technology of substituting a fossil fuel by bioenergy must be carefully considered.

6.4.2 Environmental implications

The production of biomass for energy is associated with the emission of CO_2, CH_4 and N_2O from fossil fuels used for the production of farm inputs, from fertiliser synthesis and from mechanical work on the cropped area. In addition, soils emit N_2O leaking from nitrogen cycling. Hence, the production of biomass for energy is far from climate neutrality, especially the highly fertilised annual crops. Soils represent the dominant source of N_2O. The greenhouse gas trade-offs associated with the production of biomass vary across Europe as a function of land use intensity and of "background" emissions. The latter reflect climatic and soil conditions and hence in some way also land use history (Chapter 4). Therefore, the real – undoubtedly beneficial – climate effect of carbon substitution has to be carefully determined by environmental analysis tools such as life cycle assessment.

Carbon cycle

The combustion of biomass is frequently considered as CO_2 neutral. This assumption is only true if the biomass is produced on an area where the carbon cycle is in equilibrium. This requires a constant amount of carbon stocks in biomass, litter and soil, and constant carbon turnover rates. In particular, a constant amount of litter input to the soil and long-term steady state in soil disturbance by harvest, tillage and drainage characterise equilibrium conditions. Soil memorizes disturbances over decades. Changes in land use and land management practice, associated with the increased production of energy crops, and interannual climatic variability affect the carbon cycle, so the assumption of CO_2 neutrality is oversimplified for most biomass production systems. However, the carbon loss effect will be small as compared to the carbon substitution effect.

The European political and economic environment favours the production of biomass for energy on set-aside areas, and the likely future increase in the area under biofuel production will change the common agricultural practice in terms of crop species and crop rotations. Options introducing herbaceous or woody perennials on arable land or reclaming degraded land will sink carbon in biomass and soil, while options increasing the removal of biomass or the frequency of soil disturbance will lead to carbon losses.

Nitrogen cycle

Nitrogen losses are intrinsic to intensive agricultural production systems and occur in food and feed production as well as in biomass production. The agricultural nitrogen cycle leaks to groundwater (nitrate) and to the atmosphere (N_2, N_2O, NO, NH_3) depending on the concentration of mineral nitrogen in the soil solution. The increase of mineral nitrogen, e.g. by fertilisation or by enhanced mineralisation rates, stimulates nitrogen losses from soil. Annual biofuel crops tend to demand for higher nitrogen fertilisation and to have a lower nitrogen use efficiency than perennial grasses and woody plantations, leading hence to higher nitrogen losses (Table 30). But still, there is a variety of newly introduced or almost forgotten crop types with lower N demand than the presently grown “grandes cultures”.

In addition, almost 10 g of N_2O-N are emitted during the production of each kg of nitrogen fertiliser /FNR 1999/. Over an entire life cycle, the environmentally more friendly option of nitrogen fixation by leguminous crops is difficult to apply and needs further research. 75 % of the gaseous nitrogen losses during the production of biomass originate from agricultural soils, 25 % come from fertiliser production. About half of each fraction is emitted as N_2O.

Ammonia losses in Table 30 are assumed to originate from volatilisation of mineral ammonium-based fertilisers. In case animal manures are applied, NH_3 losses will be much greater. However, in the latter cases of manure-producing farms, there is a methodological difficulty whether the NH_3 losses have to be allocated to the respective animal products or to the biomass. Life cycle comparison of different farm types (such as conventional versus organic farms with and without animals, intensive versus extensive production) producing biomass will have to be performed to sort out how to adequately allocate environmental effects of manure as compared to synthetic fertiliser.

Table 30 Nitrogen fertilisation, nitrogenous emissions and regional environmental impact from biomass production under German conditions. Assumptions as in Chapter 3 and as in /Kaltschmitt and Reinhardt 1997; Freibauer and Kaltschmitt 1998/.

		Winter wheat	**Oilseed rape**	**Ryegrass**	**Poplar (SRC)**
Fertiliser	kg N ha^{-1} a^{-1}	100-140	160-180	80	31-67
Nitrate leaching	kg N ha^{-1} a^{-1}	30-42	48-54	24	(9-20)
Ammonia	kg N ha^{-1} a^{-1}	2-3	3-4	1.6	0.6-1.3
Nitrous oxide	kg N ha^{-1} a^{-1}	2.2-5.2	2.7-5.9	2.3	(1)
Nitric oxide	kg N ha^{-1} a^{-1}	1-1.4	1.6-1.8	0.8	0.3-0.7
Nutrification[a]	kg PO_4-equ. ha^{-1} a^{-1}	23	26	12	7
Acidification[b]	kg SO_2-equ. ha^{-1} a^{-1}	14	15	7	5
Greenhouse gases[c]	t CO_2-equ. ha^{-1} a^{-1}	3.1	2.7	1.4	0.5
Substitutable primary energy	GJ ha^{-1} a^{-1}	150 (light heating oil)	30 (diesel)	~150 (light heating oil)	150 (light heating oil)
Substitution of fossil fuels	t CO_2-equ. ha^{-1} a^{-1}	9-10	2	~11	11

SRC: short rotation coppice; ( ) highly uncertain

[a] 1 kg PO_4-equivalent = 1 kg PO_4 or 0.42 kg NH_3-N or 0.42 kg NO_2-N or 0.42 kg nitrate-N

[b] 1 kg SO_2-equivelent = 1 kg SO_2 or 1.88 kg NH_3 or 0.7 kg NO_2 or 0.88 kg HCl

[c] 1 kg CO_2-equivalent = 1 kg CO_2 or 310 kg N_2O or 21 kg CH_4

/Heijungs et al. 1992/

Local environment

There is no general rule whether the local effects of biomass production in terms of species and structural diversity are positive or negative. In any case the local situation has to be considered, in particular, for what crop the biomass substitutes and how intensive is the production system. Farm and forest residues are generally considered as products without environmental burden from earlier stages of their life cycle. This is only true if the ecosystem from which the biomass is removed is in carbon equilibrium. In case biomass removal is enhanced as compared to a reference situation, soil carbon stocks will be depleted, leading to CO_2 release. Furthermore, the enhanced removal of coarse woody debris from forest negatively affects insects biodiversity. Biofuel production can also result in a more diverse structure of the landscape and potential positive effects on biodiversity /McLaughlin and Walsh 1998/ if native species are grown and pest control is reduced as compared to food crops /Borjesson 1999/. There is a wide range of crop choices available which do not require the application of pesticides. These should be preferred to conventional “grand cultures” since they concomitantly enlarge the species diversity in crop rotations.

Perennial grasses and woody plantations prevent soil erosion /McLaughlin and Walsh 1998/ and are suitable for the reclamation of degraded land and the restoration of minelands. They have also been successfully applied for the reclamation of land polluted by heavy metals and for desalination /Chiaramonti et al. 2000/.

Regional environment

The release of reactive compounds from agriculture to the regional environment results in nutrification and acidification. To quantify the emissions from a typical biomass production system, typical German agricultural conditions of the mid 1990s are assumed /cf. Kaltschmitt and Reinhardt 1997; Chapter 3/. In the life cycle assessment, all sources and sinks on the farm and in the prechains (fertiliser production, seeds, other farm input) are accounted for. As a result (Table 30), the environmental effects of nutrification and acidification from annual crops systems are twice as high as from perennial grass. Short rotation coppice leads to the lowest environmental damage on a hectare-base as well as per GJ of substituted fossil energy (Table 30, Table 31).

Global environment

Life cycle assessment of whole cereal crops substituting light heating oil (Chapter 3) and similar studies /Kaltschmitt and Reinhardt 1997/ yields higher environmental trade-offs in terms of the depletion of the stratospheric ozone layer through N_2O emissions, eutrophication and often also acidification of ecosystem through the loss of reactive nitrogen compounds from the agricultural system and burning N-rich feedstocks, but indicates significant benefits with regard to global warming when bioenergy substitutes fossil fuels. As a result of the study performed in Chapter 3, emissions of GHGs, especially of N_2O, and other environmental trade-offs associated with the production and use of biomass as an energy carrier significantly reduce its benefit of being a largely climate-neutral option for carbon substitution by up to 30 % under unfavourable conditions involving highly fertilised annual crops (Chapter 3). Less N-demanding biomass, in particular woody perennials such as short-rotation coppice, yield much more favourable results in life cycle comparisons /Kaltschmitt and Reinhardt 1997/.

Exemplary life cycle comparisons are synthesized in Table 31. The results can be interpreted as consistent and clear trends against a variety of substituded fossil fuels rather than as fixed emission savings since they have to be seen in their model context regarding the underlying reference scenario and substituted fossil energy carrier. Evidently, residues and wood turn out as options with very small or even positive environmental side effects given a relatively stable level of emission reduction per GJ of substituted fossil energy. As a result of Table 31, all options yield comparable positive effects with reference to saved kg CO_2-equv. GJ^{-1} substituted fossil energy. However, clear differences occur for environmental implications. Only woody biomass does not significantly increase N_2O emissions over the entire life cycle, which is not only a greenhouse gas, but also destroys the stratospheric ozone layer. Woody biomass also scores best with regard to nutrification and acidification effects. Perennial grasses have an intermediate ranking.

Table 31 Environmental implications of biomass for energy

Bioenergy option versus fossil fuel	**Emission reduction** Saved kg CO_2-equv. GJ^{-1} substituted fossil energy	**Environmental implications**	**Reference**
Solid biomass in heating plant			
Residues versus heating oil	65 to >70	Soil carbon loss?	Kaltschmitt and Reinhardt 1997
Straw versus heating oil	65 to 70	Soil carbon loss?, >10 g GJ^{-1} more N_2O More HCl, CO	Kaltschmitt and Reinhardt 1997
Whole cereal crops versus heating oil	35 to 70	>20 g GJ^{-1} more N_2O More HCl, CO	Chapter 3, Kaltschmitt and Reinhardt 1997
Perennial grasses versus heating oil	65 to 70	>10 g GJ^{-1} more N_2O More HCl, CO	Kaltschmitt and Reinhardt 1997
Wood from short rotation coppice versus heating oil	>70	Soil carbon loss? Less SO_2	Kaltschmitt and Reinhardt 1997
Wood from forest versus heating oil	>70	Soil carbon loss? Less SO_2	
Liquid biomass for transportation			
Plant oil versus diesel	65	>20 g GJ^{-1} more N_2O Less SO_2 , more CO	Kaltschmitt and Reinhardt 1997
Rape methyl ester (RME) versus diesel	65	>20 g GJ^{-1} more N_2O Less SO_2	Kaltschmitt and Reinhardt 1997
Ethanol from sugar beet versus gasoline	65	>20 g GJ^{-1} more N_2O More SO_2	Kaltschmitt and Reinhardt 1997
Ethanol from wheat versus gasoline	60	>20 g GJ^{-1} more N_2O More SO_2	Kaltschmitt and Reinhardt 1997

The substitution efficiency per hectare is greater for solid than for liquid biomass since more of the originally solid biomass is converted to energy without any chemical processing. But also energy yield per hectare and conversion efficiency vary greatly among energy crops.

For instance, /McLaughlin and Walsh 1998/ found that in the US, the energy efficiency of bioethanol production by switchgrass (*Panicum virgatum*) exceeds the one of corn, the primary present source of bioethanol, by up to 15 times. Carbon sequestration in soil when converting corn fields to switchgrass adds another climate benefit. /Zeddies 2001/ gives a parallel example for Europe. Whilst RME use substitutes for 2 to 3 t CO_2-equ. ha^{-1} a^{-1} only, ethanol from sugar beet is much more efficient due to higher yields and better conversion efficiency. In the latter case, substitution effects range between 16 and 22 t CO_2-equ. ha^{-1} a^{-1}. Hence the findings for solid biomass can be generalized to liquid biomass as well: Perennials lead to lower environmental trade-offs than do intensive annual cultures.

The use of agricultural and forestry residues is more environmentally friendly than the production of dedicated energy crops. Wood residues and straw might negatively affect the soil carbon balance. Biogas production from animal manure and slurry has additional benefits with regard to health and N_2O emissions after the application to soil.

The environmentally sound production of biomass should be optimised by the following indicators:

- Nitrate leaching is low
- N_2O emission is minimised (selection of site, site management, low nitrogen demand of crop)
- Synthetic fertiliser use (N, P) is minimised (e.g. replacement by manure, low demand of crop; legume-based systems need further research)
- Erosion control in winter and in seasons with higher probability of extreme events
- Good water use efficiency in areas prone to drought
- Biodiversity is enhanced
- Energy yield per input of resources is high.

In this sense, woody species are generally preferable to perennial grasses, which are again preferable to annual crops.

6.4.3 Potential and limitations

A rough estimation shall quantify to what extent increased N_2O emissions in the life cycle of bioenergy crops reduce their efficiency in reducing the total anthropogenic GHG emissions in the EU. The following assumptions are made. Biomass is produced as whole cereal crops under similar conditions as food crops, on 8 % of the agricultural area in the European Union – roughly 9.5 Mio ha /EUROSTAT 1999/, with an average energy yield of 174 GJ ha^{-1} a^{-1} /FNR 1999/ and GHG emissions and reductions as calculated in Chapter 2. The GHG emissions from this area would be equivalent to 0.5 % (0.3 to 2.7 %) of the anthropogenic emissions in the EU (EU member states). The biomass could substitute 1.65 EJ a^{-1} of heat and would hence approximately double its present share /FNR 1999/. This scenario would reduce the total anthropogenic GHG emissions of the EU in 1995 by 1.1 to 1.9 %. In contrast, if accounting for carbon substitution only, the CO_2 emissions of the EU would decrease by 1.8 to 3.1 % and the total anthropogenic GHG emissions by 1.5 to 2.5 % respectively, overestimating the GHG reduction by 30 %. Even the relatively inefficient choice of whole cereal crops substituting light heating oil would yield a reduction of CO_2-equivalents equivalents of 11 to 20 % of the agricultural GHG emissions in the EU, or up to 25 % of the EU´s commitments under the Kyoto Protocol.

Substituting C-intensive fossil fuels such as coal and more efficient energy conversion by combined heat and power clearly increase the substitution potential. Also a more nitrogen-efficient production of the biomass or the choice of a less nitrogen-demanding type of crop such as perennial herbaceous or woody cultures improves the efficiency of carbon substitution by bioenergy. This negative effect of GHG emissions in the prechains amounts to only 10 % of the substitution efficiency if biomass is produced from intensive perennials and is close to negligible if forest products are used. So does a higher energy yield per hectare. Optimising carbon substitution potential in Europe demands for high carbon substitution efficiency per heactare since the land area available for bioenergy production is limited by other use such as food and feed production.

/Kaltschmitt 2000/ estimates the potential for biomass substituting fossil fuels in Europe (continent) between 8.2 and 9.9 EJ a^{-1}, which is equivalent to 0.46 to 0.91 Pg CO_2 or 10 to 20 % of Europe´s fossil fuel emissions disregarding, however, limiting effects of prechains and non-CO_2 emissions. These scenarios consider potential technological progress in energy conversion efficiency. For the European Union and Germany, according reduction potentials of about 0.094 and 0.012 Pg a^{-1} C (0.34 and 0.044 Pg a^{-1} CO_2; 6.4 and 0.8 EJ a^{-1}), respectively, can be derived from /FNR 1999/. This is consistent with market projections for the EU by 2010 of 4 EJ a^{-1} /Grassi 1999/ of which 3.1 EJ a^{-1} are in the heat sector, 0.8 EJ a^{-1} in power generation and <0.1 EJ a^{-1} in bioethanol/biomethanol markets.

In most European states, a major part of the biomass resources is presently not yet being used. Among all possible options in agriculture and forestry, energy crops, agricultural residues such as straw, coarse woody debris from forest management /Kaltschmitt 1997/, and animal manure and slurry have the largest potential as future bioenergy sources. In the EU and Germany, about 40 % of the bioenergy potential is attributed to the production of dedicated energy crops /FNR 1999/.

6.4.4 Greenhouse gas abatement costs

Again, as for GHG abatement costs of GHG reduction measures in agriculture, reservations have to be made with regard to the interpretation of costs. Bioenergy offers a range of environmental, social, political and economic benefits, contributes to energy supply from local sources and offers potential for development in rural areas. Greenhouse gas abatement costs do not reflect these ancillary benefits. Nevertheless, cost estimates made in literature are reported here in order to give an order of magnitude at which the carbon substitution effect alone can serve to promote bioenergy. The costs given in Table 32 do not consider prechains of fuel production and provision. They rely on the assumption that biomass is implemented instead of new efficient combined cycle power plants as new capacity in order to compensate for growth in energy production and to replace decommissioned capacity /Hendriks et al. 2001/.

Table 32 EU-15 average costs and potential for emission reduction by bioenergy /Hendriks et al. 2001/

	Reduction potential	**Specific costs (€ t^{-1} CO_2) at discount rate**			
Bioenergy measure	**Tg CO_2 equ.**	**2 %**	**4 %**	**6 %**	**Sector specific***
Heat only on solid biomass	25	-42	-42	-41	-41
CHP on solid biomass	4	-38	-34	-30	-26
CHP anaerobic digestion	4	-28	-23	-17	-11
Heat only on wood	64	15	15	16	17
CHP on wood	29	17	20	24	28
Ethanol	9	228	236	246	256
Biodiesel	24	287	299	312	326
TOTAL	**159**				

* Real interest rate of 4 %, using a technical lifetime of installations

/Löthe 1999/ calculates GHG abatement costs for RME, rapeseed oil and energy crops produced in South German farms. Depending on the type of substitutable energy sources, costs are between 8 € t^{-1} CO_2-equ. (rapeseed oil), 58 € t^{-1} CO_2-equ. (RME), and 2 to 96 € t^{-1} CO_2-equ. (energy crops). Biogas produced in the model dairy farm of /Müller 2001/ is associated with costs in the range of –18 to 23 € t^{-1} CO_2-equ. according to scenarios of low to high gas yields and with/without co-fermentation. Energy crops of that model farm result in GHG abatement costs between 26 and 67 € t^{-1} CO_2-equ. if used in a heating plant or if co-fired to coal in a power plant, respectively. The differences in the abatement costs as compared to the European average estimates in Table 32 suggest that there is a significant variability in implementation costs within the EU.

To summarise, there are bioenergy options involving either woody sources from forest residues or biogas production which are already economically competitive under present frame conditions. Subsidies and tax exemptions (Chapter 1) distort the market at some extent, promoting too much the RME option, which is neither the most cost-effective nor an efficient, environmentally friendly bioenergy option.

6.5 Carbon sequestration

Carbon sequestration means the removal of CO_2 from the atmosphere to sinks in terrestrial ecosystems (vegetation and soils).

6.5.1 Description of measures

Carbon sequestration measures fall in two categories: 1) land use change, i.e., change from arable annual cultures to perennials, afforestation, and reforestation, abandonment of drained organic soils, and 2) land management, i.e. low tillage, enhanced application of organic amendments, pasture and forest management (cf. Chapter 2).

6.5.2 Environmental implications

Accounting for carbon sequestration in agroecosystems means to offset energy-related emissions against biological sinks, and to equate measures aimed at reducing the combustion of fossil fuels with the creation of sinks. However, the lifetime of terrestrial carbon sinks is generally limited to decades, given the dynamic nature of the terrestrial carbon cycle. Also the future behaviour of the biosphere under changing climate cannot be predicted over more than a few decades. Apart from the afforestation of agricultural land (Kyoto Protocol, Art. 3.3), all potential carbon sequestration measures in agriculture belong to the category of additional activities in land management (Kyoto Protocol, Art. 3.4). Admitting that there is considerable potential and feasibility for human induced carbon sinks in European forests and agricultural soils, scientific arguments stand against the accounting of carbon sink to fulfill commitments in the context of the Kyoto Protocol. Crucial issues such as variability and uncertainty of the sink, the attribution to human-induced versus natural or indirect sinks, potential for leakage through enhanced deforestation or carbon losses elsewhere, the non-permanent nature of the sink, and verification on national level /Valentini et al. 2000/ (cf. also Chapter 1.2.1) remain to be resolved in a scientifically sound way before carbon sinks match the (original, scientific) spirit of the Kyoto Protocol under the UNFCCC to stabilise the atmospheric CO_2 concentration at a level „that would prevent dangerous anthropogenic interference with the climate system“ /UNFCCC 1992/.

Some measures to sequester carbon in the agricultural phytomass and soils, like green manuring, afforestation, extensivation and low-till management, may also enhance soil fertility and biodiversity and reduce the risk of erosion. Negative side-effects can be induced by increased use of fertilisers and pesticides (Chapter 2).

Carbon sequestration in heavily C-depleted arable soils offers significant environmental benefits in terms of soil erosion control, enhanced soil fertility, stabilised soil structure and enhanced diversity of soil organisms. Fragile, silty soils in the steppe climates of the US and Russia were successfully stabilised by deep-rooting perennial grasses /McLaughlin and Walsh 1998; Larionova et al. 1998/.

There are many good ecological and economic arguments for carbon sequestration in agricultural soils and of the maintenance of the carbon stocks in ecosystems apart from the mitigation of climate change.

6.5.3 Potential and limitations

There is a potential for carbon sequestration in agricultural soils of up to 60 Tg a^{-1} C (220 Tg a^{-1} CO_2) in Europe /Smith et al. 2000/, which may even provide environmental benefits. However, the model underlying this study was derived from long-term experiments and assumes continuous implementation of carbon sequestration measures such as low-tillage, enhanced use of organic amendments and intercrops. This contrasts the present framework of agricultural subsidies and farming practice, in which measures are supported and maintained during one to several years only, but do not persist for decades. Reducing disturbance by low tillage to increase the topsoil carbon content only is worth if this new management is maintained over long periods of time /Robertson et al. 2000/. Therefore, the potential given in /Smith et al. 2000/ will be probably lower in reality. Moreover, uncertainties associated with the magnitude and non-permanence of the sink strength, the anthropogenic contribution to the sink and its future behaviour are as high as the sink strength itself. Apart from these limitations, sinks can only buy time over at maximum decades in order to further develop strategies for the decarbonisation of the energy system.

Carbon sequestration rates above 1 Mg ha^{-1} a^{-1} C when converting arable fields to permanent deep-rooted grasslands as found by /McLaughlin and Walsh 1998; Larionova et al. 1998/ suggest that highly degraded fragile soils in steppe climates are more effective carbon sinks than more stable and shallower soils in the more humid regions of Europe, which also tend to have shallower rooting depth.

According to the Kyoto Protocol, afforestation of agricultural land is only accountable if it has happened since 1990. This significantly limits the potential for carbon sequestration. Beese (pers. comm. 2001) calculated that even in an optimistic scenario of 10 years of afforestation between 2000 and 2010 of 100,000 hectare annually (total of 1 Mio ha) will yield only minor carbon sequestration because the growth of small trees is slow in the beginning and reaches maximum growth rates only after 40 years. Moreover, it is unlikely that sufficient area will be available for afforestation in the EU.

6.5.4 Greenhouse gas abatement costs

Carbon sequestration has specific features with scientific and political backgrounds which make the calculation of greenhouse gas abatement costs difficult.

In terms of science constraints, the potential non-permanency of the carbon sink introduces additional uncertainty through a probability function of the magnitude of the carbon sink strength that can be maintained during the first commitment period and even more beyond. Therefore, carbon sequestration projects require a rigorous carbon monitoring in order to be reliable, transparent and effective mitigation measures. Secondly, the carbon sink will naturally saturate in the future, so sink rates will gradually become smaller. This will not only increase the costs per ton of carbon sequestered but has major implications for accounting under the Kyoto Protocol.

Although costs for implementation of carbon sequestration measures such as afforestation will be comparatively low, the abatement costs have to consider also running costs for monitoring, reporting and verification. Since good practice guidance for carbon sinks reporting will only be elaborated by IPCC in the near future, the uncertainties are too high to identify the true costs of carbon sequestration in the frame of the Kyoto Protocol. Afforestation since 1990 achieves only limited carbon sinks in 2010, so the investement tends to be clearly above the „feasible“ line of 20 € per t of CO_2.

6.6 Socio-economic and policy measures

The GHG mitigation measures discussed so far operate at project scale, e.g. on a piece of land, a farm, or a small region. In contrast, in the following, a theoretical approach is taken. Based on literature and own calculations, the impact of the socio-economic and political frame conditions on agricultural GHG emissions is assessed. Consequently, socio-economic and political mitigation options seek to change these frame conditions. The resulting scenarios sometimes seem unrealistic because assumptions on changes in societies and in human behaviour may be hard to implement in practice until 2010. Nevertheless, the socio-economic perspective on GHG mitigation provides a good lesson about the ultimate importance of external driving forces. Subsequently, the efficiency of agricultural policy measures is discussed.

6.6.1 Socio-economic measures

Agricultural production is driven by demand for food and fibre and its political and socio-economic frame conditions. Therefore, despite all theoretical and realistic technical and management-oriented measures within the agricultural systems, the largest potential for GHG mitigation in agriculture is found beyond the farms by changing the driving forces of demand for food, feed and other agricultural products. Since the food sector dominates the consumption of agricultural products in Europe, only changes in human diet are considered here.

/Bleken and Bakken 1997/ calculate the N cost of food in the Norwegian society, defined as the ratio between fertiliser-N input (including animal manure) and the N in products. The N cost is around 3 for wheat, 14 for dairy products and 21 for meat. Only minor change in human diet could reduce the need of N for food production by 45 %.

Also according to /Isermann and Isermann 2000/, animal consumption in the EU exceeds the nutritional needs of the population by factor 2.4. Therefore, a healthier diet could reduce livestock in the EU and in Germany by 58 and 43 %, respectively. Scenarios of technical improvements of nitrogen use efficiency, sustainable agricultural production, abstinence (vegetarians only) and nutrition without protein surplus would reduce the N input to German agriculture by 24, 60, 81, and 87 %, respectively /Isermann and Isermann 2000/. Similar rates for the reduction of agricultural N_2O emissions can be assumed. The latter two scenarios are however fairly speculative.

Scenarios of hypothetical changes in food consumption patterns were developed by /Henze et al. 1998/, applying WHO diet recommendations to average European food energy consumption in 1990. The authors calculate changes in crop production areas and number of domestic animals in the EU for four scenarios of reduced consumption of fat, free sugar and total energy (Table 33 legend). For instance, scenario S 30/10 practically means the replacement of 30 g capita^{-1} day^{-1} of fat (meat, dairy products, vegetable oil) by 30 g capita^{-1} day^{-1} of carbohydrates (cereals, potatoes, vegetables) and 9 g capita^{-1} day^{-1} of proteins (pulses, fish) /Henze et al. 1998/. The scenarios are used here to derive the related changes of agricultural GHG emissions by applying the method developed in Chapter 4. *Ceteris paribus* conditions for crop and animal management are assumed. The area of rice paddies changes at the same rates as the area of total cereals. As indicated in Table 33, changing food consumption patterns can reduce agricultural N_2O and CH_4 emissions by up to 60 % below 1990 levels. The overall reduction of CO_2-equivalents drop by 40 % without any dedicated GHG mitigation measure being implemented.

Changing food consumption patterns shows a great potential for GHG mitigation in agriculture, which clearly exceeds the uncertainty in the agricultural GHG inventories and will be verifiable in the context of the Kyoto protocol. Significant additional environmental benefits with regard to reduced eutrophication and acidification can be achieved simultaneously. However, changing socio-economic patterns requires an enormous effort, and drastic changes as those underlying the scenarios of healthy diets may become realistic in the long term only if ever. Policy has a major role in promoting a change in consumers´ behaviour.

Table 33 Reduction scenarios of agricultural GHG emissions according to WHO diet recommendations /Henze et al. 1998/ against 1990 levels

	N_2O soil	**N_2O animals**	**N_2O total**	**CH_4 soil**	**CH_4 animals**	**CH_4 total**	**CO_2-equ.**	**Reduction in**
	%*	**%***	**%***	**%***	**%***	**%***	**Total %***	**Pg CO_2-equ.**
S 30/10	-6	-11	-25	-01	-22	-23	**-15**	-0.063
S 15/0	-17	-26	-57	10	-53	-53	**-35**	-0.149
S 30/10-15	-14	-19	-35	-12	-32	-33	**-23**	-0.098
S 15/0-15	-22	-31	-63	-05	-59	-60	**-40**	-0.173

* change in GHG emissions as % of 1990 emissions from agriculture in EU-15

S 30/10:	Reduction of the fat consumption to an energy share of max. 30 % and reduction of the consumption of free sugar to an energy share of max. 10 % against 1990 food energy consumption
S 15/0:	Reduction of the fat consumption to an energy share of max. 15 % and reduction of the consumption of free sugar to an energy share of max. 0 % against 1990 food energy consumption
S 30/10-15:	Reduction of the fat consumption to an energy share of max. 30 % and reduction of the consumption of free sugar to an energy share of max. 10 % against 1990 food energy consumption level reduced by 15 %
S 15/0-15:	Reduction of the fat consumption to an energy share of max. 15 % and reduction of the consumption of free sugar to an energy share of max. 0 % against 1990 food energy consumption level reduced by 15 %

6.6.2 CAP Reform and Agenda 2000

The Common Agricultural Policy (CAP) has a direct, pure impact on the production in the heavily regulated agricultural sectors of sugar and milk only, where there are quota systems. For all other sectors, any changes in the production of crops and animal products will result from the combined pressure of the economic environment – global markets, the Uruguay Round Agreement, and internal EU markets as well as CAP measures. General trends can be detected although it is difficult to identify their final causes.

The CAP reforms in 1992 are not related to the mitigation of climate change and only tangentially to the environmental agenda. The driving forces for change of price support of agriculture are the large budgetary cost of such support and the incorporation of agriculture into multilateral trade agreements (GATT, WTO) /Adger et al. 1997/. Consequently, the effect of CAP measures on GHG emissions is small.

In the arable sector, the shift from production based payment to direct area based payment has led to an optimised/reduced input of N fertiliser per hectare. However, due to significant N_2O emissions as a consequence of land use history and natural background emissions, the lower N input reduced the N_2O release from agricultural soils insignificantly by 2 % since 1990 only, as derived from the inventories in Chapter 5. No effort is undertaken to reduce the farming intensity on drained organic soils.

In the livestock sector, the lower market support prices and the milk quota have promoted a shift in livestock type from cattle towards swine and poultry and a general decline in cattle numbers. According to /Storey and McKenzie-Hedger 1997/, however, surplus beef production has continued to be encouraged by export subsidies in the livestock sector.

Overall, CAP and macroeconomic conditions resulted in a reduction of CH_4 emissions by 7 % and of nitrogen spread as manure by 6 % between 1990 and 1997 (Chapter 5).

The Agenda 2000 CAP Reform is likely to continue the trend of reducing dairy and other cattle and maximising N fertiliser use efficiency, so future trends will probably be similar, unless specific measures for GHG mitigation are undertaken. Nevertheless, without being specifically designed for GHG mitigation, Agenda 2000 has some links to climate change policy: 1) Shift from production-based payments to area-based subsidies (Market measures; „First pillar“), 2) maintenance of set-aside instrument, and option to produce non-food biomass on set-aside land, and 3) emphasis on sustainable rural development („Second pillar“), e.g. Agri-environmental measures (protection from nitrate leaching,…), Farm investment aid scheme (for units to dispose of animal manure…), Forestry measures.

As effect of the first pillar, in general, the move from production-based payment to direct payments provides the potential for support payments be better targeted to meet environmental objectives and to move to less intensive agricultural systems. A working document of the European Commission, DG Agriculture /DG VI 2001/ on the impact of measures agri-environmental measures under the regulation (EEC) no. 2078/92 gives insight in the present links to GHG abatement in the framework of the second pillar. Among the GHG abatement options identified above, some are indeed already part of agri-environmental measures:

- Reduced input measures, especially organic farming, nature protection measures (wetlands and peatlands are not separately addressed but could fall in this category) reduce N input and surplus, and hence N_2O emissions.
- Extensivation, set-aside for 20 years should reduce N_2O emissions and sequester carbon. However, the 20-year running time led to low take up.
- Input reduction measures (formal integrated production methods, controlled by an authorised body; specific reductions per farm according to soil type and following a soil analysis; precise application of fertiliser according to the changing needs of the plants show substantial reductions in fertiliser where programmes are applied widely, reducing N_2O emissions.
- Measures to convert arable land to grassland, assure mixed farming and a more sustainable rotation show better preservation of segetal vegetation, and soil quality should also favour carbon sequestration.
- Fire prevention measures show limited results as data runs are too short; but considerable evidence of positive results shown from erosion prevention measures (such as mulch seeding) and N-leaching reduction measures, such as green-cover crops, leading to reduced N_2O emissions and carbon sequestration.
- Extensivation of livestock and maintenance of extensive systems reduces GHG emissions per area only, but not necessarily per unit of product. However, beef and milk consumption trends in Europe are stable or declining, so the overall climate effect will probably be favourable.

Following the subsidiary principle, agri-environmental programmes are not implemented uniformly in the EU, but are in the responsibility of the member states or even further down the scale, of federal states. Taking, for instance, the agri-environmental programmes of Baden-Württemberg and North-Brandenburg /Baudoux 2000; Kazenwadel 1999/, carbon sequestration is promoted by subsidies for intercrops, reduced tillage, and conversion of arable land to grassland. The maintenance of extensive grasslands can contribute to the preservation of carbon in the ecosystems, depending on how alternative use would look like. Refraining from synthetic fertiliser will reduce N_2O emissions from soils if it goes along with reduced N input. In any case, the measure avoids N_2O emissions from fertiliser production. /Baudoux 2000; Kazenwadel 1999/ conclude that the environmental efficiency of measures under agri-environmental programmes is enhanced by a better definition of goals and regionalisation. The same applies to GHG mitigation efficiency.

Consequently, important first steps have been already undertaken. Their effect on GHG emissions, however, is hard to evaluate since emissions have a wide natural variation. Present GHG inventories, operating with average emission factors, do not capture management changes yet. Adequate monitoring and accounting systems need to be developed which operate at local scale in order to consider changes in GHG emissions at farm scale. Nevertheless, still more can be done, given the fact that agricultural subsidies are not yet fully directed towards sustainable development. If climate change mitigation and the other environmental issues were fully incorporated in multiple objective policy reforms, many aspects of the natural environment could be enhanced to a much greater degree that is at present expected /Storey and McKenzie-Hedger 1997/. Multiple targets encompass erosion control, abatement of eutrophication and acidification, preservation of landscape beauty and recreational value, enhancement of biodiversity and human health, but also global goals in terms of mitigation of climate change and protection of the stratospheric ozone layer.

Such specific measures to be adopted targeting the reduction of GHG emissions more directly, in short term, should encompass:

- Include GHG in advise for best agricultural practice
- Programmes to encourage the adoption of technical mitigation measures (e.g. further investments in the frame of the Rural Development Regulation, e.g. for animal houses)
- Maintenance and further development of programmes to enhance the production and use of renewables and of biogas.

In addition, regulatory instruments aimed at or linked to reducing GHG emissions have been proposed, e.g. CO_2 tax /Angenendt et al. 2000/, high nitrogen tax /Kazenwadel 1999/, regulations for nitrogen application and standards on nitrogen levels in soil /Storey and McKenzie-Hedger 1997/, animal product tax /Barnick 2001/. However, as illustrated by /Angenendt et al. 2000; Müller 2001/ based on farm models in South Germany, regulatory instruments such as CO_2 tax, higher energy prices, nitrogen tax, GHG emission tax, caps on emissions per area or product, emission trading, do not operate in a cost-efficient way since small GHG reductions go along with relatively high microeconomic and/or macroeconomic costs as compared to the costs of technical options given above.

6.7 Conclusions and outlook

Except for the promotion of renewables, none of the potential GHG mitigation strategies has ever been purposely tested in farming practice nor been properly monitored at national or European scale. Apart from political and socio-economic inertia, the lack of scientific knowledge, particularly the large uncertainty in the quantification of emissions and emissions reductions has still prevented the implementation of GHG mitigation measures although some of them are feasible and viable. Therefore, it is crucial to reduce the scientific and methodological uncertainty in the quantification of sources and sinks and to develop adequate tools for assessing environmental trade-offs such as leakage, indirect GHGs and other environmental side effects. There are two undispensable prerequisites science must provide in order to make GHG mitigation in agriculture attractive for implementation:

1) Reducing uncertainty in the quantification of sources and sinks to the level required for meeting the Kyoto commitment and
2) Developing tools that make potential mitigation measures transparent and verifiable.

This study has contributed to both aspects.

6.7.1 Reducing uncertainty in quantification of sources and sinks

The uncertainty in the quantification of N_2O emissions from agricultural soils on national and European scale was significantly reduced by improving the methodology for the inventory calculation. The importance of a distinction at sub-national scale between climatological, pedological and management factors that control the annual N_2O release was highlighted. Subsequently, a more detailed methodology compatible to the IPCC methodology to quantify biogenic N_2O, CH_4 and CO_2 emissions from agriculture was developed and applied. It reduced the uncertainty in the national and European inventories to a degree that makes maximum possible reductions verifiable under criteria of the Kyoto Protocol.

Given the complexity of the agricultural sector and the variety of GHG sources, inventories of national annual agricultural GHG emissions and sinks like the one presented here cannot give direct recommendations for mitigation strategies. However, they identify the important sources. 85 % of the agricultural GHG emissions originate from cattle, mineral and organic agricultural soils. Cattle produce predominantly CH_4, mineral agricultural soils mainly N_2O, and CO_2 originates from peat oxidation. The methodology to calculate national GHG inventories from agriculture was improved here in relation to /IPCC 1997/, but still involves uncertainty of about 20 % in national inventories. The methodology suits for the more accurate fullfilment of the national reporting commitments under the UNFCCC and, by a refinement of emissions factors, has identified major gaps and shortcomings of the IPCC approach with regard to N_2O emissions from soils, emissions from farmed organic soils, and animal housing. Monitoring of GHG reduction in agriculture, however, asks for the application of sophisticated process-oriented ecosystem models working at local level that capture the natural variability of GHG fluxes and of their driving forces on temporal time steps of days. Concomitant comprehensive measurements are needed.

6.7.2 Assessment of mitigation measures

Based on the detailed quantification of agricultural sources and sinks of GHGs, the study synthesized potential technical, socio-economic, and political mitigation measures in agriculture with regard to their scope, strategy, environmental implications, potential and limitations, GHG abatement cost ranges and feasibility. The three possible strategies of carbon sequestration, carbon substitution and GHG reduction were analysed and compared.

Carbon sequestration. There is some potential for carbon sequestration in agricultural soils, which may even provide environmental benefits. However, uncertainties associated with the sink strength, the anthropogenic contribution to the sink and its future behaviour are as high as the sink strength itself. Terrestrial carbon sinks are non-permanent and hence can only alleviate and postpone the increase in atmospheric CO_2 for some decades. Carbon sequestration is no real GHG mitigation strategy, but can only buy time to develop other measures. The slow dynamics of carbon uptake after afforestation of arable land and lack of economic incentives for conversion of arable land to permanent grassland limit the scope of land use change on most of the available land in Europe. Carbon sequestration through low tillage only works if soil is being undisturbed for decades, opposing the present mainly short-term oriented agri-environmental subsidies. Best long-term effects are achieved through abandonment and rewetting of drained organic soils.

Carbon substitution. All solid, liquid and gaseous bioenergy options reduce the anthropogenic greenhouse effect and can make significant short-term and long-term contributions to GHG mitigation. Emissions of greenhouse gases, especially of N_2O, and other environmental trade-offs associated with the production and use of intensively grown biomass as an energy carrier significantly reduce its benefit of being a largely climate-neutral option for carbon substitution by up to 30 %. This constrains from an environmental perspective the use of annual crops and favours the use of perennials, especially of residues and woody biomass. The biological potential of bioenergy in the EU allows to substitute for 10 to 20 % of its anthropogenic GHG emissions. Bioenergy belongs to the most promising mitigation options with regard to feasibility of implementation and reduction potential:

- Production of biogas by anaerobic digestion of slurry and waste for substitution of fossil fuels

- Production of biomass for bioenergy from residues and, preferably woody, perennials.

Greenhouse gas reduction. Agriculture offers a range of feasible technical and sectoral approaches to reduce biogenic GHG emissions. Among the technical options, the most promising ones with regard to feasibility of implementation and reduction potential are:

- Optimisation of manure management by complete and frequent removal from animal houses and storage under cool temperatures
- Optimisation of nitrogen fertiliser management and replacement of synthetic fertilisers by biological nitrogen fixation and manures
- Abandonment of farmed organic soils for subsequent restoration of peatlands against compensation for income losses.

Sectoral and socio-economic mitigation options with high reduction potential are:

- Reduction of livestock, especially of cattle with concomitant reduction of demand
- Integration of animal and crop production (for closing nutrient cycles) as opposed to the present trend of specialisation
- Promotion of organic farming with concomitant education of consumers
- Promotion of local product flows and local nutrient cycling
- Reduced consumption of meat and dairy products.

If economic and social constraints are disregarded, technical GHG mitigation options offer GHG reduction of 20 to 30 % of the agricultural GHG emissions, i.e. of 2 to 3 % of the anthropogenic GHG emissions of the EU. Social changes reducing the demand for animal products can double this potential and will have ancillary great benefits with regard to other environmental and health issues. In case successfully implemented, the sectoral options offer a potential for GHG mitigation well above 30 % of the agricultural GHG emissions - larger than any combination of technical measures.

6.7.3 Link to the Kyoto Protocol and Climate Policy

The potential for carbon sequestration in the agricultural sector has been might have been overestimated for the first commitment period. In addition, the Bonn Agreement on Accounting for carbon sinks greatly reduces the opportunities for carbon sequestration under Art. 3.4 of the Kyoto Protocol. Carbon substitution and the technical GHG reduction options fit well to the requirements for activities accountable under the Kyoto Protocol because they are well-defined measures which will be additional to any ongoing activities. The implementation of technical options can be readily monitored at farm level. The efficiency of the measures with regard to real reductions of GHG emissions is, however, difficult to control because a large number of small changes will have to be integrated. In addition, even in optimal scenarios, the biogenic GHGs will not be reduced by more than 20 to 30 %, which makes the overall effect hardly verifiable against the uncertainty in the national inventory or against inverse atmospheric models. The effect of the bioenergy options, however, will be easily measurable and verifiable by reduced consumption of fossil fuels. The adoption of technical options can be stimulated by regulations, advise to farmers or direct agreements with farmers. Many of the technical options favour extensivation and technical innovation and go in line with ongoing efforts addressing other environmental issues.

The implementation of the sectoral options is impossible to monitor on project level and generally difficult to trace except for changes in livestock numbers, and the consumption of meat and dairy products, but direct cause-effect relationships are restricted by external influences of macro-economic and social factors. Nevertheless, a verification of the overall effects could be realised by large-scale atmospheric monitoring, and indirect effects such as reduced consumption of synthetic fertilisers or smaller numbers of animals are reflected in agricultural census. If the effect of reduced CH_4 and N_2O emissions can be traced in the atmospheric signal, the sectoral measures go conform with the spirit of the UNFCCC. Sectoral options imply drastic effects on farmers´ income and consumers´ behaviour and require therefore stringent encouraging political actions.

6.7.4 Outlook

European agriculture theoretically offers a significant potential for GHG mitigation via a range of technical and market-based measures. Their implementation is most effective if based on local environmental and socio-economic conditions and if clear incentives are provided. Maintenance and strengthening political support for bioenergy from perennials and residues and measures to promote technical innovation in animal husbandry and extensivation in arable cropping seems most efficient.

Other promising options still require further research. In particular, the abandonment of farmed organic soils possibly offers a wide range of environmental benefits in terms of water resources and quality, biodiversity and climate, but alternative uses, dynamics of peatland recovery, carbon sequestration and methane release need to be clarified. Another promising research area is the replacement of synthetic fertilisers by nitrogen-fixing legumes and better use of organic fertilisers. Management of legumes in crop rotations poses unsolved risks to nitrogen losses through leaching or outgassing as N_2O after harvest or ploughing. On the other hand, there are clear benefits from the avoidance of emissions during synthetic fertiliser production.

This study provides a useful, readily implementable methodology to quantify agricultural GHG sources and sinks for national and subnational inventories submitted to the UNFCCC. As a next challenge, in order to account for GHG mitigation in agriculture under the Kyoto Protocol, a further refinement of GHG inventories towards processed-based spatialized models and concomitant long-term observations at farm level are as essential as the efficient implementation of mitigation measures.

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Summary

Motivation, aim and scope

Climate change is one of the most urgent dangers for human society in the 21st century. The Third Assessment Report of the Intergovernmental Panel on Climate Change (IPCC) unquestionably identifies Man as major cause for climate change. In the European Union (EU), 11 % of the anthropogenic greenhouse gases (GHGs) are released from agriculture, which also represents the largest source of methane (CH_4) and nitrous oxide (N_2O).

Policy makers already recognized the need for action about a decade ago and set up political frame conditions for the mitigation of anthropogenic GHG emissions. The United Nations Framework Convention on Climate Change (UNFCCC) signed in 1992 commits the signatory parties to annual submissions of the national anthropogenic GHG sources and sinks and to mitigate emissions. Targets for the latter are defined in concrete terms in the Kyoto Protocol (1997). Consequently, science has the task to facilitate the implementation of these political treaties in the fields of

1. Elaboration of a transparent methodology and application to produce national GHG inventories (Art. 4.1 of the UNFCCC)
2. Identification and assessment of GHG mitigation measures (Art. 4.2 of the UNFCCC)
3. Monitoring and verification of the actual magnitude of GHG abatement accountable under the Kyoto Protocol (Kyoto Protocol and Agreement on its implementation at the Conferences of the Parties in Bonn (CoP6-II) and Marrakech (CoP7)).

This dissertation contributes to all of the three issues mentioned above in the field of European agriculture by developing methodologies for GHG inventories and through scientific information. It analyses essential components of efficient potential GHG mitigation strategies in the agricultural sector of the EU by identifying where important mitigation potential is located and what uncertainty, environmental ancillary effects and costs are associated with it. Measures analysed here encompass carbon sequestration, carbon substitution by bioenergy, and GHG reduction in agriculture through technical, socio-economic and political means. Since the Kyoto Protocol allows the accounting of mitigation measures at project scale, relevant information and methodologies need to be provided at an adequate, regionally disaggregated resolution in order to consider regional differences in agricultural practice and emissions.

Structure and methodology

The dissertation uses a cumulative form, i.e. the chapters follow a series of papers published in or submitted to peer-reviewed scientific journals. These are amended by a shorter chapter on carbon sequestration (Chapter 2) and embedded in a frame provided by the introductory (Chapter 1) and synthesis chapter (Chapter 6). The latter involves also an assessment of GHG mitigation measures in agriculture. The papers have been published in or submitted as follows:

- Chapter 3: published in Umweltwissenschaften und Schadstoff-Forschung. Zeitschrift für Umweltchemie und Ökotoxikologie 10(6), 1998: 353-365
- Chapter 4: submitted to Biogeochemistry
- Chapter 5: submitted to European Journal of Agronomy
- Chapter 6: utilised as a basis for the report of the European Commission, European Climate Change Programme Working Group on Agriculture, to be published by end of 2001.

A part of the data underlying Chapter 3 were established in the project “Nachwachsende Energieträger: Grundlagen, Verfahren, Ökologische Bilanzierung” funded by the Deutsche Bundesstiftung Umwelt (DBU). Nevertheless, the important methodological integration of soilborne GHG emissions, the calculations and the assessment were independently performed by the author. Chapters 4 and 5 emerge from the co-ordination of the Concerted Action FAIR3-CT96-1877 sponsored by the Commission of the European Communities, Agriculture and Fisheries (FAIR) specific RTD programme. The partners provided literature on GHG emissions and national data on agriculture and agricultural census. The author has independently performed the entire methodological development, calculations, and assessment. All other chapters are independent of specific projects.

In order to achieve the aim, literature reviews, further development of methodologies for environmental assessment and GHG accounting, and calculations of GHG emissions and reduction potentials were performed. The lack of political guidance for definitions and accounting rules hampers the assessment of carbon sequestration, so here, the state of scientific knowledge was synthesized only. Bioenergy was analysed by a case study of the production and utilisation of an exemplary bioenergy crop. As a basis, the methodology of life cycle assessment is further developed and applied for evaluating environmental trade-offs of carbon substitution. The detailled knowledge about agricultural GHG emissions represents the undispensable basis for decisions upon greenhouse gas mitigation strategies on national and European level. Therefore, a cause-oriented spatialized methodology for national GHG inventories was developed to allow a regionally adjusted quantification of biogenic GHG emissions and to reduce the uncertainty in the national and European estimates of agricultural greenhouse gas emissions. Based on the detailed inventory, the GHG mitigation potential in the agricultural sector of the European Union was quantified and assessed. The methodology combined literature surveys with methodological development and modelling. Literature surveys provided the scientific information to the assessment of GHG mitigation measures and data from European GHG measurements. There were three aspects of methodological development. 1) Life cycle assessment of bioenergy was extended to a full integration of biogenic GHGs and of reactive nitrogen species and related uncertainties and consequences for climate, global and regional environment, using sensitivity analyses. 2) In order to spatially disaggregate the quantification of soilborne N_2O fluxes, statistical models were set up on the basis of a stepwise multiple linear regression analysis of annual N_2O emissions and combinations of various soil, climate, and management variables. The data were derived from measurements across the EU reported in literature. 3) A cause-oriented methodology for GHG inventories was developed that allows a detailed quantification of GHG sources in the various steps of the agricultural production chain and a more management- and animal-specific estimation of the share of the agricultural sub-sectors than the default approach of the IPCC. Specifically, as innovation, manure management was separated into the CH_4 and N_2O sources "animal houses", "manure storage", and "manure spreading". For soils, the CH_4 sink was included and N_2O emissions were disaggregated on a regional basis with respect to climate, site conditions and management by the methodology developed in step 2). Applying the new methodologies and information from agricultural census, soil maps and climate data, the agricultural GHG sources and sinks were calculated at sub-national resolution for the period 1975 to 1997. The results were discussed in the context of overall anthropogenic GHG emissions and served as a reference to identify the GHG mitigation potential and associated uncertainties.

Results and discussion

Carbon sequestration

There is significant potential for carbon sequestration in Europe, which is maximised by a mix of measures including conservation tillage, organic amendments, and in particular regeneration of woodlands and afforestation. However, there is a fundamental drawback of carbon sequestration in the long run. The sink may saturate or reverse to a source under changing climatic or management conditions. So carbon sequestration in the terrestrial biosphere only buys some time, but cannot offset fossil fuel emissions in the sense of a full GHG mitigation option. European agriculture will at maximum sequester carbon in the order of 200 Tg a^{-1} CO_2, which may also provide other environmental benefits with regard to soil protection, biodiversity and water resources.

Carbon substitution

In the bioenergy case study of whole cereal crops, the comparison of a heat provision based on whole cereal crops versus light oil over the entire life cycle showed that heat from biomass results in a lower environmental impact regarding the anthropogenic greenhouse effect, but a higher one with respect to eutrophication, acidification and the depletion of the stratospheric ozone layer. The environmental trade-off will be more favourable if the energy crops are produced in extensive agricultural systems or if other mitigation measures are undertaken or if the emissions from the combustion in the biomass fired boiler are reduced. In a broader perspective, all solid, liquid and gaseous bioenergy options reduce the anthropogenic greenhouse effect and can make significant short-term and long-term contributions to GHG mitigation. Emissions of GHGs, especially of N_2O, and other environmental trade-offs associated with the production and use of intensively grown biomass significantly reduce its benefit of being a largely climate-neutral option for carbon substitution by up to 30 %. This constrains from an environmental perspective the use of annual crops and favours the use of perennials, especially of woody biomass, as well as of residues. The biological potential of bioenergy in the EU allows to substitute for 400 to 800 Tg a^{-1} CO_2-equivalents, more than half of which is directly from agriculture.

GHG reduction in agriculture

Methodological achievements and quantification of emissions

Before measures for GHG reduction in agriculture can be identified the respective sources and sinks need to be well described in a regionally disaggregated way. Therefore, first an adequate methodology to quantify biogenic GHG emissions in European agriculture was developped and applied. Based on a review of N_2O field studies in Europe, major soil, climate and management controls of N_2O release from agricultural mineral soils in the European Union were identified. In arable soils, fertilisation as well as climate and soil parameters, especially the topsoil nitrogen content, significantly affect the level of annual N_2O emissions. Arable soils show lower emissions in oceanic and dry subcontinental temperate climate than in wet-continental climate with high freeze-thaw emissions and in boreal climate. Therefore, two separate regression models were developed. Nitrous oxide emissions from arable soils in oceanic and dry subcontinental temperate regions lie in average below 2 kg ha^{-1} a^{-1} N_2O-N and rarely exceed 5 kg ha^{-1} a^{-1} N_2O-N. They were estimated by fertiliser, topsoil nitrogen content and a soil moisture indicator. In wet-continental and boreal climate, N_2O emissions range much wider between 0 and 17 kg ha^{-1} a^{-1} N_2O-N in dependence of available nitrogen, estimated by fertiliser and topsoil nitrogen content. Compared to existing methods for large scale inventories, the regression models allow a better regional fit to measured values since they integrate additional driving forces of N_2O emission. For grasslands, a fertiliser-based model was slightly modified towards higher emission estimates. Due to an extreme variability, no climate, soil nor management parameters could be included in the empirical grasslands model.

Also for the entire agricultural sector, a detailed methodology was developed compatible to the Guidelines of the Intergovernmental Panel on Climate Change (IPCC) to quantify the annual biogenic emissions of greenhouse gases released from European agriculture. This approach relies on emission factors and regression equations derived from all long-term measurements in Europe available by the end of 1999. Applying the methodology, the biogenic greenhouse gas emissions from agriculture within the European Union (EU) and within its Member States are calculated for the period from 1975 to 1997 at a spatial resolution of regions or federal states. As a result, in 1995, European agriculture emitted 0.84 ± 0.29 Tg N_2O, 8.1 ± 1.9 Tg CH_4 and 39 Tg ± 25 CO_2, which adds up to 470 ± 80 Tg CO_2-equivalents or 11% of the overall anthropogenic GHG emissions of the EU. At the EU level, these numbers are surprisingly close to the official inventory submitted under the United Nations Framework Convention on Climate Change (UNFCCC). But compared to the latter, the approach taken here leads to higher agricultural CH_4 emissions in Denmark and the Netherlands, higher N_2O emissions in Austria, Finland, Germany, the Netherlands, (and Sweden, depending on reporting year), and lower N_2O emissions in Italy. In countries with – even small – areas of farmed organic soils, CO_2 emitted from peat oxidation can significantly contribute to the overall emissions. Hence, only the detailed approach adequately resolves regional and national specifics of agricultural conditions. It furthermore reduces the uncertainty in the emissions estimates to half of the one in inventories based on the IPCC Guidelines. Fair agreement with inverse atmospheric models was achieved. These results suggest that the methodology developed and applied here could serve as a significantly improved standard for official inventories of biogenic greenhouse gas emissions from EU Member States.

Evaluation of GHG reduction measures

Based on the detailled inventory, potential GHG reduction measures were described and evaluated. Promising measures for reducing agricultural GHG emissions promote the extensivation of arable cropping by reduced synthetic and overall nitrogen inputs and technological innovation in animal husbandry, the latter being best accompanied by a further decline in animal numbers. Abandonment of drained, farmed organic soils could double the potential for GHG reduction, but feasibility and long-term environmental and socio-economic effects need further research before this measure can be implemented. Many measures will also provide ancillary environmental benefits, especially those tightening local nutrient cycles. The biological potential for technical GHG reduction measures in EU agriculture is between 100 and 200 Tg a^{-1} CO_2-equivalents. Socio-economic changes, i.e. of human dietary preferences and of agricultural markets can stimulate even larger changes in GHG emissions. Efficient GHG mitigation in agriculture needs strong financial, political and educational support.

Zusammenfassung

Motivation und Aufgabenstellung

Der Klimawandel ist eine der größten Bedrohungen für die Menschheit im 21. Jahrhundert. Der Dritte Bericht des Zwischenstaatlichen Gremiums für Klimawandel (IPCC) läßt keine Zweifel an den weitgehend anthropogenen Ursachen der Klimveränderung. In der Europäischen Union (EU) stammen 11 % der anthropogenen Treibhausgase aus der Landwirtschaft, die gleichzeitig die größte Quelle von Methan (CH_4) und Distickstoffoxid (N_2O) darstellt.

Die Notwendigkeit zum Handeln wurde bereits vor einem Jahrzehnt erkannt und in politische Rahmenbedingungen zur Minderung der anthropogenen Treibhausgasemissionen umgesetzt. Die 1992 verabschiedete Klimarahmenkonvention der Vereinten Nationen (UNFCCC) formuliert jährliche Berichtspflichten der anthropogenen Treibhausgasquellen und -senken in den Unterzeichnerstaaten ebenso wie die Verpflichtung zur Emissionsreduktion, deren Umfang das Kyoto-Protokoll (1997) konkretisiert. Damit ergeben sich klare Anforderungen an die Forschung, um die Implementierung der politischen Abkommen zu ermöglichen:

1. Erarbeitung einer transparenten Methodik und Erstellung der nationalen Treibhausgasinventare (Art. 4.1 der UNFCCC)
2. Identifikation und Bewertung von Treibhausgasminderungsmaßnahmen (Art. 4.2 der UNFCCC)
3. Monitoring und Verifikation der tatsächlichen, im Rahmen des Kyoto-Protokolls anrechenbaren Treibhausgasminderung (Kyoto-Protokoll und Vereinbarungen zur Implementierung bei der Konferenzen der Mitgliedsstaaten in Bonn (CoP6-II) und Marrakesch (CoP7)).

Diese Dissertation leistet zu allen drei oben genannten Anforderungen Beiträge im Bereich der Landwirtschaft der EU. Dabei stehen die Entwicklung von Methoden und die Bereitstellung wissenschaftlicher Informationen zu Treibhausgasemissionen und Treibhausgasminderung im Vordergrund. Die Arbeit analysiert wesentliche Elemente möglicher Klimaschutzmaßnahmen in der europäischen Landwirtschaft im Hinblick auf Potenziale, Umweltwirkungen, Minderungskosten und jeweils damit verbundene Unsicherheiten. Diese umfassen Maßnahmen zur Kohlenstoffspeicherung, Kohlenstoffsubstitution durch Bioenergie und Treibhausgasminderung durch technische, sozioökonomische und politische Maßnahmen. Da das Kyoto-Protokoll Maßnahmen auf Projektebene anerkennt, müssen entsprechende Methoden und Datengrundlagen zur Emissionsberechnung auch regional differenziert bereitgestellt werden, um regionale

Unterschiede in der landwirtschaftlichen Praxis und bezüglich der Emissionsraten zu berücksichtigen. Vor diesem Hintergrund widmet sich ein Teil der Arbeit der Entwicklung einer regional differenzierten Methodik zur Erstellung von Treibhausgasinventaren. Sie wird angewendet, um detailliert nach Quellen und Regionen die biogenen Treibhausgasemissionen der europäischen Landwirtschaft zu quantifizieren.

Struktur der Arbeit und Vorgehensweise

Die Dissertation ist in kumulativer Form verfasst, d.h., die Kapitel bestehen aus einer Reihe von Artikeln, die entweder in referenzierten Zeitschriften bereits erschienen oder eingereicht sind. Diese Artikel ergänzt ein kürzeres Kapitel zur Kohlenstoffspeicherung (Kapitel 2). Den Rahmen bilden ein einleitendes (Kapitel 1) und ein Synthese-Kapitel (Kapitel 6), das darüber hinaus die Bewertung von Treibhausgasminderungsmaßnahmen zum Thema hat. Die Artikel sind in den folgenden Zeitschriften erschienen bzw. eingereicht:

- Kapitel 3: veröffentlicht in Umweltwissenschaften und Schadstoff-Forschung. Zeitschrift für Umweltchemie und Ökotoxikologie 10(6), 1998: 353-365
- Kapitel 4: eingereicht bei Biogeochemistry
- Kapitel 5: eingereicht bei European Journal of Agronomy
- Kapitel 6: dieses Kapitel diente als Grundlage für einen Bericht der Europäischen Kommission, Europäisches Programm zum Klimawandel, Arbeitsgruppe Landwirtschaft, der Ende 2001 erscheinen wird.

Ein Teil der in Kapitel 3 verwendeten Daten wurde im Projekt “Nachwachsende Energieträger: Grundlagen, Verfahren, Ökologische Bilanzierung” erarbeitet, das die Deutsche Bundesstiftung Umwelt (DBU) finanziell unterstützte. Die Autorin hat allerdings die methodisch wichtige Integration der bodenbürtigen N_2O-Emissionen, die Berechnungen und die Bewertung eigenständig durchgeführt. Die Kapitel 4 und 5 entstanden im Rahmen der Koordination des EU-Projektes „Concerted Action FAIR3-CT96-1877“, gefördert von der Europäischen Kommission, Spezielles Forschungsprogramm zu Landwirtschaft und Fischerei (FAIR). Die Projektpartner lieferten Daten und landwirtschaftliche Statistiken aus den jeweiligen Partnerländern sowie Literatur zu Treibhausgasemissionen. Die Autorin hat die gesamte Methodenentwicklung, Berechnung und Bewertung eigenständig durchgeführt. Alle anderen Kapitel entstanden unabhängig von laufenden Forschungsprojekten.

Die Vorgehensweise war wie folgt: Es wurden Literaturstudien durchgeführt, Lebensweganalysen und Treibhausgasinventare methodisch weiterentwickelt und Treibhausgasemissionen und deren Minderungspotenzial berechnet. Das Fehlen politischer Vorgaben zu Definitionen und Regelungen zur Anrechnung von Maßnahmen im Bereich der Kohlenstoffspeicherung erschwerte eine fundierte Analyse dieser Strategie in der europäischen Landwirtschaft vor dem Hintergrund des Kyoto-Protokolls. Daher wurde hier nur der derzeitige Stand der wissenschaftlichen Diskussion skizziert. Bioenergie wurde am Beispiel einer Fallstudie zur Produktion und Nutzung von Getreideganzpflanzen analysiert. Dazu wurde die Methodik der Lebensweganalyse weiterentwickelt und angewandt, um die Umwelteffekte der Bioenergie zu erfassen. Die detaillierte Analyse von Treibhausgasminderungsmaßnahmen erfordert als unverzichtbare Grundlage zuerst eine genaue, quellenorientierte Quantifizierung der Treibhausgasflüsse in der Landwirtschaft. Daher wurde eine ursachenorientierte, räumlich spezifische Methodik zur Berechnung nationaler Treibhausgasinventare entwickelt, die erlaubt, Quellen und Senken von Treibhausgasen regional differenziert zu ermitteln und die gegenwärtigen Unsicherheiten in Inventaren auf nationaler und europäischer Ebene zu reduzieren. Auf der Grundlage dieses detaillierten Inventares wurde das Potenzial von Treibhausgasminderungsmaßnahmen in der europäischen Landwirtschaft ermittelt und im Hinblick auf Umwelteffekte und Effizienz diskutiert. Methodisch verband die Arbeit Literaturstudien mit Methodenentwicklung und Modellierung. Die Literaturstudien lieferten wissenschaftliche Informationen für die Bewertung von Klimaschutzmaßnahmen in der Landwirtschaft und Daten von Treibhausgasmessungen in Europa. Die Methodenentwicklung erstreckte sich auf drei verschiedene Bereiche. 1) Die Lebensweganalyse zur Bewertung von Bioenergie wurde erweitert, damit alle, also insbesondere auch biogene, Treibhausgasemissionen und die Freisetzung aller reaktiven Stickstoffverbindungen berücksichtigt werden konnten. Die damit verbundenen Unsicherheiten und Auswirkungen auf den Klimawandel, sowie globale und regionale Umweltwirkungen wurden durch Sensitivitätsanalysen erfaßt. 2) Um bodenbürtige N_2O-Emissionen räumlich spezifisch zu ermitteln, wurden auf der Grundlage von multivariaten, linearen Regressionsrechnungen statistische Modelle entwickelt, die die jährlichen N_2O-Flussraten mit Hilfe von Steuergrößen aus Boden, Klima und Management abschätzen. Die zugrundeliegenden Messdaten wurden aus der gesamten in der EU erschienenen Literatur zusammengetragen. 3) Eine ursachenorientierte Methodik für Treibhausgasinventare wurde entwickelt, um Treibhausgasemissionen aus den verschiedenen Bereichen und Vorgängen in der landwirtschaftlichen Produktionskette zu ermitteln. Diese ermöglicht im Vergleich zur Standardmethodik des IPCC eine stärker management- und tierspezifische Berechnung. Genauer umfaßte die innovative Leistung: die Treibhausgasquellen von Mist- und Güllemanagement wurden gemäß der Quellorte Stall – Lager – Ausbringung differenziert; die CH_4-Oxidation in landwirtschaftlichen Böden wurde in die Bilanzen aufgenommen; bodenbürtige N_2O-Emissionen wurden regional differenziert mit den unter 2) entwickelten Gleichungen abgebildet. Mit Hilfe der neuen Methodik sowie von Agrarstatistiken, Bodenkarten und Klimadaten wurden die landwirtschaftlichen Treibhausgasemissionen regional differenziert auf Bundeslandebene für den Zeitraum 1975 bis 1997 berechnet. Die Ergebnisse wurden vor dem Hintergrund der gesamten anthropogenen Emissionsbilanz in Europa diskutiert und als Grundlage für die Abschätzung von Emissionsminderungspotentialen und deren Unsicherheiten verwendet.

Ergebnisse und Diskussion

Kohlenstoffspeicherung

Die Landwirtschaft Europas bietet ein erkennbares Potenzial für Kohlenstoffspeicherung, das durch eine optimale Kombination verschiedener Maßnahmen im Bereich von bodenschonender Bodenbearbeitung, Applikation organischer Rückstände und v.a. durch die Förderung von Waldstrukturen und durch Aufforstung maximal ausgeschöpft werden kann. Andererseits hat die Kohlenstoffspeicherung entscheidende Nachteile in längerfristiger Perspektive. Die Senkenfunktion der Biosphäre wird sich zunehmend sättigen oder sich gar zu einer Quelle umkehren, falls sich die klimatischen Bedingungen oder das Management der Flächen ändert. Daher kann Kohlenstoffspeicherung in der Biosphäre zwar dazu dienen, Zeit zu gewinnen, um technische Maßnahmen zu entwickeln und implementieren, aber sie kann nicht fossile Treibhausgasemissionen im Sinne einer vollwertigen Klimaschutzmaßnahme aufwiegen. Die europäische Landwirtschaft wird kurzfristig maximal um 200 Tg a^{-1} CO_2 aufnehmen können, wobei entsprechende Maßnahmen auch andere positive Umweltwirkungen bezüglich Bodenschutz, Biodiversität und Wasserhaushalt zeigen.

Kohlenstoffsubstitution

In der Fallstudie zur Bioenergie zeigt der Vergleich zwischen einer Wärmebereitstellung aus nachwachsenden und fossilen Energieträgern, dass Wärme aus Getreideganzpflanzen mit einer Verminderung der Freisetzungen von klimarelevanten Spurengasen im Vergleich zu Wärme aus leichtem Heizöl verbunden ist. Gleichzeitig kommt es aber zu Mehremissionen von Gasen mit versauernder und eutrophierender Wirkung sowie von Gasen, die zum Abbau der stratosphärischen Ozonschicht beitragen. Diese Umwelteffekte können positiver sein, wenn Getreideganzpflanzen extensiv produziert werden oder andere Minderungsmaßnahmen in Bezug auf die Emission von gasförmigen Stickstoffverbindungen im landwirtschaftlichen und anlagetechnischen Bereich ergriffen werden. Im Allgemeinen vermindern alle festen, flüssigen und gasförmigen Bioenergieträger den anthropogenen Treibhauseffekt und können somit einen deutlichen kurz- und langfristigen Beitrag zum Klimaschutz leisten. Emissionen von Treibhausgasen während der Produktion und Nutzung der Biomasse, v.a. von N_2O und reaktiven Stickstoffverbindungen, reduzieren allerdings den Bruttoeffekt der Kohlenstoffsubstitution um bis zu 30 %. Aus Umweltaspekten heraus erscheint daher der Einsatz von annuellen, intensiv erzeugten Biomassen wenig effizient, vielmehr sollten perenne Kulturen, insbesondere Bäume, sowie die Nutzung von Reststoffen, in Betracht gezogen werden. Das biologische Substitutionspotenzial der Bioenergie in der EU liegt zwischen 400 und 800 Tg a^{-1} CO_2-Äquivalente, wovon über die Hälfte direkt aus der Landwirtschaft stammt.

Treibhausgasminderung in der Landwirtschaft

Methodische Ergebnisse und Emissionsinventar

Bevor Treibhausgasminderungsmaßnahmen in der Landwirtschaft identifiziert und bewertet werden können, müssen die genauen Quellen regional differenziert bekannt sein. Daher wurde zuerst eine geeignete Methodik zur Quantifizierung der Treibhausgasflüsse in der europäischen Landwirtschaft entwickelt und die Emissionen für den Zeitraum 1975 bis 1997 berechnet. Auf der Grundlage von N_2O-Freilandmessungen in Europa wurden wesentliche Faktoren aus Boden, Klima und Management identifiziert, die die jährlichen N_2O-Flüsse aus landwirtschaftlichen Mineralböden Europas steuern. Zum Beispiel beeinflussen Düngung, Klima- und Bodenparameter, v.a. der Stickstoffgehalt des Oberbodens, die Emissionen von N_2O aus Ackerböden. Ackerböden weisen niedrigere durchschnittliche Emissionsraten in den atlantischen und trocken-kontinentalen temperaten Klimazonen Europas auf als in den feucht-kontinentalen und borealen. Daher wurden zwei verschiedene Regressionsmodelle entwickelt. Die Emissionen von N_2O in den atlantischen und trocken-kontinentalen Klimazonen liegen meist unter 2 kg N_2O-N ha^{-1} a^{-1} und übersteigen selten 5 kg N_2O-N ha^{-1} a^{-1}. Sie wurden am besten von einer Kombination aus Düngerstickstoff, Stickstoffgehalt des Oberbodens und einem Bodenfeuchteindex geschätzt. Im feucht-kontinentalen und borealen Raum schwanken die N_2O-Emissionen viel stärker zwischen 0 und 17 kg N_2O-N ha^{-1} a^{-1} und hängen vom verfügbaren Stickstoff, im Modell dargestellt als Düngerstickstoff und Stickstoffgehalt des Oberbodens, ab. Im Vergleich zu etablierten Methoden geben die entwickelten Regressionsmodelle auf regionaler Ebene die Messergebnisse genauer wieder, da sie außer der Düngung auch weitere, natürliche Steuergrößen berücksichtigen. Für Grünland wurde ein düngungsbasiertes Modell leicht verändert und an höhere Emissionen angepaßt. Aufgrund der großen Variabilität der Grünlandemissionen konnten keine weiteren Steuergrößen ins Modell integriert werden.

Auch für den gesamten landwirtschaftlichen Sektor wurde eine detaillierte Methodik zur Berechnung der Treibhausgasemissionen erstellt, die mit der Standardmethodik des IPCC kompatibel ist. Hier wurden Emissionsfaktoren und Regressionsgleichungen verwendet, die mit Messdaten aus Europa entwickelt wurden. Mit Hilfe der detaillierten Methodik wurden die biogenen Treibhausgasemissionen der Landwirtschaft in der EU und ihren Mitgliedsstaaten auf Bundeslandebene differenziert für den Zeitraum 1975 bis 1997 berechnet. Demnach emittierte die europäische Landwirtschaft im Jahr 1995 0.84 ± 0.29 Tg N_2O, 8.1 ± 1.9 Tg CH_4 und 39 Tg ± 25 CO_2, zusammengenommen 470 ± 80 Tg CO_2-Äquivalente oder 11 % der gesamten anthropogenen Treibhausgase der EU. Auf EU-Ebene liegen diese Ergebnisse erstaunlich dicht an denen, die in den offiziellen nationalen Berichten für die UNFCCC genannt werden. Im Vergleich zeigen sich aber auf nationaler Ebene deutlichere Unterschiede. Der hier gewählte Ansatz kommt zu höheren CH_4-Emissionen in Dänemark und den Niederlanden, höheren N_2O-Emissionen in Österreich, Finnland, Deutschland, den Niederlanden und – je nach Berichtsjahr – Schweden bzw. zu niedrigeren N_2O-Emissionen in Italien. In Ländern mit – sogar kleinen – Flächen von landwirtschaftlich genutzten Moorböden tragen auch CO_2-Emissionen aus der Torfoxidation signifikant zu den Gesamtemissionen bei, die in bisherigen Inventaren unberücksichtigt blieben. Im Vergleich zeigt sich, dass nur der hier gewählte Ansatz regionale und nationale Charakteristika der landwirtschaften Produktion adäquat abbilden kann. Er reduziert außerdem die Unsicherheit in den Inventaren auf die Hälfte der IPCC-Methodik. Die erzielten Inventarergebnisse stimmen gut mit den Ergebnissen aus inversen atmosphärischen Modellen überein. Dies legt den Schluß nahe, dass sich die hier entwickelte und angewandte Methodik als deutlich verbesserter Standard für EU-Mitgliedsstaaten eignen könnte, um auf nationaler Ebene Treibhausgasemissionen aus der Landwirtschaft zu schätzen.

Bewertung von Treibhausgasminderungsmaßnahmen

Auf der Grundlage des detaillierten Inventares wurden Minderungsmaßnahmen beschrieben und diskutiert. Vielversprechende technische Maßnahmen zur Treibhausgasminderung in der Landwirtschaft Europas zielen auf eine Extensivierung der Pflanzenproduktion durch reduzierten Einsatz von synthetischem Dünger und von Stickstoff im Allgemeinen bzw. auf technologische Innovation im Bereich der Tierhaltung, die möglichst von einem weiteren Abbau der Tierzahlen flankiert werden sollte. Die Wiedervernässung von gedränten, landwirtschaftlich genutzten Moorböden könnte das Potenzial zur Treibhausgasminderung verdoppeln, aber die Durchsetzbarkeit sowie die langfristigen ökologischen und sozioökonomischen Folgen bedürfen noch weiterer Forschung, bevor diese Maßnahme implementiert werden kann. Das Minderungspotenzial der technischen Maßnahmen liegt zwischen 100 und 200 Tg a^{-1} CO_2-Äquivalente. Sozioökonomische Veränderungen, wie etwa der menschlichen Ernährungsgewohnheiten und der Agrarmärkte, können auch stärkere Veränderungen in den landwirtschaftlichen Treibhausgasbilanzen verursachen. Ein effizienter Klimaschutz in der europäischen Landwirtschaft ist nur möglich, wenn er finanziell, politisch und durch Bildungsmaßnahmen stark unterstützt wird.