CLIMATE CHANGE

Scientific Assessment and Policy Analysis

WAB 500102 010

Assessment of the relation between climate and nitrogen related policy for the Dutch situation

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Report
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This study has been performed within the framework of the Netherlands Research Programme on Scientific Assessment and Policy Analysis for Climate Change (WAB) project ‘Assessing the relation between climate policies and nitrogen-related policies and the effect of (new) reactive nitrogen abatement on greenhouse gas emissions and sinks in the Netherlands and Europe’.

Wetenschappelijke Assessment en Beleidsanalyse (WAB) Klimaatverandering

Het programma Wetenschappelijke Assessment en Beleidsanalyse Klimaatverandering in opdracht van het ministerie van VROM heeft tot doel:

- Het bijeenbrengen en evalueren van relevante wetenschappelijke informatie ten behoeve van beleidsontwikkeling en besluitvorming op het terrein van klimaatverandering;
- Het analyseren van voornemens en besluiten in het kader van de internationale klimaatonderhandelingen op hun consequenties.

De analyses en assessments beogen een gebalanceerde beoordeling te geven van de stand van de kennis ten behoeve van de onderbouwing van beleidsmatige keuzes. De activiteiten hebben een looptijd van enkele maanden tot maximaal ca. een jaar, afhankelijk van de complexiteit en de urgentie van de beleidsvraag. Per onderwerp wordt een assessment team samengesteld bestaande uit de beste Nederlandse en zonodig buitenlandse experts. Het gaat om incidenteel en additioneel gefinancierde werkzaamheden, te onderscheiden van de reguliere, structureel gefinancierde activiteiten van de deelnemers van het consortium op het gebied van klimaatonderzoek. Er dient steeds te worden uitgegaan van de actuele stand der wetenschap. Doelgroep zijn met name de NMP-departementen, met VROM in een coördinerende rol, maar tevens maatschappelijke groeperingen die een belangrijke rol spelen bij de besluitvorming over en uitvoering van het klimaatbeleid.

De verantwoordelijkheid voor de uitvoering berust bij een consortium bestaande uit PBL, KNMI, CCB Wageningen-UR, ECN, Vrije Universiteit/CCVUA, UM/ICIS en UU/Copernicus Instituut. Het PBL is hoofdaannemer en fungeert als voorzitter van de Stuurgroep.

Scientific Assessment and Policy Analysis (WAB) for Climate Change

The Netherlands Programme on Scientific Assessment and Policy Analysis Climate Change has the following objectives:

- Collection and evaluation of relevant scientific information for policy development and decision–making in the field of climate change;
- Analysis of resolutions and decisions in the framework of international climate negotiations and their implications.

We are concerned here with analyses and assessments intended for a balanced evaluation of the state of the art for underpinning policy choices. These analyses and assessment activities are carried out in periods of several months to a maximum of one year, depending on the complexity and the urgency of the policy issue. Assessment teams organized to handle the various topics consist of the best Dutch experts in their fields. Teams work on incidental and additionally financed activities, as opposed to the regular, structurally financed activities of the climate research consortium. The work should reflect the current state of science on the relevant topic. The main commissioning bodies are the National Environmental Policy Plan departments, with the Ministry of Housing, Spatial Planning and the Environment assuming a coordinating role. Work is also commissioned by organisations in society playing an important role in the decision-making process concerned with and the implementation of the climate policy. A consortium consisting of the Netherlands Environmental Assessment Agency, the Royal Dutch Meteorological Institute, the Climate Change and Biosphere Research Centre (CCB) of the Wageningen University and Research Centre (WUR), the Netherlands Energy Research Foundation (ECN), the Netherlands Research Programme on Climate Change Centre of the Vrije Universiteit in Amsterdam (CCVUA), the International Centre for Integrative Studies of the University of Maastricht (UM/ICIS) and the Copernicus Institute of the Utrecht University (UU) is responsible for the implementation. The Netherlands Environmental Assessment Agency as main contracting body is chairing the steering committee.

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This report in pdf-format is available at www.pbl.nl

Preface

This report was commissioned by the Netherlands Programme on Scientific Assessment and Policy Analysis Climate Change (WAB).

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Abstract

In the framework of the Netherlands Programme on Scientific Assessment and Policy Analysis Climate Change (WAB-CC) the major drivers and interactions for flows of nitrogen and greenhouse gasses in the Netherlands were assessed. This was done to determine opportunities for synergism of climate and nitrogen policies, but also to identify possible risks for antagonisms. Because of the nature of this assessment, its conclusions may help in defining and prioritising future research and possible policy developments on the different topics.

In view of the complexity of the interactions between greenhouse gas (GHG) emissions and reactive nitrogen a division was made into five more comprehensive smaller subsystems. The subsystems were chosen in such a way that they are also relevant for (future) policies. These five subsystems and their research topics are:

1. Emission of NH_3, N_2O, CO_2 and fine particles in energy production
    - here the trends in and interactions between different GHG and nitrogen emissions are assessed, as well as the possibilities for specific measures and the effect of changing particle emissions on global warming.
2. Impact of N-fertilization on GHG emissions in agricultural systems
    - here the effects of climate change (temperature and precipitation) on the agricultural emission of GHG and N are assessed, as well as the effects of N measures on GHG emissions
3. Impact of biofuel use, and associated land use change, on GHG emissions in the energy sector and agriculture
    - here the impact of biofuel use (and its production) on the GHG emissions is assessed in the whole chain from fertilizer production to the conversion of bio fuels
4. Impact of N deposition and climate policies on greenhouse gas emissions in terrestrial ecosystems
    - here the effect (both qualitative and quantitative) of changing nitrogen depositions on GHG emissions in different terrestrial ecosystems (forested and non-forested) in Europe is assessed
5. Impact of N-inputs on GHG emissions in aquatic ecosystems
    - here the effect of changing nitrogen inputs (mainly via water) on the GHG emissions in aquatic ecosystems is assessed, as well as the effect of climate change on aquatic systems.

The main conclusions for each subsystem are:

1. Emission of NH_3, N_2O, CO_2 and fine particles in energy production
    - looking at the energy sector as a whole, the way in which energy is produced now and in the future is largely determined by fuel prices and availability of fossil fuels, but also by consumer behaviour (which can be influenced by changes in climate);
    - synergies and antagonisms between GHG and N emission reductions are possible in the energy productions sector, for instance by means of the use of sustainable energy sources and, more in general, by chosing different fuel types. The net effect is not always clear and should be investigated in more detail;
    - measures targeting at the reduction of GHG emissions can result in reduced emissions of aerosols. This is contra productive since fine particles can partly compensate global warming, due to their cooling effect. However, the regional impact of emission reductions on this effect is still poorly understood.
2. Impact of N-fertilization on GHG emissions in agricultural systems
    - Nitrogen use in the animal production sector (feed, production and storage of manure) and arable sector (use of fertilizer and animal manure) leads to substantial GHG emissions. Possibilities for lowering these emissions are: reduced nitrogen use and adjusted nitrogen management.;
    - NEC ammonia measures enhance N_2O emissions through mandatory manure incorporation (antagonism). At present there is no adequate method available to determine the net environmental effect with respect to low emission manure spreading (like e.g. manure incorporation);

- the nitrate directive has lead to 30% reduction of total N inputs, with direct co-benefits for climate via a reduction of N_2O emissions (synergy);
- (co-) digestion of manure can be a worthwhile measure to reduce methane and nitrous oxide emissions from manure storage, and to increase the fertilizing value of manure as compared to untreated manure.

3. Impact of biofuel use, and associated land use change, on GHG emissions in the energy sector and agriculture
   - Use of biofuels in decentralised units without emission reducing measures leads to higher NO_x emissions;
   - (co-) digestion is a widely promoted option to produce biogas and/or heat and power. However, there is a competition between animal feed and energy/biofuels. Although addition of plant material results in a higher energy output, it also results in higher GHG emissions;
   - increased biomass production yield higher N emissions in the existing cascade, as a result of the needed additional fertilizer;
   - 2nd generation biofuels is favourable in relation to energy/GHG balances. However, the (indirect) competition with food production for humans and animals still needs attention.
4. Impact of N deposition and climate policies on greenhouse gas emissions in terrestrial ecosystems
   - nitrogen deposition has overall a positive effect on the global warming potential (GWP) in terrestrial ecosystems because of the additional carbon uptake;
   - assessing the efficiency of existing of future measures related to reduced N use requires a full quantification of its effects on the exchange of CO_2, N_2O and CH_4 by complete life cycle analyses;
   - the overall impact of 1 kg N used in agriculture on the global warming potential may be negligible when the N is applied in the form of animal manure. When it is applied in the form of N fertilizer, it is likely to have an negative impact due to CO_2 and N_2O emissions during the production of fertilizer.
5. Impact of N-inputs on GHG emissions in aquatic ecosystems
   - both fresh waters and coastal zones are a net source of GHG, especially because of N_2O emission originating from leaching and run-off from agriculture;
   - the overall net effect of aquatic processes on the GWP is uncertain, since no quantitative information is available on the net carbon sequestration of the aquatic system.

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Nederlandse samenvatting

In het kader van het Nederlandse Programma Wetenschappelijke Assessment en Beleidsanalyse (WAB) Klimaatverandering zijn de belangrijkste interacties tussen de cycli van stikstof en broeikasgas onderzocht. Waar mogelijk en zinvol zijn resultaten gericht op de Nederlandse situatie. Het doel van het het onderzoek is om de mogelijkheden voor de synergie tussen klimaat- en stikstofbeleid te bepalen, maar ook om de mogelijke kansen op negatieve terugkoppelingen te identificeren. De complexiteit en de onzekerheid van de interacties blijken zeer groot. De conclusies van dit onderzoek bieden handvatten voor het definiëren en prioriteren van toekomstig onderzoek en mogelijk ook van beleidsontwikkelingen.

Vanwege de complexiteit van de interacties tussen emissies van broeikasgassen (BKG) en reactief stikstof, is een onderverdeling gemaakt in vijf deelsystemen. De deelsystemen zijn mede gekozen voor aansluiting op het (toekomstige) beleid. De onderzoeksonderwerpen per deelsysteem zijn:

1. Emissies van NH_3, N_2O, CO_2 en fijn stof bij de energieproductie
    - Hier zijn de trends en interacties tussen BKG en stikstofemissies onderzocht, maar ook de mogelijkheden voor specifieke maatregelen en het effect van veranderende emissies en concentraties van aërosolen op het broeikaseffect.
2. Invloed van stikstofbemesting op BKG emissies in landbouwsystemen
    - In dit systeem zijn de effecten van klimaatverandering (temperatuur en neerslag) op de emissie van BKG en stikstof vanuit de landbouw onderzocht, maar ook het effect van stikstofmaatregelen op de BKG emissies.
3. Invloed van het gebruik van biobrandstoffen en de bijbehorende verandering in landgebruik op de BKG emissies in de energie- en landbouwsector
    - De invloed van het gebruik (en de productie) van biobrandstoffen op de BKG emissie zijn onderzocht in de hele keten van kunstmestproductie tot de conversie van biobrandstoffen.
4. Invloed van stikstofdepositie en klimaatbeleid op BKG emissies voor terrestrische ecosystemen
    - Hier is het effect van veranderende stikstofdeposities op de BKG emissies in verschillende terrestrische ecosystemen (bos en overig) in Europa onderzocht.
5. Invloed van stikstofaanvoer op de BKG emissies voor aquatische ecosystemen
    - Het effect van veranderende stikstof aanvoer (voornamelijk via water) op de BKG emissies in aquatische ecosystemen is onderzocht, naast het effect van klimaatverandering op de kwaliteit van aquatische systemen.

De belangrijkste conclusies per deelsysteem zijn:

1. Emissies van NH_3, N_2O, CO_2 en fijn stof bij de energieproductie
    - voor de energiesector in zijn geheel hangt de manier waarop energie nu en in de toekomst wordt geproduceerd voornamelijk af van de brandstofprijzen (politiek) en de (fysieke) beschikbaarheid van fossiele brandstoffen. Ook veranderend consumentgedrag (welke mogelijk beïnvloed kan worden door veranderingen in het klimaat) is van belang;
    - in de energieproductiesector is er synergie en antagonisme mogelijk tussen emissiereducties van BKG en stikstof; bijvoorbeeld door de inzet van duurzame energie of, meer algemeen, door keuze voor andere brandstoffen. Het netto effect is niet altijd duidelijk en zou in meer detail onderzocht moeten worden;
    - maatregelen ter reductie van emissies van BKG kunnen leiden tot minder emissie van aërosolen. Dit is contraproductief omdat aërosolen het broeikaseffect compenseren door kun koelend effect. Echter, op regionale schaal is dit effect nog niet kwantificeerbaar.
2. Invloed van stikstofbemesting op BKG emissies in landbouwsystemen
    - Stikstofgebruik in veeteelt (ruwvoer en krachtvoer, productie en opslag van mest) en de akkerbouw (gebruik van kunstmest en dierlijke mest) leidt tot aanzienlijke BKG

emissies. Er zijn veel mogelijkheden om deze emissies te verminderen door minder stikstofgebruik en aanpassing stikstofmanagement;

- Het verplicht onderwerken van mest als maatregel om de ammoniakdoelstelling van de NEC richtlijn te halen leidt tot een verhoging van de N_2O emissies (antagonisme; ook voor nitraat). Er is nog geen goede methode om de netto milieuwinst van emissiearme aanwending te bepalen;
- De Nitraatrichtlijn heeft daarentegen geleid tot een reductie van de totale stikstof aanvoer van 30%, met bijkomende voordelen voor het klimaat door reductie van N_2O emissies (synergie);
- (co-) vergisting van mest kan een effectieve maatregel zijn voor het reduceren van emissies van methaan en lachgas vanuit mestopslagen. Tevens levert vergisting een residu met een betere bemestende waarde dan onbewerkte dierlijke mest.

3. Invloed van het gebruik van biobrandstoffen en de bijbehorende landgebruiksverandering, op de BKG emissies in de energie- en landbouwsector
   - de toepassing van biobrandstoffen in kleinere regionale eenheden zonder emissiereducerende maatregelen zal leiden tot hogere NO_x emissies;
   - (co-) vergisting is een breed gepromote optie voor het produceren van biogas en/of warmtekrachtkoppeling. Er is een trade-off tussen het gebruik van plantenresten voor (co-) vergisting en als diervoer. Verder heeft toevoeging van plantaardig materiaal weliswaar een hoger energierendement tot gevolg, maar tevens een hogere BKG emissie;
   - een verhoogde biomassa productie zal leiden tot hogere stikstof emissies in de bestaande cascade als gevolg van de extra benodigde (kunst)mest;
   - tweede generatie biobrandstoffen verdienen de voorkeur gelet op energie- en BKG balansen. Echter, gelet moet worden op de (indirecte) competitie met voedsel productie voor mens en dier.
4. Invloed van stikstofdepositie en klimaatbeleid op BKG emissies voor terrestrische ecosystemen
   - over het algemeen heeft stikstofdepositie een netto positief effect op de ‘global warming potential’ (GWP) in terrestrische systemen vooral door de extra CO_2 opname;
   - om de effectiviteit van bestaande en toekomstige maatregelen gerelateerd aan reductie van stikstofgebruik te bepalen, is een complete kwantificering van de effecten op de uitwisseling van CO_2, N_2O en CH_4 nodig door middel van een ‘ketenanalyse’ (LCA);
   - Een analyse van de hele keten laat zien dat het effect van bemesting van 1 kg stikstof in de vorm van dierlijke mest waarschijnlijk verwaarloosbaar is op de GWP. Wanneer direct toegediend als kunstmest, heeft het waarschijnlijk een negatieve invloed ten gevolge van CO_2 en N_2O emissies tijdens de productie van het kunstmest.
5. Invloed van stikstofaanvoer op de BKG emissies voor aquatische ecosystemen
   - zowel zoete oppervlaktewateren als kustzones zijn een bron van BKG, vooral vanwege de N_2O emissies, die vooral afkomstig zijn uit- en afspoeling uit de landbouw;
   - het totale netto effect van aquatische processen op de GWP is onzeker. Dit komt doordat er geen kwantitatieve informatie beschikbaar is over de netto koolstofvastlegging in aquatische systemen.

Naast de bovenstaande specifieke conclusies worden ook enkele algemene inzichten gegeven. Deze richten zich met name op de belangrijkste relaties (zowel positief als negatief) tussen stikstof- en klimaatbeleid, die op dit moment nog niet verankerd zijn in dat beleid. De observaties zijn gebaseerd op de inzichten verkregen uit de onderliggende studies en vormen toekomstige belangrijke aandachtsgebieden.

Observatie 1 Grootschalige teelt biobrandstoffen

Het gebruik van biobrandstoffen en bioenergie vraag om meer kennis voordat het op grote schaal kan worden toegepast, vanwege (i) de onzekerheid met betrekking tot het netto CO_2 effect, (ii) de negatieve neveneffecten voor de stikstofkringloop en (iii) de competitie met de vraag naar land voor natuur en/of voedselproductie. Dit is vooral het geval voor de zogenaamde eerste generatie brandstoffen. Negatieve neveneffecten op de stikstofkringloop zijn de additionele emissies van N_2O en NH_3 en een toename van de stikstof uitspoeling naar aquatische systemen door een toename van kunstmestgebruik. Ook is er sprake van additionele NO_x emissies tijdens de productie en de conversie van biobrandstoffen zoals bij decentrale warmte-kracht toepassing. Er is een duidelijke behoefte aan integrale assessments die alle effecten in de hele keten gerelateerd aan het gebruik van biobrandstoffen onderzoeken.

Observatie 2 Bevordering koolstofopslag in natuur door stikstofbemesting

Het stimuleren van koolstofopslag in organisch materiaal van (semi) natuurlijke ecosystemen door een verhoogde aanvoer van stikstof is een potentieel relevante, maar tijdelijke, optie om BKG emissies te compenseren. Het gebruik van een dergelijke optie als onderdeel van klimaatbeleid conflicteert met stikstofbeleid, omdat een toename (of lagere ambities voor reductie) van de productie van reactief stikstof door de productie van kunstmest en het verbranden van fossiele brandstoffen zal leiden tot extra effecten, zoals vermesting, verzuring, luchtverontreiniging, etc. Een complete kwantificering van de effecten van stikstofgebruik op de uitwisseling van CO_2, N_2O en CH_4 door middel van een ‘ketenanalyse’ (LCA) geeft meer inzicht in de meest optimale toepassing van stikstof in de landbouw met het oog op het netto BKG effect.

Beleid gericht op het bevorderen van koolstofvastlegging in bossen, (bijvoorbeeld) door het langer toestaan van de huidige hoge N depositieniveaus, impliceert dat een groter gewicht wordt toegekend aan schade voor de samenleving door klimaatverandering dan door eutrofiëring. De wetenschap kan echter op dit moment nog geen geschikte methodieken leveren voor het ondersteunen van een dergelijke beslissing. Maatschappelijke schade door klimaatverandering en eutrofiering vloeien beiden voort uit biodiversiteitverandering, maar een correcte afweging vereist dat deze schade (en andere schadelijke effecten) worden uitgedrukt in vergelijkbare eenheden.

Observatie 3 Noodzaak en ruimte voor het reduceren van stikstof kunstmest

Gegeven de negatieve effecten van overbemesting, zowel in termen van eutrofiëring, verzuring, grootschalige luchtverontreiniging, waterverontreiniging als van klimaatverandering (broeikasgasemissies), is het zeer relevant om de productie en het gebruik van kunstmest te optimaliseren. De emissie van N_2O tijdens de productie van kunstmest kan worden verminderd door de implementatie van bestaande procestechnologieën. Er zijn ook verschillende opties om de efficiëntie bij het gebruik van kunstmest te verhogen. De huidige opvatting in de landbouw is dat, wanneer goedkope kunstmest beschikbaar is, het gebruik van grote hoeveelheden stikstof een goede optie is om het risico van lagere winst bij verminderde gewasopbrengsten te verlagen. Wanneer echter ook de maatschappelijke schade door klimaatverandering en stikstofgerelateerde effecten beschouwd worden zal het economische optimale bemestingsniveau lager zijn. Met name in geïndustrialiseerde landen zijn maatregelen gericht op een vergroting van de kunstmestefficiëntie effectief bij de mitigatie van klimaatverandering, en andere stikstofgerelateerde effecten, zonder bijkomende risico’s voor de voedselproductie.

Observatie 4 Integraal stikstofbeleid vereist meer inzicht in kosten en baten

Het is momenteel niet goed mogelijk om de kosten en baten van verschillende opties voor stikstof- en klimaatbeleid tegen elkaar af te wegen. Hoewel sommige effecten wel kunnen worden gemonetariseerd is de bijdrage van stikstof niet altijd te kwantificeren. Dit is bijvoorbeeld het geval voor koolstofvastlegging door verhoogde stikstoftoevoer, de bijdrage van stikstof aan de globale koeling door aërosolen, de uitwisseling tussen ammonia en lachgas, bijdrage aan biodiversiteitsverlies, de keten van biobrandstoffen, etc. Er is meer onderzoek nodig om te komen tot een gemeenschappelijke basis voor de vergelijking van verschillende maatregelen en voor het kwantificeren van schade per eenheid stikstof voor de verschillende stikstof-componenten.

Summary

The overall objective of this assessment was to investigate the relation between climate policies and nitrogen related policies and (where possible) to determine the effect of (new) reactive nitrogen abatement on greenhouse-gas emissions and sinks in the Netherlands. Although there are different procedures possible when conducting an assessment, we did this by summarizing and synthesis of existing material on this subject. Because of the nature of this assessment, its conclusions may help in defining and prioritising future research on the different topics. In general we tried to identify the major drivers and interactions for flows of nitrogen and greenhouse gasses in the Netherlands in order to determine:

- the opportunities for synergism of climate and nitrogen policies
- the risks for antagonism of climate and nitrogen policies.

Investigating the 'climate - nitrogen' system showed different interlinkages related to nitrogen and greenhouse gasses that are presented in the following overview (Figure I).

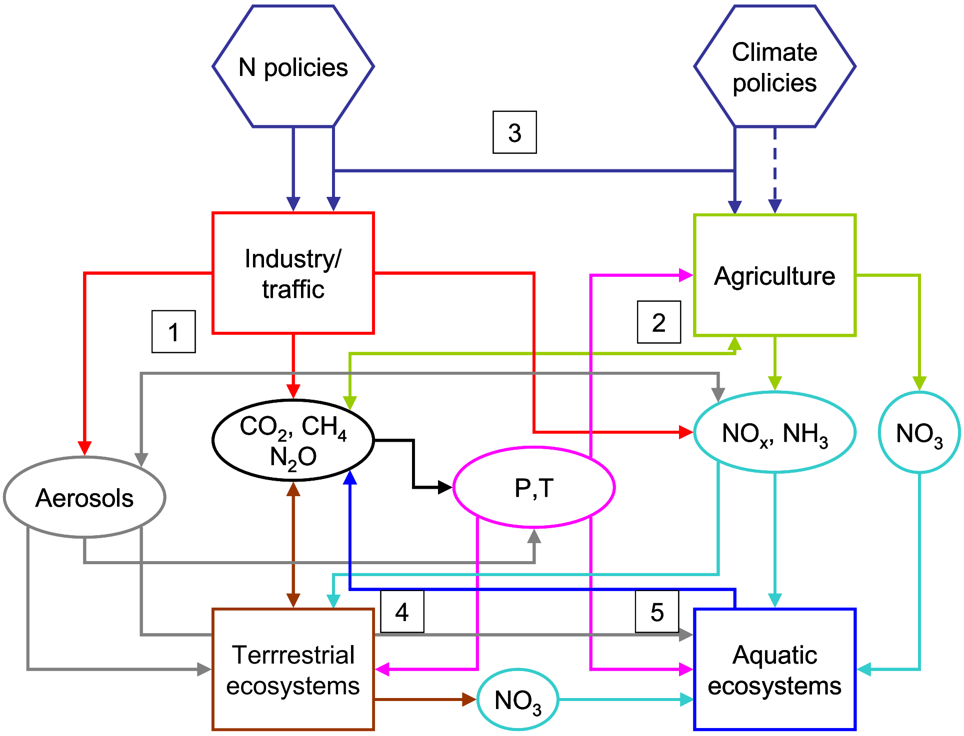

Figure I Overview of the main interlinkages between nitrogen and greenhouse gases assessed here (explanation of the numbers is given in the text below).

In general the major producers of reactive nitrogen are also major greenhouse gas (GHG) emitters: industry (including energy), traffic and agriculture. Industry and traffic are responsible for more than 90% of the emission of CO_2 equivalents in the Netherlands, while the nitrogen emission from these source categories mainly consists of the emission of NO_x (90% of total Dutch NO_x emission). Agricultural activities contribute relatively more to the emission of nitrogen than to GHG. However, due to the interactions between the nitrogen and carbon cycle, agriculture is important both as source and sink of GHG. On the receptor side terrestrial and aquatic (eco)systems (including coastal waters) are important sinks for nitrogen and GHG emissions. At the same time these systems can also act as sources and therefore complicating relations between nitrogen and GHG considerably.

Five comprehensive smaller subsystems were identified for which the interactions between Nitrogen and GHG-emission could me more easily assessed and which are also relevant domains for (future) policies (denoted by the numbers in Figure I):

1. Emission of NH_3, N_2O, CO_2 and fine particles from **energy production**
2. Impact of N-fertilization on net GHG emissions in **agricultural systems**
3. Impact of **biofuel use**, and associated **land use change**, on N and GHG emissions in the energy sector and agriculture
4. Impact of **N deposition** and climate policies on net GHG emissions in **terrestrial ecosystems**
5. Impact of N-inputs on net GHG emissions in **aquatic ecosystems**

Apart from the first section, the following interactions were taken into account:

- Emissions of CO2 and N2O related to the production of nitrogen fertilizer (Chapter 3);
- CO2 sequestration in agricultural soils due to N input (fertilizer and manure) and in terrestrial ecosystems due to N deposition in response to elevated NH3 and NOx emissions (Chapter 5);
- N2O exchange from agricultural soils (Chapters 3 and 4), non agricultural soils (Chapter 5) and re-emission from aquatic systems in response to changed N input and N deposition (Chapter 6);
- CH4 emissions from animals, stored manure, agricultural soils and terrestrial ecosystems in response to changed N input and N deposition.

Section 1: Emission of NH_3, N_2O, CO_2 and fine particles in energy production

The way in which energy is produced is largely determined by oil price and availability of fossil fuels, while the amount of energy that is used is depending on e.g. additional factors such as developments in consumer behaviour (which in turn is also related to the energy price). Presently this behaviour can be influenced by changes in climate (e.g. higher temperature - less heating, but more air conditioning). The energy production sector has contributed to issues like acidification through the emission of SO_2 and NO_x. In the past decades a large suite of measures has been implemented Europe wide to reduce the negative effects caused by these emissions. The effect has been a reduction of 81 and 34% in European SO_2 and NO_x emissions between 1990 and 2005, respectively. Looking at the energy production sector, the reductions have been achieved mainly by fuel switch and end-of-pipe techniques. NO_x measures included improved efficiencies e.g. high efficiency co-generation, fuel injection at gasoline engines; reducing air excess in industrial installations and Selective Catalytic Reductions (SCR). Side-effects of these measures include additional fuel use of 3% by cars and ~1% for refineries and coal based power stations.

Synergies between GHG and N emission reductions are possible in the energy production sector where NO_x and CO_2 can be reduced simultaneously by e.g. sustainable sources and energy saving. In some cases also reduction of SO_2 and particulate matter is possible at the same time. However, it should be noted that antagonisms are also possible between measures focussing at specific compounds. This is especially true for the use of biofuels and bio-energy (see also following sections), but also for e.g. the earlier mentioned additional energy use through end of pipe techniques.

For the energy sector most of the new emission reduction measures focus on CO_2 emission reduction (with a possible simultaneous reduction of NO_x and SO_2). Within the Option document on Energy and Emissions 2010/2020 (Daniels & Farla, 2006) it is concluded that the CO_2 reduction potential for The Netherlands to reach NO_x (and SO_2) targets is about 10 Tg CO_2. However, to reduce CO_2 emissions with 30% in 2020 (being the new Dutch government target) energy saving, CO_2 capture and storage, nuclear power and renewables will all be necessary. All of these options at the same time will reduce NO_x emissions, except some options for bio-energy. Especially the decentralised bio-energy options will increase NO_x emissions, because it is not cost-effective to install expensive SCR technology.

So far, there are only a few new options targeting NO_x and SO_2 emissions directly, such as increased efforts in the emission trading for energy producers. In general the quantification of coupled $CO_2/NO_x/SO_2$/PM emission reductions is possible, but requires further investigation.

A possible side-effect of the use of biofuels in e.g. co-firing systems is a change in the size-distribution of aerosols being emitted during the process. Some indications show that there is a shift towards smaller aerosols that may give a larger contribution to different human health issues. Aerosols not only contribute to these human health issues, but also have a cooling effect, compensating part of the global warming. In this context aerosols have a direct and indirect effect: direct because of the adsorption of radiation which will thus not reach the earth surface and indirect because aerosols form the cloud condensation nuclei as a basis for clouds, which also prevent radiation to reach the earth surface. Measures to reduce NO_x and/or NH_3 limit the formation of ammonium nitrate aerosol that is responsible for a cooling of about 2.5 $W.m^{-2}$ in the Netherlands region (sum of direct and indirect). This cooling effect cannot directly be translated into a temperature change, because the relations are complex (temporal variations in aerosol concentrations and sizes, feedback mechanisms, interactions, etc.). Furthermore, the regional climate impact of aerosols is still poorly understood and it is not known if there is a positive or negative impact. It is advisable to quantify these side-effects of emission reductions and adopt the degree of implementation of policies/measures related to air pollution to new measures to reduce GHG emissions and vice versa.

Section 2: Impact of N-fertilization on GHG emissions in agricultural systems

The processes and issues under this section also apply to the next section on biofuels. The difference is made by the policies affecting N and GHG; cultivation of biomass for generation of fuels and electricity is mainly driven by climate policies, while conventional agriculture is mainly driven by food and feed demand. This section mainly focuses on the effects of N-measures on GHG emission:

- N-measures following from implementation of the EU National Ceilings directive for ammonia and the EU Nitrates directive,
- Production and use of chemical fertilizer for food, fodder and fibers.

Effects of climate change (temperature and precipitation) on the N and C cycle in relation to GHG emission and N-losses for agro-ecosystems can also be substantial but are hardly quantified for the Dutch situation and need further attention. Effects of climate measures on N-emissions in agriculture were not considered.

The agricultural system and the GHG and N flows can be schematised as shown in Figure II.

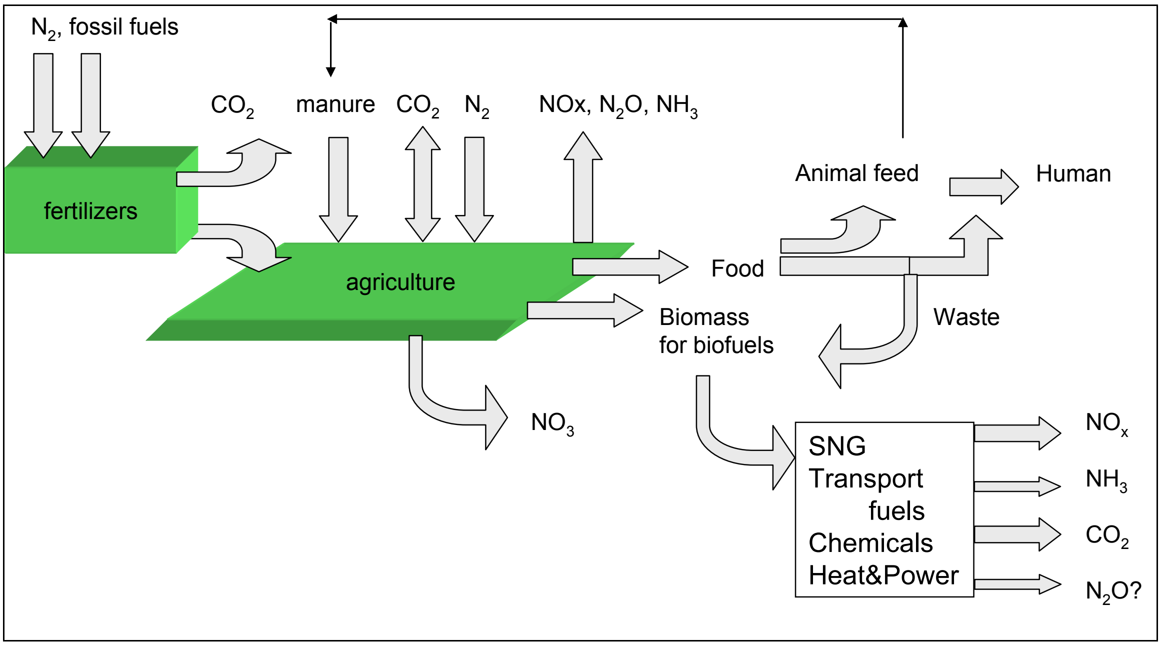

Figure II Nitrogen cascade through the agricultural system.

The figure shows the cascade of nitrogen through the system, starting with the production of chemical fertilizer and ending in food products for animals/humans and in energy application (bio-energy). During the cascade the nitrogen losses are apparent and there are different interactions with the emission of GHG. Following the cascade, the losses and interactions become more uncertain in terms of quantification and comprise a much broader scale in time and space. N_2O is a large contributor to the overall GHG balance for food and bio-energy. N_2O is emitted during fertilizer production, application and further in the cascade (producing animal food, manure, losses to groundwater and atmosphere, etc.).

Even though the start of the cascade seems reasonably simple there are different estimates of the emissions of CO_2, N_2O and NH_3 along with fertilizer application. The total GHG emissions range from about 1 to 12 kg CO_2-eq.kg^{-1} N in the final agricultural products, the contribution of N_2O to the total GHG emission is highest and most uncertain. This is also complicated by the fact that emission factors for N_2O are constantly being revised. The current range is so large that in Life Cycle Analysis or Well to Wheel analysis it is uncertain whether cultivation of energy crops are a net source or sink of GHG! It is therefore very effective to abate N_2O emissions from fertilizer production. However, in general it can be stated that agriculture has substantial and partly evitable net GHG effects both by the arable sector (fertilizer) and animal sector (enteric fermentation and manure storage)

Current measures to reach the EU National Emission Ceilings for ammonia have a negative impact on N_2O emissions through mandatory manure incorporation, implying the need of a policy evaluation weighing the ammonia-ecology benefits against the GHG effect of additional nitrous oxide emission. The nitrate directive has lead to 30% reduction of total N inputs to Dutch agricultural soils and has direct co-benefits for NH_3 emissions and for climate via reduction of N_2O emission. Further reduction to increase nitrogen efficiencies is possible. Ideally the approach of marginal gain of Euros or CO_2-eq emission should be used to find the optimal fertilizer dose, including damage costs (and benefits) of the GHG-effects. When taking a national perspective (as opposed to a farmers perspective), the very likely outcome would be that the most appropriate fertiliser dose would be substantially lower than present N-fertilizer recommendations, for both food/fibre and energy crops.

Effective measures for agricultural production to limit nitrogen pollution should be aimed at reduction of the creation/use of reactive nitrogen, and will require increasing the nitrogen use efficiency in the various agricultural systems. If the nitrogen use efficiency can be increased without affecting the yield of crops, there is no net effect on the carbon sequestration in soils. If the yield is smaller, less CO_2 will be sequestered. However, the overall nutrient availability also determines the distribution between tradable and non-tradable (e.g. roots and stems) yield. At lower nutrient availability plants will have to invest more in the non-tradable parts compared to the tradable parts, acting as a potential carbon sink when left at the fields.

A worthwhile measure to reduce CH_4 and N_2O emissions from manure handling seems (co-) digestion of manure. This will increase the fertilizing value of manure as compared to chemical fertiliser at the same time. Focusing too much on biogas and energy generation may lead to unsustainability if this would reduce replacement of chemical fertilizer by manure and reduce the organic matter supply to the soil. Furthermore, irrespective of applying co-digestion, if feed stuff is imported, animal production likely will enhance depletion of nutrients and carbon of soils in the regions were the feed is produced. It is questionable if there is net GHG-emission reduction when (in particular poultry) manure is incinerated. As all N is inevitable lost and P is not recycled to agriculture, this loss of N and P would have be compensated by increasing use of chemical N and P-fertiliser (either here or in the region where the feed stuff was important from).

Section 3: Impact of biofuel use, and associated land use change, on GHG emissions

The stimulation of bio-energy will increase nitrogen losses because of increased fertilizer use, similar to the nitrogen-food cascade. Some energy crops might need more fertilizer than food because they most likely will be produced on marginal land used for agricultural or nature, and production will be driven by higher ton biomass per ha with a low fertilizer price compared to a high (biofuel) yield and price. Policies to stimulate bio-energy should focus on the sustainability criteria. Furthermore, food production and imports, especially for animal feed should be regulated using the same sustainability criteria as those for bio-energy, to prevent trade-off of environmental side effects of large scale biomass production from industrialized countries to developing regions!

Digestion or co-digestion of manure primarily is a climate measure. However, as it also generates new residues and products, it also affects implementation of the EU Nitrates directive and NEC ammonia directive. Digestion (with or without co-substrates) of manure can be a worthwhile measure to reduce methane from manure storage, and to increase the fertilizing value of manure as compared to chemical fertilizer, while producing energy (biogas or heat/power). At the same time digestion will reduce the manure volume and may change field application procedures, which may have an effect on application emissions of ammonia and nitrous oxide. Focusing too much on biogas and energy generation may lead to unsustainability if this would reduce replacement of chemical fertilizer by manure and reduce the organic matter supply to the soil. Addition of additional substrates (e.g. silage maize) to manure for co-digestion increases the energy yield, but is less attractive to reduce GHG emissions. Digestion of manure alone is less attractive for energy production, because the yields (and revenues) are much smaller, but the reduction of GHG emissions is much higher. While (co-)digestion is a widely promoted option to produce biogas and/or heat and power, it should be noted that this may lead to a competition between animal feed (silage maize) and energy/biofuels. The use of 2^{nd} generation biofuels is then favourable in relation to energy/GHG balances. However, the competition with animal feed still needs attention.

When opting for a more decentralised bio-energy use for heat and power higher NO_x emissions may be the result. This is in contrast to large scale production facilities where possibilities exist for de-NO_x and SCR installations and fuel use with possibilities for catalytic converters.

Table I gives an overview of several mitigation options in agriculture that have an effect on N and GHG.

Table I Mitigation options for the agricultural sector with an effect on N and GHG.

		Mitigative effects					
Measure	**Examples**	**CO_2**	**CH_4**	**N_2O**	**NO_x**	**NH_3**	**NO_3**
Cropland management	Agronomy	+		+/-	+/-	-	-
	Nutrient management	+		+	+	+	+
	Tillage/residue management	+		+/-	+/-	+/-	+/-
	Water management	+/-		+	+/-	+/-	+/-
	(irrigation/drainage)	+		+/-	+/-	+/-	+/-
	Agro-forestry	+	+	+	+	+	+/-
	Set-aside, land-use change						
Grazing land management / pasture improvements	Grazing intensity	+/-	+/-	+/-	+/-	-	-
	Increased productivity (e.g.	+		+/-	+/-	-	-
	fertilization)	+		+/-	+/-	+/-	+/-
	Nutrient management	+	+	+/-	+/-		
	Fire management	+		+/-	+/-		
	Species introduction (including legumes)						
Management of organic soils	Avoid drainage of wetlands	+	-	+/-	+/-		
Restoration of degraded lands	Erosion control, organic amandments, nutrient amendments	+		+/-	+/-		
Livestock management	Improved feeding practices		+	+	+	+	+
	Specific agents and dietary		+			+/-	+/-
	additives		+	+	+	+	
	Longer term structural and management changes and animal breeding						
Manure/biosolid management	Improved storage and handling		+	+/-	+/-	+	+
	Anaerobic digestion		+	+/-	-	+	+
	More efficient use as nutrient source	+		+	+	+	+
SCR options in (fertilizer) industry	De-NO_x, N_2O abatement	-	+/-	+	+	-	+/-
Sustainable energy	Wind mills, solar panels, energy saving, etc.	+	+/-	+/-	+	+/-	+/1
Bio-energy	Energy crops, solid, liquid, biogas, residues	+	+/-	+/-	-	-	+/-

+ denotes reduced emissions or enhanced removal (positive mitigative effect);
- denotes increased emissions or suppressed removal (negative mitigative effect);
+/- denotes uncertain or variable response

Section 4: Impact of nitrogen deposition on greenhouse gas emissions in terrestrial ecosystems

An increase in N deposition affects GHG exchange from these systems as it may lead to increased CO_2 sequestration, due to enhanced biomass production, but also to enhanced N_2O emission and a reduced CH_4 sink.

The productivity of many temperate ecosystems is nitrogen limited. Adding N via deposition thus has the potential to increase growth, and therefore to sequester CO_2 from the atmosphere. Besides by increased growth, an increase in N deposition on forests may increase C sequestration by an increased accumulation of soil organic matter. An increased rate of soil organic matter accumulation in response to increased N deposition may be due to (i) stimulation of growth and subsequent increased leaf/needle biomass and litter production and (ii) reduced long-term decomposition rates of organic matter, caused by an increased recalcitrance of N-enriched litter. Recent estimates range from a net carbon sequestration of 725 kg C per kg N added[1] (as wet deposition) to 428 kg C per kg N added[2] (as total deposition). Atmospheric

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[1] Magnani et al (2007) The human footprint in the carbon cycle of temperate and boreal forests. Nature 447, 849-851.

nitrogen deposition is also one of the key factors influencing N_2O emissions together with the soil C/N ratio.

An increased availability of nitrogen for the microbial processes of nitrification and denitrification, due to additional nitrogen deposition, will result in an enhanced N_2O emission from forest soils due to elevated nitrification and denitrification. Not only deposition is responsible for this increase; in many commercial forests nitrogen additions in the form of fertilizer N (urea or ammonium nitrate) can result in a doubling of the N_2O emission rates from forest soils. The indirect N_2O emission from surface waters may also be increased by the increased input of nitrogen via deposition, because it affects the nitrogen leaching/runoff from forests.

The relation between increased nitrogen availability and methane ($CH_{4)}$ exchange is still poorly understood. The mechanisms of CH_4 exchange differ from site to site, tipping the balance from a net source to a net sink of CH_4.

In this WAB study the quantification of the effect of additional nitrogen deposition on the GHG emissions in terrestrial ecosystems focused on European forest ecosystems. This is mainly because much of the research addressing this relationship was targeted on forested ecosystems. After estimating the, sometimes, counteracting relationships between nitrogen availability and GHG exchange, an overview can be given of the impact of nitrogen deposition on the net emission of the different GHG by European forests. This overview is listed in Table II, where estimated ranges in the long term annual average GHG emissions and the impact of nitrogen deposition on those emissions is given (together with changing GWP for the different GHG).

Table II Overview of the impact of nitrogen deposition on the net GHG emission by European forests.

GHG	Emission change (kg.ha^{-1}.yr^{-1}) per kgN		GWP change (kg CO_2-eq^{-1}.ha^{-1}.yr^{-1}) per kgN	
CO_2-C	-21 to -33	(-26)	-78 to -118	(-100)
N_2O-N	+0.018 to +0.039	(0.029)	+ 7.1 to +17,9	(12.5)
CH_4-C	+0.001 to +0.018	(0.0093)	+ 0.036 to +0.39	(0.21)

1 The N deposition impacts are given for an estimated increase in total N deposition of 1 kg N.ha^{-1}.yr^{-1} (and 0.43 kg NH_4-N.ha^{-1}.yr^{-1})

From the results in Table II it can be concluded that the GWP of CO_2 sequestration in response to nitrogen deposition is clearly outweighing the GWP of increased N_2O emissions and that the effect on the CH_4 sink can be ignored. However, CO_2 sequestration as a result of increased N-deposition can be regarded as a positive side-effect of a negative environmental change. N-deposition in itself might have other environmental impacts, such as acidification, eutrophication, loss of biodiversity, NO_3 pollution of ground water, etc.

A similar conclusion can be drawn for other ecosystems, based on the ratio of carbon sequestration to N input, which mainly determines the balance between N input and GHG exchange. The below-ground C sequestration, which is the ultimate sink for C, is comparable in forests and other terrestrial ecosystems, such as moorlands and heathlands. However, the positive relationship between N deposition and C sequestration in moorlands is debatable. Recent research in European peat bogs showed that N deposition promotes C losses.

Quantification of all above pathways shows that, when looking at the total chain from source to receptor (i.e. production, use, emission and subsequent deposition on e.g. nature areas), the overall impact of 1 kg N used in agriculture on the GWP (in kg CO_2-eq) may be negligible when the N is applied in the form of animal manure. When it is applied in the form of N fertilizer, it is

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[2] De Vries et al. (2008) Ecologically Implausible Carbon Response? Nature 451, E1-E3. / Sutton et al. (2008) Uncertainties in the relationship between atmospheric nitrogen deposition and forest carbon sequestration. Global Change Biology 14, 2057-2063

likely to have a negative impact on the GWP due to the CO_2 and N_2O emissions during the production of fertilizer which are, however, highly uncertain.

Assessing the efficiency of existing or future measures related to reduced N use requires a full quantification of its effects on the exchange CO_2, N_2O and CH_4 by complete life cycle analyses, including: (i) release of CO_2 and N_2O related to the production of N fertilizer, (ii) C sequestration in agricultural soils in response to a change in N input, (iii) C sequestration in terrestrial ecosystems (forests) in response to a change in N (NH_3) input induced by the change in N use, including on-site and off-site sequestration and (iv) N_2O exchange from agricultural soils, non agricultural (forest) soils and re-emission from aquatic systems.

Section 5: Impact of N-inputs on GHG emissions in aquatic ecosystems

The role of aquatic systems in the exchange of GHG has not been investigated as thoroughly as compared to terrestrial systems, but it may be as important. When looking at aquatic systems a distinction can be made between upland fresh water lakes/streams, reservoirs, rivers, estuaries, seas/oceans. Obviously both N- and GHG- flows in these different aquatic systems are connected, but integrated studied are rare. The main focus of this section was on fresh water systems, although the other systems are also touched upon. Important drivers for the overall exchange of GHG from aquatic systems are the residence time and the load of both nitrogen and carbon. The residence time of a system strongly regulates the N/C processes within aquatic systems. An example of this is the nitrogen removal from the system. This will be enhanced at longer residence times through nitrification and denitrification, immobilization and/or retention. The nitrification and denitrification processes are important sources of N_2O emission.

It is known that aquatic systems can both act as a source or a sink of GHG, depending on the type of GHG (CO_2, CH_4, N_2O) and of the system. When looking at fresh waters and coastal zones, evidence is available to suggest that these systems are a net source of GHG. The total anthropogenic contribution to the GHG emission through surface waters is relatively small. There is a clear relation between the N_2O emission and the anthropogenic contribution to these systems through waste water and agricultural N losses.

When assessing specific measures targeting at reduction of N loads to aquatic systems, it is clear that N_2O and possibly also CH_4 emissions will be decreased. However, at the same time the decrease in nitrogen loads may result in a decrease of the CO_2 sequestration. Due to these counteracting effects, the total effect of nitrogen measures on the GWP is uncertain. A complicating factor in determining the overall effect on the GWP is the fact that no quantitative information is available on the net CO_2 sequestration of aquatic systems (neither actual, nor past and future situation). Table III shows the quantification of a reduction in nitrogen load on N_2O, CH_4 and CO_2 emissions from aquatic systems.

Table III Quantification of effect of changing N load on N_2O, CH_4 and CO_2 emissions from aquatic systems.

Driver	**Effect**	**N_2O**	**CH_4**	**CO_2**
Reduction fertilizer use	Less Nin	++ [1)]	?	-
Reduction N deposition	Less Nin	++	?	-
Reduction animal numbers (manure)	Less Nin and Norg	++	?	?
Improvement wastewater treatment	Less Nin and Norg	++	+	?

[1)]. ++ = a strong reducing effect; + = a reducing effect; +/- = an uncertain effect; - = an increasing effect; -- = a strong increasing effect; ? = uncertain effect

The effect of climate change on the emissions of GHG from aquatic systems is shown in Table IV. Although the effects of climate change on GHG emission from surface waters might be large, the net effect is uncertain. Possible measures addressing different water quality and quantity issues may counteract: higher discharges in spring and autumn yield a decrease in GHG, whereas anthropogenic water management measures to regulate discharge and to prevent (summer) drought may increase GHG emissions.

Table IV Quantification of climate change on GHG emission from aquatic systems.

Effect CC	Effect	N_2O	CH_4	CO_2
Temperature increase	Decomposition > Degassing >	++	+/-	++
Higher discharge	Not relevant	Not relevant	Not relevant	Not relevant
Shorter up stream residence time	Methonagesis > Biomass > Degassing >	+	++	++

Another way nitrogen can influence the climate system is via the formation of aerosols in the marine system. The cooling effect of aerosols was already mentioned when discussing the emissions from energy production. The marine system can also contribute to this cooling effect by the production of dimethylsulphide (DMS), which is a result of the decomposition of demethylsulphoniopropionate (DMSP). DMSP is a biogenic product in seawater formed by phytoplankton and the diffusive transfer of DMS from sea-to-air is known to be the most important source of natural non-sea salt sulphur in the atmosphere. The relation between nitrogen and climate via the DMSP-DMS pathway can be described in short as follows: more nitrogen - more phytoplankton - more DMSP - more aerosols (more clouds above seas: cloud condensation nuclei).

General observations from this assessment

Besides the different conclusions for the different subsystems, some general policy relevant observations are presented here. They mainly focus on the most important interlinkages (both positive and negative) between nitrogen and climate policies, not yet firmly embedded in present policies. These observations are inspired by insights from the underlying studies and are important areas of interest for the future

Observation 1: Large scale production of biofuels

Use of biofuels or bio-energy requires more knowledge before it is applied on a large-scale because of (i) the uncertainty about their net CO_2 effect, (ii) the negative side-effects on the nitrogen cycle and (iii) the competition with the need of land for nature and/or food production. This is especially true in relation to the so-called first generation fuels. Negative side effects on the nitrogen cycle are additional emission of N_2O and NH_3, and increased leaching of nitrogen to aquatic systems due to an increased used of artificial fertilizer. There are also additional NO_x emissions during the production and use of biofuels, like with decentralised combined heat & power installations. There is a clear need for integrated assessments addressing the all impacts related to the use of biofuels.

Observation 2: Enhanced carbon storage in natural systems through nitrogen fertilisation

Stimulating carbon storage in organic matter of natural and semi-natural ecosystems by increased input of nitrogen is potentially a relevant but temporary option to compensate for the emission of GHG. The use of this option as part of climate policy can conflict with nitrogen policy, because the increased generation (or lower ambitions for reduction) of reactive nitrogen by production of fertilizer and fossil fuel combustion leads to effects such as eutrophication, acidification, air pollution, etc. A complete quantification of the effects of nitrogen use on the exchange of CO_2, N_2O and CH_4 by means of a life cycle analysis (LCA) will give more insight in the most optimal use of nitrogen in agriculture with a focus on the net GHG effect.

Policies aimed at enhancing carbon sequestration in forests, (e.g.) by allowing a continuation of the present high nitrogen deposition levels, implies a larger value being put to the societal damage due to climate change compared to eutrophication. At the moment science cannot provide a suitable method for supporting this decision. Societal damage by climate change and eutrophication are both originating from changes in biodiversity, but proper weighting requires that this damage (and other negative effects) are expressed in comparable units.

Observation 3: Need and opportunity for the reduction of nitrogen fertilizer

Given the negative effects of an excess use of fertilizers, in terms of eutrophication, acidification, large scale air pollution, water pollution and climate change (GHG emissions), it is very relevant to optimise the production and use of chemical nitrogen fertilizer. Emission of N_2O during production of chemical fertiliser can be reduced by implementing existing process technology. There are also various options to increase the efficiency of the use of chemical fertilizer. The current opinion in agriculture is that when cheap fertilizer is available, using large quantities of nitrogen is a good option to lower the risks for reduced profits by lower crop yields. However, when also considering societal damage due to climate change and other nitrogen related effects, the economical optimal fertilisation level will be lower. Particularly in the industrialized countries measures aimed at increasing the fertiliser efficiency are effective to mitigate climate change and other nitrogen related effects without additional risks to food production.

Observation 4: Integrated nitrogen policy requires more insight in costs and benefits

It is currently almost impossible to weigh the costs and benefits of different options for nitrogen and climate policy. Whereas some effects can be monetised, the contribution of nitrogen can not always be quantified. This holds e.g. for the additional CO_2 sequestration by nitrogen enrichment, the contribution of nitrogen to global cooling of aerosols, the pollutant swapping between ammonia and N_2O, contribution to biodiversity loss, the chain of biofuels, etc. More research needs to be done to find a common ground for comparison of different measures and to quantify damage effects per unit of nitrogen for the various nitrogen compounds.

1 Introduction

In recent years there has been growing recognition of the link between nitrogen and carbon cycling in relation to greenhouse gas exchange between the biosphere and atmosphere. Integration of these items is an important issue, since climate change policy requires an integrated assessment of the net greenhouse-gas exchange (NGE), involving not only carbon dioxide (CO_2) but also other greenhouse gases like nitrous oxide (N_2O) and methane (CH_4). This integration is vital for future strategy development, since approaches that maximise CO_2 uptake may not optimise NGE. Apart from the obvious links between nitrogen and carbon cycles, there is a requirement to assess overall ecosystem nitrogen budgets, since other nitrogen losses, e.g. ammonia (NH_3) emissions and leaching of nitrate (NO_3^-), are considered as indirect sources of N_2O emissions under the IPCC methodology. Furthermore it should be recognised that the analysis of N and GHG and their interaction has to be performed at linked field-, local-, regional- and national-scales, including consideration of the spatial interactions with NH_3 emissions, aerosol formation and NO_3^- leaching. This is mainly due to the fact that processes involved in the different interactions can be differently quantified at these changing scales.

Nitrogen and carbon cycles also have a clear link at the source side. This is especially true when considering fertilizer production from fossil fuels and its use for food production or the production of biomass for bio-energy, biofuels or bio based chemicals. With a stabilisation of the area for biomass/food production, and the growing need for both food and biomass, there is a need to increase fertilizer use. An increase in the use of fertilizer will lead to increased N_2O emissions and to an increase in the availability of carbon neutral fuels. The other link at the source side is the relation between transport using and carrying fossil fuels. Ship NO_x emissions are increasingly contributing to the nitrogen enrichment of land based ecosystems. International transport of food, fossil fuels and, in the future possibly biomass, will increase and need to be considered as a major contributor to the changes in nitrogen and carbon cycles. The last link at the source side is the obvious coupling between CO_2 and NO_x emissions when combusting fossil fuels, also including CO and CH_4 emissions. In the Clean Fossil Fuel options for CO_2 reduction of other gases is not jet considered, and vice versa.

Currently, scientific research is oriented towards quantification of the relation between nitrogen budgets (resulting from e.g. agricultural, industrial and mobile sources) and greenhouse-gas emissions (CO_2, N_2O and CH_4) at different scales in Europe, both at the source and at the sink side, and the interrelations between these. Insight in this relation, both in terms of processes and fluxes, is important in order to better understand how policy on these items can effectively address these topics simultaneously. Some initial work has been done to include the trade-offs and linkages of different policy options for the different gases in the Netherlands in the so-called new 'Optiedocument". There is a need, however, to list the interrelations between nitrogen and GHG emissions and to assess the effect of current and upcoming policies in the Netherlands and Europe in order to see if there are trade-offs that need to be taken into account in future policies.

The overall objective of this assessment was to investigate the relation between climate policies and nitrogen related policies and (where possible) to determine the effect of (new) abatement of reactive nitrogen on greenhouse-gas emissions and sinks in the Netherlands. Although there are different procedures possible when conducting an assessment, we did this by summarizing and synthesis of existing material on this subject. In general we tried to identify the major drivers and interactions for flows of nitrogen and greenhouse gasses in the Netherlands in order to determine:

- the opportunities for synergism of climate and nitrogen policies
- the risks for antagonism of climate and nitrogen policies.

To visualize on the main interlinkages between nitrogen and greenhouse gasses the following scheme was drawn showing the major interrelations (Figure 1.1).

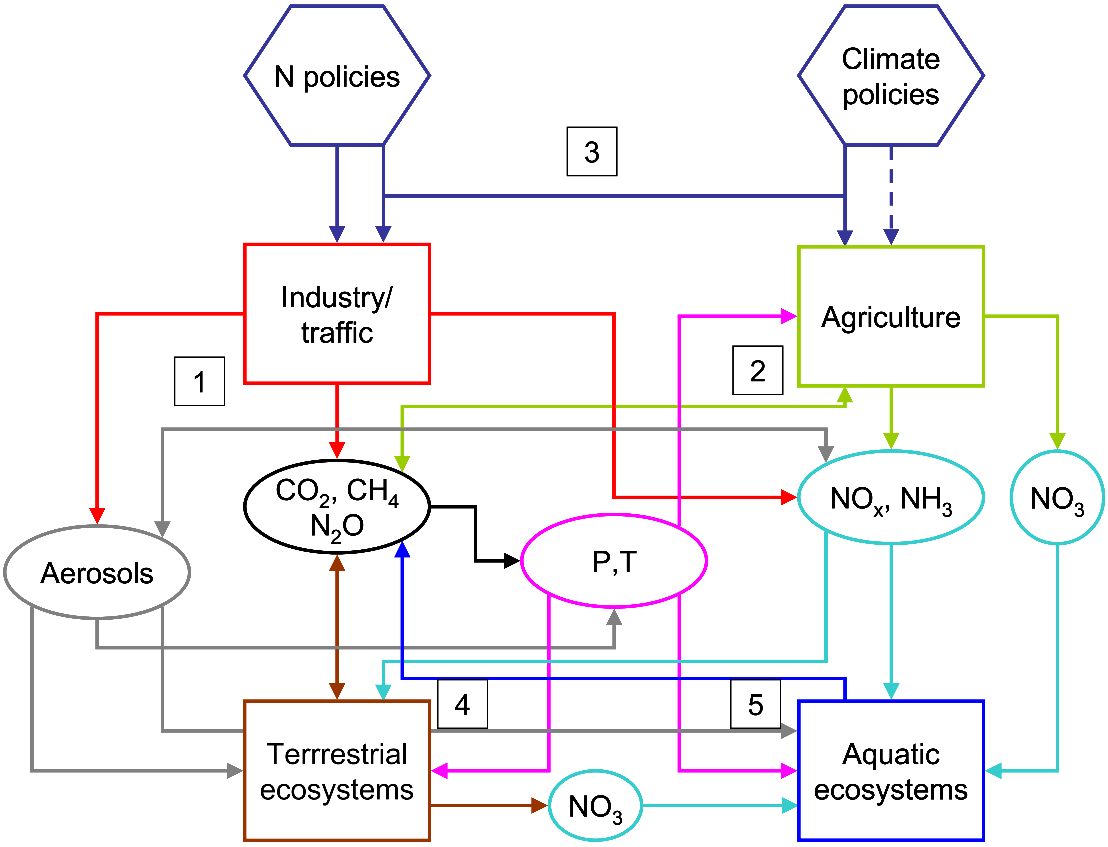


Figure 1.1 Qualitative linkages between the implementation of nitrogen and climate policies in industry/ traffic and in agriculture and greenhouse gas emissions in terrestrial and aquatic ecosystems.

From the overview it is clear that the major reactive nitrogen producers are similar to the GHG emitters: industry (including energy) and traffic and agriculture. When considering the agriculture sector it seems that this sector is contributing more to nitrogen related issues than to GHG, but the interactions related to this sector are complex including both sources and sinks. The industry, traffic and energy sector emit more than 90% of the overall CO_2 equivalents and the relevance for nitrogen is mainly the emission of NO_x. The emitters and receptors (sinks and sources) of nitrogen and GHG include terrestrial and aquatic (eco) systems (including coastal waters). Five comprehensive smaller subsystems were identified for which the interactions between Nitrogen and GHG-emission could me more easily assessed and which are also relevant domains for (future) policies (denoted by numbers in Figure 1.1). These subsystems are:

1. Emission of NH_3, N_2O, CO_2 and fine particles in energy production (Chapter 2)
2. Impact of N-fertilization on GHG emissions in agricultural systems (Chapter 3)
3. Impact of biofuel use, and associated land use change, on GHG emissions in the energy sector and agriculture (Chapter 4)
4. Impact of N deposition and climate policies on greenhouse gas emissions in terrestrial ecosystems (Chapter 5)
5. Impact of N-inputs on GHG emissions in aquatic ecosystems (Chapter 6)

The linkage between the last four sections is taken into account by evaluating the effect of measures related to changing nitrogen use in agriculture on GHG emissions from agriculture, and from terrestrial and aquatic ecosystems. The effects that are taken into account are:

- Release of CO_2 and N_2O related to the production of nitrogen fertilizer (Chapter 3);

- CO_2 sequestration in agricultural soils due to N input (fertilizer and manure) and in terrestrial ecosystems due to N deposition in response to elevated NH_3 emissions (Chapter 5);
- N_2O exchange from agricultural soils (Chapters 3 and 4), non agricultural soils (Chapter 5) and re-emission from aquatic systems in response to changed N input and N deposition (Chapter 6);
- CH_4 exchange from animals, stored manure, from agricultural soils and terrestrial ecosystems in response to changed N input and N deposition.

2 Emission of NH_3, N_2O, CO_2 and fine particles in energy production

2.1 Introduction

The use of fossil fuels for the production of energy (both electricity and heating) is one of the largest sources of CO_2, NO_x and SO_2 emissions. The other important source is traffic; this emission source will not be discussed here (Chapter 4 will provide some data in relation to the use of biofuels). This chapter will deal with the overall energy production and the relation with the emissions of the mentioned components. Furthermore, emission reduction options will be discussed, related to different policy issues. Here the linkages between N and GHG policies will be investigated, where the most important feedbacks between these policies will be listed.

In this chapter energy production will mainly focus on a restricted part of what e.g. the IPCC defines as Energy Industry. The IPCC has three distinct stages within this industry:
- Primary fuel production
- Conversion to secondary/tertiary fuels (refineries)
- Conversion to non-fossil fuel vectors (e.g. electricity/heat)

Here the focus will be on the last stages, the conversion of fossil fuels to electricity and heat. However, due to policy influences and other developments, also the (changing) production of energy by sustainable means (solar, wind, etc.) will be discussed in the following sections. An overview of the drivers and interactions considered in this section is shown in Figure 2.1.

In these following sections, first a general overview of the energy production will be given (2.2) and the effects of existing policies (both N and GHG) on the GHG emissions are quantified (2.3). The quantification of these effects can then act as input in a further discussion about the effects of future policy on N and GHG emissions (2.4).

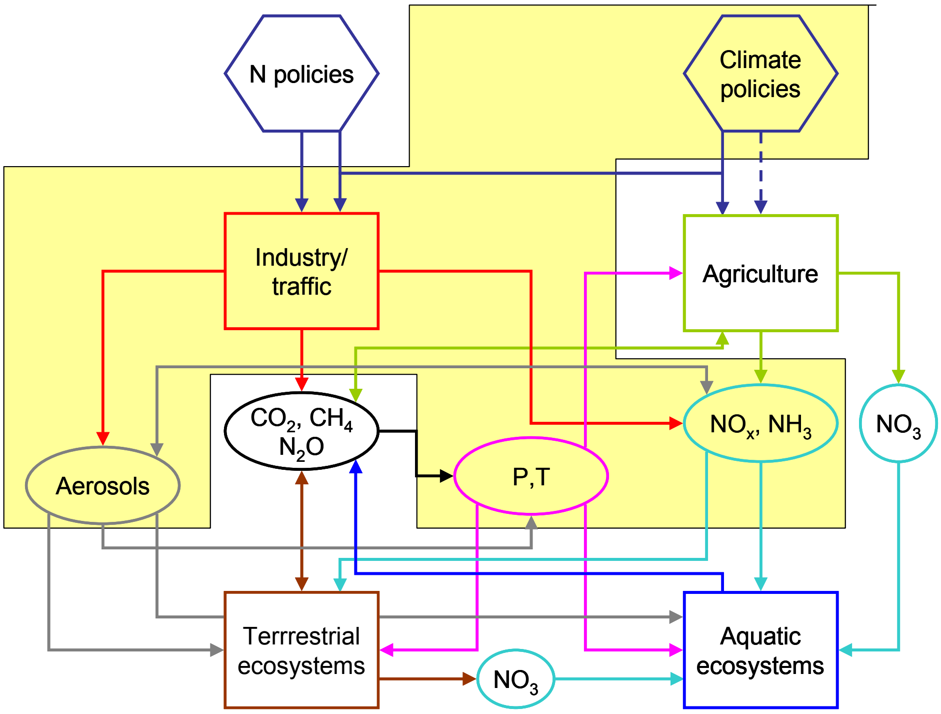


Figure 2.1 Overview of drivers and interactions considered for assessment of the relation between NH3, CO2, N2O and particulate matter emissions from energy production and N and Climate policy.

2.2 Overview of Energy Production

The overall demand for energy is determined by different factors and actors. The main actors are: consumers, industry & agriculture. The way in which this energy (both heat and electricity) will eventually be produced, depends on the availability of energy sources (natural gas, coal, wind, etc), the price of these sources, policy on issues related to the energy production and a combination of these different topics. Choices made with respect to energy production methodology will then result in different emissions of a variety of compounds (e.g. CO_2, SO_2, NO_x, Particulate Matter, Non-CO_2 GHG, Non Methane VOCs, CH_4). Figure 2.2 outlines this overall relation between demand/sources/output schematically.

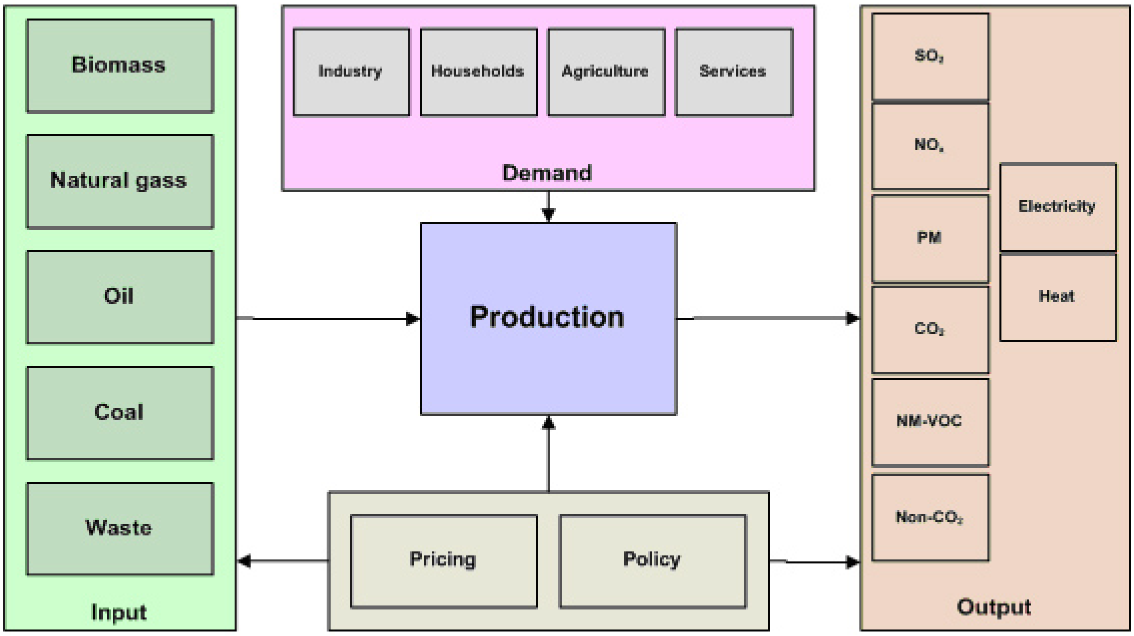


Figure 2.2 Outline of relations in the energy production process.

Factors determining the energy demand are related to different issues, but clear examples are temperature, sunlight availability. It is obvious that climate change will thus have an effect on the overall energy demand in the future. Figure 2.3 shows the development of heat-degree days in the period 1950-2020. A heat-degree day is an indicator of the demand for space heating: the number of degrees the average outdoor temperature is below 18 °C, summed over all days of the year. The calculation of these heat-degree days is based on information about statistical and modelled trends of the temperature over the last decades and is supported by e.g. the Royal Dutch Meteorological Institute (Visser, 2005).

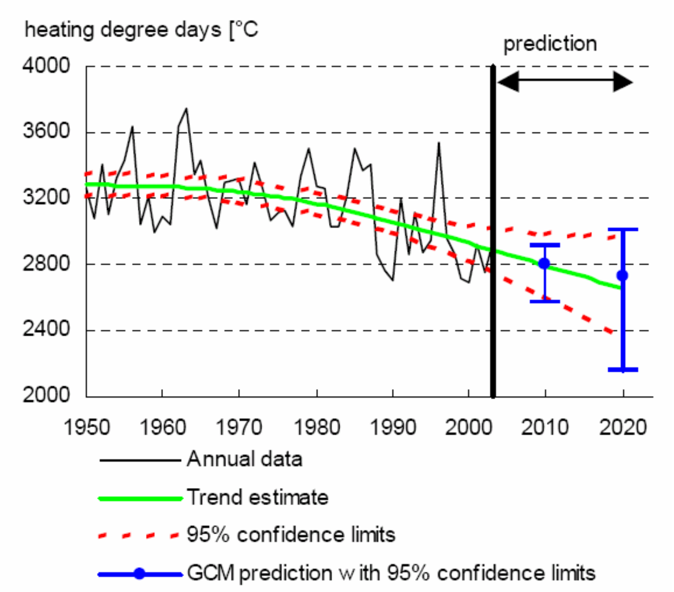


Figure 2.3 Development of the trend in degree days, uncertainty ranges and projections using climate models. Source: (Visser, 2005).

In general, not only the standard winter temperature will be higher in future, but also the summers will be warmer. Overall this results in the following effects:

- less use of space heating installations, especially for households, but also for glass horticulture;
- more investments in space cooling installations (e.g. air conditioners)
- more intensive use of cooling installations for buildings
- less investments in building isolation and efficient heating installations
- more use of air conditioners in cars
- other effects such as product cooling, changing consumer patterns, etc. are marginal compared to those related to space heating and cooling.

The overall effects of changing temperatures on the different emissions will be presented in Section 2.3.

From Figure 2.2 it was clear that another important factor influencing the energy demand and/or production is policy. Most important policy plans related to energy are listed below, categorized by source:

Industry:	CO_2 emission trading
	Benchmarking Agreement
	MJA-2 (Multiple Year Agreement on Energy)
	CO_2 reduction plan
	EIA (Energy Investment Deduction)
	Stimulating Coupled Heat & Power Systems (WKK)
Consumers:	Implementation Plan Climate Policy
Trade & Government:	Implementation Plan Climate Policy
Agriculture:	Implementation Plan Climate Policy
	CO_2 reduction plan
	Stimulating Coupled Heat & Power Systems

In different ways these policy plans have their influence on the emission of (mainly) CO_2, while reduction of other compounds might benefit from these plans as well. This will be discussed in more details in the following sections. Policy on nitrogen issues, affecting the emissions from the energy sector, is limited to measures on the emission of NO_x from e.g. power plants. The interaction of these measures with GHG emission reduction will also be treated in the following sections.

Directly or indirectly these plans are aimed at meeting different targets, most important being the targets for GHG and NO_x. For the GHG these targets are laid down in the Kyoto Protocol and in the recent Government Agreement, where 20% energy saving, 20% sustainable energy and 30% GHG emission reductions are aimed for in 2020 and for NO_x in the NEC (National Emissions Ceiling) Directive. Other compounds that contribute to the overall emission from the energy sector are e.g. NH_3 and Particulate Matter (PM). Emission targets for NH_3 are also laid down in the NEC Directive (128 Gg NH_3). For PM there is no emission target at the moment. However, air quality limit values have been laid down in the EU Air Quality Directive (40 $\mu g.m^{-3}$ annual average PM_{10} concentration).

According to the Kyoto Protocol, the European Union has committed itself to an emission reduction of GHG of 8% on average over the period 2008-2012 compared to 2012. This European target was translated to a Dutch target of 6% in the so-called Burden Sharing Agreement. This results in an overall CO_2 emission target of 199 Tg CO_2-eq for the period 2008-2010. The recent EU energy policy and the Dutch Governmental Agreement is much more ambitious.

The NEC Directive gives emissions targets for different compounds for the individual European countries. For the Netherlands the emission target for NO_x is 260 Gg, to be met in 2010. For NH_3 the target is 128 Gg.

2.3 Quantification of nitrogen and GHG emissions due to energy production

An important input to this section is the Reference Projection 2005-2020, covering the future development of Dutch energy use, greenhouse gas emissions and air pollution up to 2020. The Reference Projection builds on two scenarios, that are based on assumptions regarding economic, structural, technological and policy developments (Van Dril & Elzenga, 2005). The two scenarios are: Strong Europe (SE) and Global Economy (GE). SE is characterized by moderate economic growth and strong public responsibility, while GE assumes high economic growth and has a strong orientation towards private responsibility. These scenarios will be further discussed in the following sections.

2.3.1 The energy production sector

In Section 2.1 the demand for energy was already mentioned as an important driver for the way in which energy will eventually be produced. This demand can be divided over the different economical sectors that will be discussed briefly.

Industry

The industrial sector is the largest user of primary energy, with a total share of about 40%. Besides the demand for energy, this sector also produces energy. More than 20% of the total Dutch energy production occurs in this sector. Looking at the overall energy demand, a distinction can be made between the demand for electricity, heat or for non-energetic purposes (feedstock). Figure 2.4 shows the development of these energy types in the period 1990-2020 for the different scenarios. A clear increase of the demand is prognosed, where the demand for the non-energetic purposes shows the highest increase.

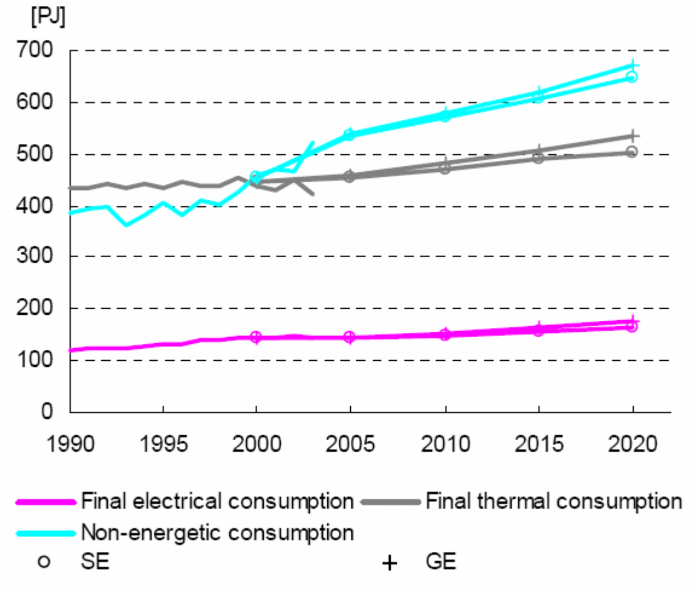


Figure 2.4 Total heat, electricity and non-energetic energy demand (in PJ) for the industrial sector (excluding refineries).

Traffic & Transport

This sector has an overall share of 17% in the primary energy demand in 2000 and is therefore also an important contributor to the energy related problems. The energy demand is determined for 87% by road traffic. It is assumed that the use of fuel for traffic will show an increase from 485 PJ in 2000 to 638 PJ in 2020. The emission of CO_2 is directly related to the fuel consumption, so the same change between 2000 and 2020 can be found here.

Consumers

Consumers are also responsible for 17% of the total primary energy demand, mainly in the form of natural gas and electricity. Figure 2.5 shows the overall electricity demand for the consumer sector. Both in the SE and GE scenarios the demands grows in the period 1990-2020. This is due to an increase in the average use of electricity per household and of an increase in the number of households. Especially some new electricity using devices will cause a (possibly large) increase of the energy demand. Examples are: property security equipment, upgrade of kitchens (electrical cooking) and bathrooms (electrical floor heating). Also the use of home PCs has increased dramatically, causing an increase in the overall electricity demand. The demand for space cooling will increase 10-25 fold (respectively SE and GE) in 2020 compared to 2000.

The direct CO_2 emission due to natural gas use in the consumer sector is about 20 Tg in 2000. This will reduce to about 17 Tg in 2020. The indirect CO_2 emission due to electricity consumption was 9 Tg in 2000. These emissions will however increase to about 13 Tg in 2020.

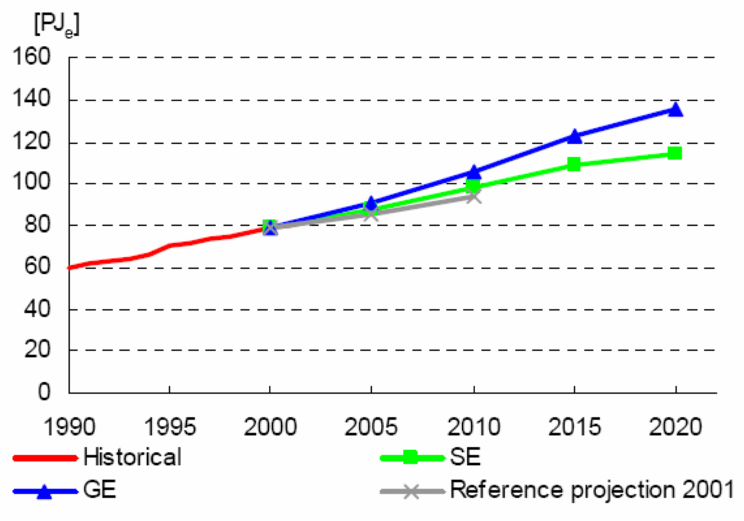


Figure 2.5 Total electricity demand (in PJ) for the consumer sector.

Trade & Government
Like with the consumer sector, this sector also mainly uses energy in the form of space heating and electrical equipment. This thus means mainly using natural gas and electricity and only a small share of oil products. The overall use of energy in the period 1990-2020 will increase from 97 PJ in 2000 to about 150 PJ in 2020. This growth is mainly related to an increase of ICT related office applications like computers, etc.

The direct CO_2 emission will decrease only slightly (10.5 Tg in 2000 to 9.1 Tg in 2020), while the indirect CO_2 emission increases in the same period (11.7 to 17.5 Tg). However, like with the other sectors, this indirect emission is allocated to the electricity production sector and not to these specific sectors.

Agriculture
The agricultural sector has a share of 5% of the total energy demand. Glass horticulture plays a dominant role and is responsible for 4% of the national energy demand. In the coming years the total electricity demand will increase in this sector, while the total heat demand will decrease. Besides this, there is an increase of the own electricity production in the sector over the coming years.

The CO_2 emissions of the agricultural sector will decrease slightly from 7.4 Tg in 2000 to 5.6 Tg in 2020 in the SE scenario, while the emission stays about the same over this period in the GE scenario.

National energy demand
When adding up all the different sectors, a total energy demand for the Netherlands can be calculated. According to Van Dril & Elzenga (2005) this is done without the transport sector demands. In this overall picture for the Netherlands, different issues are brought together that determine the actual energy demand. Examples are: energy savings and climate effects (see also Section 2.2). Table 2.1 gives an overview of the different effect of climate on the energy demands for the individual sectors. For the 2010 GE scenario, the effect of the climate correction is shared over the following sectors:

- Households, heating -1.8 Tg; cooling +0.1 Tg
- Services, heating -1.1 Tg; cooling +0.5 Tg
- Agriculture, heating -0.5 Tg
- Industry, heating -0.6 Tg

These overall effects were taken into account when calculating the total Dutch energy demand.

Table 2.1 Effect of climate (in Tg) on the energy demand for the different scenarios.

[Tg]		**Strong Europe**		**Global Economy**	
	2000	**2010**	**2020**	**2010**	**2020**
Direct effect on the use of space heating installations	-3.0	-4.3	-5.4	-4.3	-5.5
More investments in new installations for space cooling	0.0	0.6	1.6	0.7	1.8
Used in scenarios	-3.0	-3.7	-3.9	-3.6	-3.7
Estimate of less investments in building isolation	0	0 - 0.1	0 - 0.2	0 - 0.1	0 - 0.2
Estimate car airco and other effects	0	-0.2 - 0.2	-0.5 -0.5	-0.2 - 0.2	-0.7 - 0.7

Taking into account the different increases and decreases in energy demand for the two scenarios, the following trends were calculated:

- period 2000-2010
  - SE : 0,9% growth per year
  - GE : 1,2% growth per year
- period 2010-2020
  - SE : 0,7 % growth per year
  - GE : 1,2 % growth per year

2.3.2 Emissions from the energy sector

From the previous sector it is clear that the total energy demand mainly consists of a demand for heat and power. Although both issues will be dealt with in this section, most attention will be paid to the power production. Especially changes in terms of the production methods and the relation with other chapters of this study (i.e. biofuels), make it worthwhile investigating the changes in the power production sector in some more detail.

The electricity production sector consists mainly of central production units, large-scale city heating and industrial facilities with coupled heat and power systems (WKK) and other smaller scale power installations (e.g. small WKK systems, waste incinerators and small scale sustainable systems like wind and waterpower installations). This decentralized power can be rather substantial, mainly because of the coupled heat and power systems.

The annual change of the overall electricity demand is shown in Figure 2.6. It is clear that in both the SE and GE scenarios, the demand will keep growing until 2020 with about 2% annually.

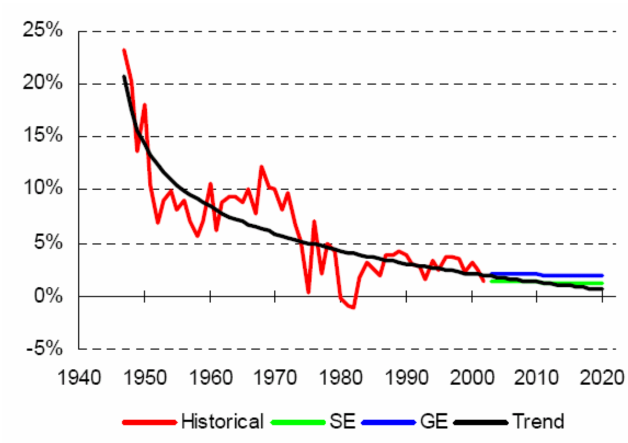


Figure 2.6 Annual change (in %) in electricity demand in the period 1940-2020 according to the two scenarios.

Figure 2.7 shows the different production options for electricity for the two scenarios for the period 2000-2020. For 2000, most of the electricity is produced by coupled heat/power units, coal and gas power plants and import. For the future there are different options: increase of coal plants after 2015 (GE), differences between the use of nuclear power in the different scenarios (e.g. Borssele closing or not), decrease of import and a drastic increase of the use of sustainable inland sources in both scenarios.

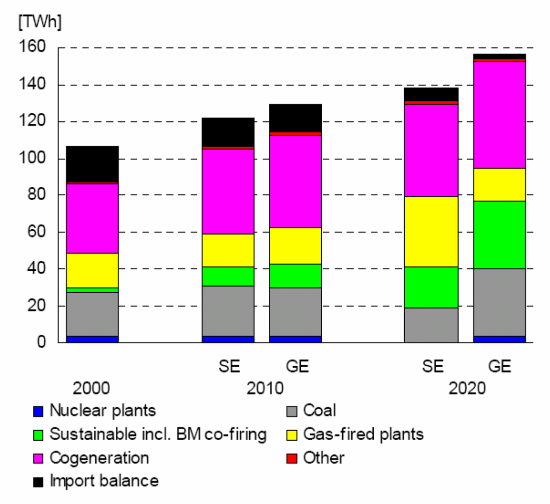


Figure 2.7 Electricity production (in TWh) per production type for the period 2000-2020.

In the year 2000 about 66% of the electricity was produced using natural gas and 25% coal. The share of gas will increase to 68% for 2020, while the share of coal will decrease to 16% for the SE scenario. In this scenario the use of biomass/waste will increase to 11%. The GE scenario assumes something else: 27% coal, 53% gas and 13% biomass/waste.

In 2003 the sustainable energy share was only 1.5% of the total Dutch energy production (CBS, 2004). Figure 2.8 shows the distribution of different production options, where the largest share is from imported energy (62.1%). For the future, again different options are possible for the SE and GE scenarios. Figure 2.9 shows the possible development over the period 2000-2020 with respect to the sustainable energy production. Until 2010 biomass provides the largest contribution, but its share is then taken over by wind power.

The respective targets for 2010 and 2020 (5 and 10%) for the total share of sustainable energy in the total energy production will most likely not be met. For 2010 the estimated share is about 3.5% and for 2020 it is about 7.3%. Most of these sustainable energy options are electricity related (about 95%).

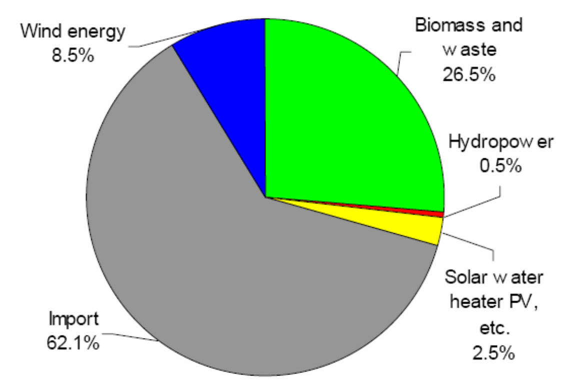


Figure 2.8 Distribution of sustainable energy supply in 2003.

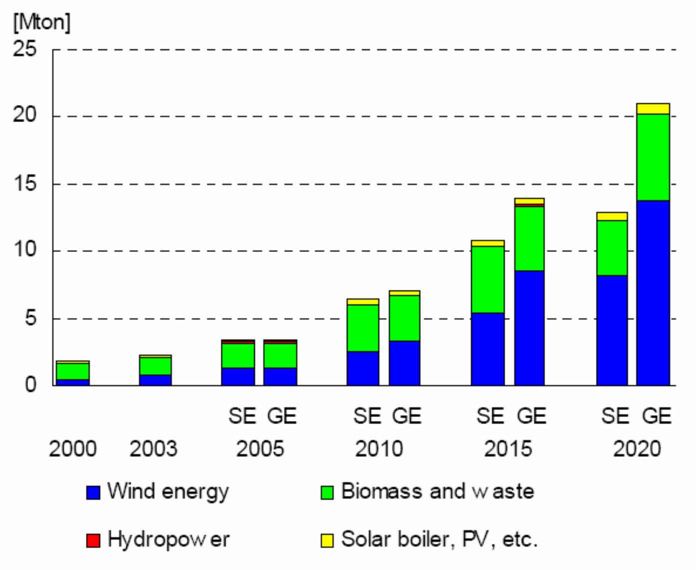


Figure 2.9 *Development of the amount of sustainable energy until 2020 (expressed as Tg avoided CO_2 emission).*

In the period 1990-2020 the CO_2 emissions of the electricity production sector will increase. Figure 2.10 shows the overall CO_2 emission (in Tg) for the different scenarios. Despite some emission reducing developments (wind power off shore, growth of sustainable power) the emission grows to 58.4 Tg for SE and 67.8 Tg for GE.

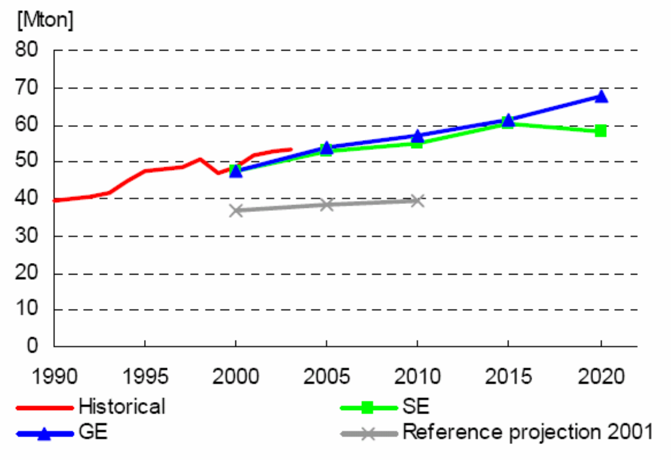


Figure 2.10 *Emission of CO_2 (in Tg) for the electricity production sector in the period 1990-2020 according to the SE and GE scenarios.*

Figure 2.11 shows the total emission of non-CO_2 GHG in the Netherlands (in Tg CO_2-eq) in the period 1990-2020. Overall the following sectors are responsible for the different emissions:

- Agriculture (N_2O, CH_4): 48%
- Waste (CH_4): 20%
- Industry (N_2O, HFK, PFK, SF_6): 26%
- Traffic (N_2O): 1%
- Energy (CH_4): 3%
- Other (N_2O, CH_4): 2%

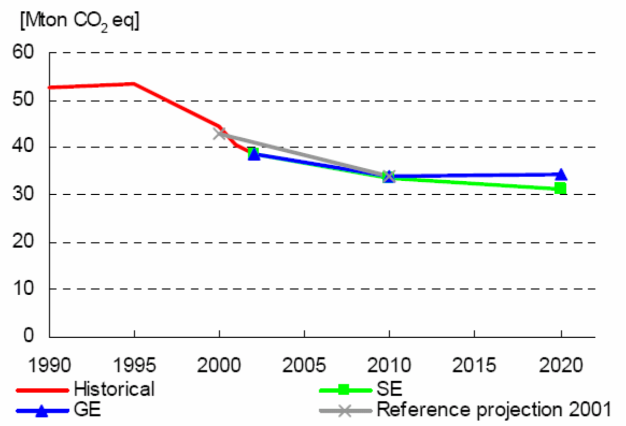


Figure 2.11 Development of the total emission of non-CO_2 *GHG (in Tg) in the Netherlands.*

The main nitrogen compound being emitted during combustion processes is NO_x. Almost 66% of the total emission is caused by the traffic sector, while industry and energy both have a share of about 12%. The remainder is emitted by households, trade, and agriculture. When looking at the other compounds, NH_3 is emitted mainly by agricultural sources. The transport sector is responsible for about 2% of the total emission of NH_3, which is caused by the three-way catalytic converters used in cars. Due to the developments in the car fleet in the coming years, the emission is expected to increase slightly in the period 2000-2020.

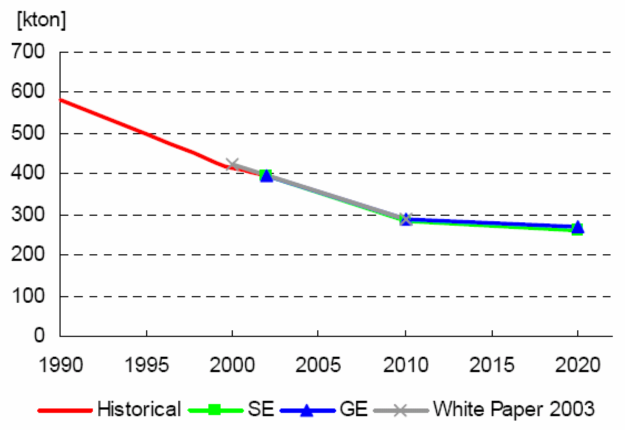


Figure 2.12 Development of NO_x *emissions (in Gg) in the Netherlands.*

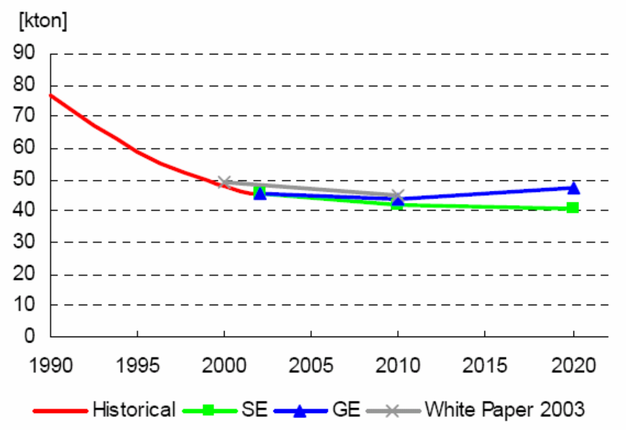


Figure 2.13 Development of PM_{10} *emissions (in Gg) in the Netherlands.*

Figure 2.13 shows the trend in total PM_{10} emissions for the period 1990-2020. No large changes are observed for the period 2010-2020. On average the energy sector contributes about 1% to the overall PM_{10} emission in the Netherlands. This emission is mainly due to the use of coal and of co-firing of biomass. A recent study from De Wilde *et al.* (2006) provided an overview of the PM_{10} emission due to the use of biomass for energy production. Table 2.2 provides an overview of the different emission sources when using biomass. The total PM_{10} emission amounts to 141 tonnes in 2004 and 471 tonnes in 2020. In comparison the emission from private wood burning amounts to 1395 and 1120 tonnes in 2004 and 2020, respectively.

Table 2.2 Overview of PM_{10} emissions for biomass use processes (De Wilde et al., 2006).

Type	Fuel use 2004 [PJ]	Fuel use 2020 [PJ]	Emission factor PM_{10} [g/GJ]	Emissions PM_{10} 2004 [ton]	Emissions PM_{10} 2020 [ton]
Co-firing coal power plant	11.1	79.5	2	22	159
Co-firing gas power plant		8.1	10	28	81
Co-combustion coal power plant			2		
Co-combustion gas power plant	2.8	0.0	10	28	0
Small scale biomass plant	3.3	4.1	5	16	21
Bio-oil in stationary diesel engines		6.5	25		162
Co-combustion in cement production	1.7		?		
Large biomass power plant (stand-alone)		0.0	5	0	0
Municipal Waste Incineration (50% biomass)	27.7	31.0	0.5	14	16
Fermentation of manure and organic waste	0.2	2.0	0.5	0	1
Landfill gas	2.8	1.2	0.5	1	1
(of which flares)	0.8				
Wastewater treatment and sewage purification	3.3	1.6	0.5	2	1
(of which flares)	0.2				
Industrial wood stoves	1.9	1.9	15	29	29
Total	55	136		141	471

According to De Wilde *et al.* (2006), an increase in biomass use will not cause significant changes in the overall PM_{10} emissions. This is mainly because the additional use of biomass in centralized power plants will benefit from existing filtering systems. However, there are indications that the contribution of submicron particles will increase, since these filters have a reduced efficiency for these aerosols (ToMeRed, 2006). Especially these very small particles are important for the cooling effect (De Groot *et al.*, 2003; see also Section 2.3.4). The shift in (the very small) particle size is also shown in Figure 2.14, where the particle size distribution for two cases is shown: coal burning and coal+biomass burning.

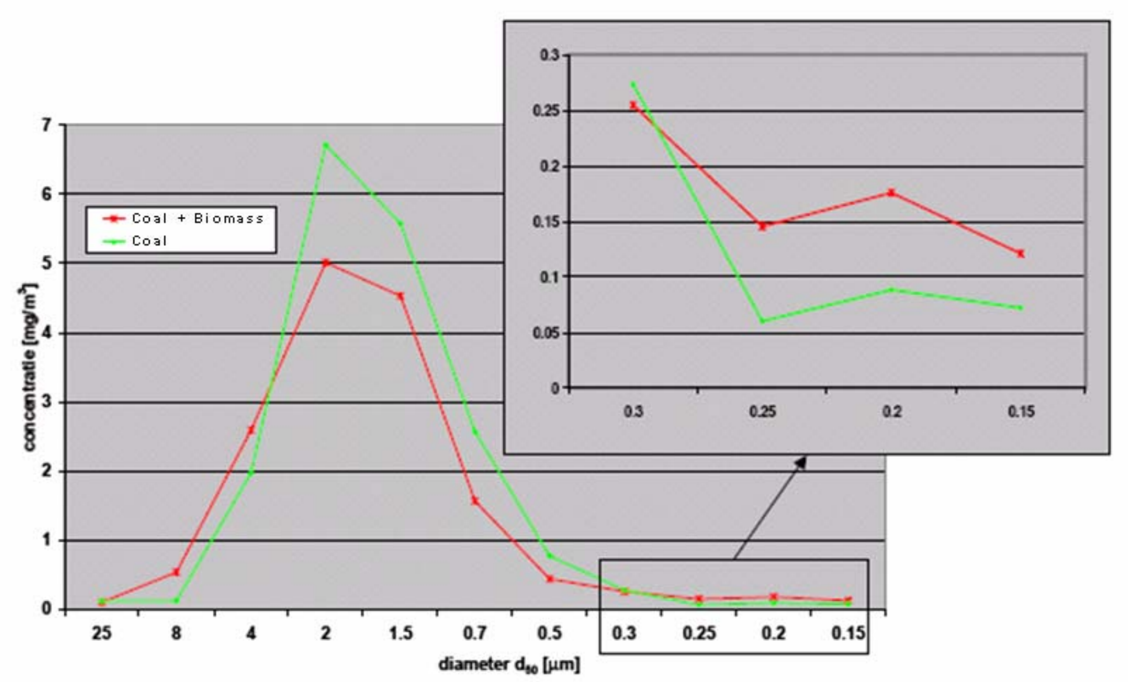


Figure 2.14 Shift in PM particle size for coal and coal+biomass burning.

2.3.3 Interaction between nitrogen and GHG emission reduction options

Most of the measures implemented in the energy sector are at present targeted at reducing GHG emissions, where the main component being addressed is CO_2. In the 'Option Document Energy & Emissions' (Daniels & Farla, 2005) different options have been evaluated with respect to their emission reduction potential.

Table 2.3 lists 23 options from this document with the highest GHG emission reduction potential. The four 'top' options are primarily focused at reducing CO_2 emissions, but also most of the other options focus at CO_2. Only for 3 out of 23 options, other components are targeted. Please note that the options listed in Table 2.3 are targeting the energy production sector in a broad sense; so not only the primary energy production is addressed here.

The different emission reduction from Table 2.3 can be attributed do specific source categories. Table 2.4 shows the overall reductions for these categories. About 89% of the total GHG emission reductions is achieved by CO_2 reductions. The energy sector is responsible for about 35% of the possible emission reductions, while the industry provides another 26% reduction.

Table 2.3 Overview of options with the largest emission reduction potential for GHG.

	Target Compound	Reduction	
		Tg CO_2-eq	%
Wind energy at sea	CO_2	10.0	10
Building new nuclear power plants	CO_2	8.7	9
New concepts large-scale coupled heat&power with CO_2 capture	CO_2	6.9	7
Using biofuels in transport	CO_2	4.6	5
N_2O reduction HNO3 factories	N_2O	4.0	4
Digestion manure and co-substrate dairy farms	CH_4	3.7	4
Heat demand reduction industry	CO_2	2.9	3
Green gas from (co-) digestion of manure (and biomass)	CO_2	2.7	3
Electricity saving by increased efficiency equipment households	CO_2	2.7	3
Gas fueled power plants instead of new coal plants	CO_2	2.7	3
Recycling of plastics	CO_2	2.6	3
CO_2 capture refineries	CO_2	2.3	2
Electricity saving in trade & services sector	CO_2	2.3	2
CO_2 capture at large scale coupled heat&power systems	CO_2	2.3	2
CO_2 capture ammonia production	CO_2	2.2	2
Biomass power plants	CO_2	1.9	2
Different tax & subsidies options	CO_2	1.7	2
Electricity savings equipment trade & services	CO_2	1.7	2
Heat demand reduction Glass horticulture	CO_2	1.6	2
EU agreement on CO_2 emission transport vehicles	CO_2	1.6	2
Reduction of the use of F-gasses	F-Gasses	1.5	2
Kilometer tax for personal cars, cargo vehicles, etc.	CO_2	1.5	2
Other		21.5	22
Total		***96***	***100***

Table 2.4 Total emission reduction for the options from Table 2.3 per sector.

Sector	Reduction (Tg CO_2-eq)						Reduction (% contribution to max. reduction)					
	GHG	CO_2	OGHG	CH_4	N_2O	F-gas	GHG	CO_2	OGHG	CH_4	N_2O	F-gas
Agriculture	10.2	4.4	4.9	3.2	1.7		11	5	5	3	2	
Build areas	14.3	13.9	0.1	0.1			15	14	0	0		
Transport	12.5	12.5					13	13				
Industry	25.4	19.9	5.5		4.0	1.5	26	21	6		4	2
Energy	33.5	34.9					35	36				
Total	***96.0***	***85.6***	***10.5***	***3.3***	***5.7***	***1.5***	***100***	***89***	***11***	***3***	***6***	***2***

Although the different options, listed in Table 2.3 and Table 2.4, are primarily targeted at specific compounds, synergies and antagonisms between different compounds and policy themes can be taken into account when evaluating the overall cost-benefits of the different options. An example is reducing the electricity demand: it not only reduces the CO_2 emission, but also has a positive effect on the reduction of NO_x and SO_2 emissions. While the previous tables only focused on the individual compounds, Table 2.5 gives a rough estimate of the synergy between different options. This table shows for the individual compounds and themes the percentage of the maximum feasible emission reduction, as a result of options that are focused at maximizing the emission reduction of other compounds or themes. Table 2.5 clearly shows that synergies exist between compounds/themes and how strong this synergy is. The percentages shown only represent the interactions when looking at emission reduction targets; if other targets are considered, different synergies will result. As an example, the maximum realization of the GHG reduction potential at the same time reduces 38% of the acidification emissions. When aiming at a maximum reduction of acidification, also 71% of the GHG emissions is reduced. These figures can be explained by the fact that acidification measures only play a small role when trying to reduce GHG emissions, while GHG measures play a larger role when realizing the emission reduction for acidification.

Table 2.5 Overview of additional emission reductions for compounds/themes at maximum reduction for other compounds/themes.

% Additional reduction for other compounds/themes	Targets for maximum reduction								
	CO_2	OGHG	GHG	NO_x	SO_2	NH_3	Acidification	NM-VOC	PM_{10}
CO_2	100	4	100	71	34		78	3	29
OGHG	24	100	98	4	0	13	17		0
GHG	92	15	100	63	31	1	71	3	25
NO_x	29	4	32	100	11		99	-2	34
SO_2	44	5	44	47	100		100	0	47
NH_3	0	39	39			100	100		37
Acidification	21	19	38	46	28	42	100	-1	39
NM-VOC	19		19	21	9		21	100	21
PM_{10}	8	1	9	25	10	44	69	0	100

2.3.4 Aerosols and their cooling effect

Global warming is partly compensated by the cooling effect of aerosols. Aerosols have a direct and indirect effect: direct because of the adsorption of radiation which will not reach the earth surface and indirect because aerosols form the cloud condensation nuclei as a basis for clouds, which also prevent radiation to reach the earth surface. Measures to reduce NO_x and/or NH_3 limit the formation of ammonium sulphate and ammonium nitrate aerosol that is responsible for a cooling of about 2.5 W/m^2 in the Netherlands region (sum of direct and indirect). This cooling effect cannot directly be translated in a temperature change, because the relations are more complex (temporal variations, feedback mechanisms, interactions, etc.). Furthermore, the regional climate impact of aerosols is still poorly understood and it is not known if there is a positive or negative impact. It is advisable to quantify the side-effects of emission reductions and adopt the degree of implementation of policies/measures to the implementation to reduce GHG emissions.

2.4 Effect of possible future policies/measures

In Daniels & Farla (2005b) an assessment on the possibilities for reducing the Dutch GHG emission in 2020 was presented. Different possible measures were combined and ranked based on minimising the national costs of emission reduction. They concluded that the identified measures can be combined to represent a technical emission reduction potential of 90 Tg CO_2-eq emissions in 2020. This implies a technical potential to reduce the national GHG emission from 251 Tg for 2020 to 160 Tg. Several emission targets, ranging from 220 to 180 Tg CO_2-eq have been studied in detail.

The different option 'bundles' that were constructed are shown in Table 2.6. These bundles consist of different possible options that are described in more detail in Daniels & Farla (2006a). For these individual options details are listed with respect to GHG emission reduction, costs for implementation, additional emission reductions (for e.g. NO_x, NH_3, etc). Based on a combination of these detailed information, the most cost-effective option bundles were constructed.

Table 2.6 Identified option categories from Daniels & Farla (2006b).

Categories	Examples
OGHG options	Reduction of emissions of other GHG, like F-gasses, N_2O and CH_4, mainly in industry and agriculture.
NH_3 options	Options in animal farming, focussing on the reduction of NH_3 emission (also reducing GHG emissions in some cases).
Reducing energy demand	Isolation of buildings, efficient cars, efficient equipment and lighting. Activity or function doesn't change, while energy demand is reduced.
Volume and structure effects	Less car use, reduced purchase and use (electrical) equipment, reduced growth of energy-intensive sectors. Activity or function changes, causing a reduced energy demand.
Nuclear power	New power plants
Efficiency energy production	More efficient gas and coal plants, earlier replacement of old plants.
Fuel substitution	Gas plants instead of coal plants.
Coupled heat&power	More and/or advanced coupled heat&power plants, mainly for industry, agriculture and services
CO_2 storage processes	Unground storage of CO_2 produced in industrial processes.
CO_2 storage power generation	Unground storage of CO_2 produced in power plants.
Renewable energy	Solar panels and wind turbines, biomass for electricity production, transport fuels and gas.

Figure 2.15 shows an overview of different options needed for reaching possible targets (note that the actual CO_2-eq target for the Netherlands is 199 Tg), compared to the 2000 reference situation (251 Tg). Depending on the costs of the individual measures, but also the possibility to simultaneous reach other goals (like e.g. NEC targets), different options are needed to reach the actual target emission. The figure shows for instance that the use of renewable energy is only cost effective when the target will be lowered to 180 Tg CO_2-eq. For the present target of 199 Tg CO_2-eq the option that will contribute most to the overall (possible) reduction are reduction of the energy demand, nuclear power and CO_2 storage.

Table 2.6 showed the interaction between different components and themes, when trying to reach the emission levels needed for reaching specific targets in a cost effective way. A more thorough evaluation of specific options and their effects on the respective emissions of nitrogen and GHG is possible, but this requires a further investigation of all possible measures and their consequences on the different emissions. An example for the energy sector is given in Table 2.7, where the different potential emission reductions for possible measures are listed. Not only for the target compound (in most cases CO_2), but also for other relevant compounds (e.g. NO_x, NH_3, PM). Including also possible measures for other economical sectors would enable a more complete overview of the interactions between nitrogen and GHG emission reductions, but also of the subsequent consequences for other compartments (see following chapters).

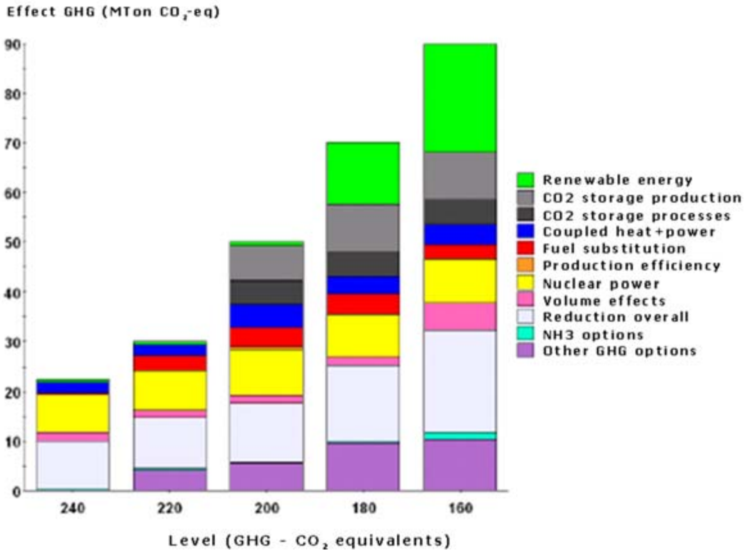


Figure 2.15 *GHG emission reductions per category of the option bundles listed in Table 2.6 , compared to the 2020 GE emission (251 Tg CO_2-eq).*

Table 2.7 *Emission reduction for the Netherlands for different compounds/themes for specific measures available for the energy sector.*

	Target	CO_2	GHG	NO_x	SO_2	Acid.	PM
		Tg	Tg CO_2-eq	Mg	Mg	mld-eq	Mg
Waste incinerators	CO_2	0.2	0.2	-0.1	0.1		
Co-firing gas power plants	CO_2	0.2	0.2	-0.2	-0.2	-0.01	0
Co-firing new coal power plants	CO_2	1	1	-0.1	0.6	0.02	0
Co-firing old coal power plants	CO_2	2.4	2.4	-0.1	1.4	0.04	-0.1
Biomass power plants	CO_2	1.9	1.9	-0.3	0.6	0.01	0
Biomass co-firing in gas power plants	CO_2	0.9	0.9	-0.2	-0.1	-0.01	-0.2
Building of new nuclear power plants	CO_2	4.3		2.6	2	0.12	0.06
CO_2-capture at existing gas power plants	CO_2	2		2	-0.4	-0.01	
CO_2-capture at existing coal power plants	CO_2	7		-1.2	2.5	0.05	0.04
CO_2-capture at new gas power plants	CO_2	13.9		0.8	10.3	0.34	0.32
CO_2-capture at new coal power plants	CO_2	19.6		-0.9	8.8	0.26	0.15
CO_2-capture at 5 oldest coal power plants	CO_2	15.6		-4	4.8	0.06	0.06
Gas power plants in stead of new coal power plants	CO_2	7		2	11.5	0.4	0.35
Green gas from (co-) digestion of manure	CO_2	0.8		-0.2	-0.2	-0.01	
Green gas from waste water treatment plants	CO_2	0.3					
Green gas from gasification of biomass	CO_2	0.1					
Increased number of working hours gas power plants instead of working hours existing coal power plants	CO_2	0.7		0.2	1	0.04	0.03
Increased number of working hours gas power plants instead of working hours new coal power plants	CO_2	0.5		0.1	0.8	0.03	0.02
Coal power plants changing to natural gas	CO_2	4.6		0.1	12.3	0.39	0.38
New coal power plants with higher efficiency	CO_2	0.7		0.4	0.5	0.02	0.13
Increased efficiency by change of operational use	CO_2	0.3		0.2			
Earlier replacement gas power plants with low efficiency	CO_2	0.4		0.3		0.01	
Earlier replacement coal power plants with low efficiency	CO_2	0.8		0.4	0.6	0.03	0.02
Wind power on land	CO_2	0.4		0.3	0.2	0.01	0.01
Wind power at sea	CO_2	7.7		4.6	3.5	0.21	0.11
CO_2-delivery to greenhouses	CO_2	0.2		0.1			
CO_2-storage refineries	CO_2	1		1			
Process integrated coupled heat&power plants for refineries	CO_2	0.4		0.2	0.4	0.02	0.01
Improvement energy book keeping refineries	CO_2	0.2		0.1			
Improvement refinery process	CO_2	0.1		0.1			
Higher targets for participants in NO_x emission trading for electricity production	NO_x			6.7		0.14	

2.5 Conclusions

- When looking at the energy sector as a whole, the way in which energy is produced now and in the future is largely determined by fuel price and availability of fossil fuels. The amount of energy used depends on developments in consumer behaviour, which can also be influenced by changes in climate (higher temperature, less heating - more air conditioning);
- Synergies between GHG and N emission reduction are possible in the energy production sector where NO_x and CO_2 can be reduced simultaneously e.g. by sustainable sources and energy saving. In some cases also reduction of SO_2 and PM is possible at the same time;
- It should be noted however that antagonisms between measures focussing at specific compounds (i.e. GHG and N) are also possible. This is especially true for the use of biofuels and bio-energy. The net result is not always clear and should be investigated in more detail.
- More in general, GHG emission reductions are possible when introducing N (NO_x, NH_3) reduction measures. However, this is not exclusively valid for the energy sector, but holds for different other economic sectors, such as transport and agriculture; Pollutant swapping, however, needs to be investigated, e.g. when ammonia is reduced by manure injection without reducing the N overload, leading to increased N_2O emission.
- When only looking at the energy sector:
    - o measures that are defined mainly focus on CO_2 reduction (with possible simultaneous reduction of NO_x and SO_2);
    - o there are only a few options defined for reduction of NO_x and SO_2 in this sector
    - o quantification of the coupled $CO_2/NO_x/SO_2/PM$ emission reduction is possible, but requires further investigation
- Aerosols can partly compensate global warming, due to their cooling effect. When reducing the emission of NOx and/or NH_3 it will have an effect on the overall cooling effect. However, the regional impact of such emission reductions on the cooling effect is still poorly understood. It is advisable to quantify the side-effects of emission reductions and adopt the degree of implementation of policies/measures to the implementation to reduce GHG emissions.

3 Impact of N-fertilization on GHG emissions in agricultural systems

3.1 Introduction

The use of chemical nitrogenous fertilizer has contributed much to the increase in food production world wide. However large differences exist in nitrogen use efficiencies among crops, soils and regions (Dobermann & Cassman, 2005). Over-dosage of nitrogen fertilizer will increase losses to the environment and can lead to a suite of impacts on ecology and human health (Galloway & Cowling, 2002). The Netherlands have a tradition of productive and intensive agriculture with high live stocking densities. This led to high emissions of ammonia to air and nitrate to ground and surface water (Erisman *et al*., 2005). By reducing agriculture inputs through feed and fertilizer, and application of technical measures, environmental losses have decreased substantially (Grinsven *et al.*, 2005). Although further reduction of agricultural emissions is required to meet targets of EU directives on nitrate and ammonia, stricter limits on application of nitrogen via fertilizer and manure are still a subject of political and scientific debate. Issues are risk of loss of soil fertility, crop yields, food supply, farm income and gross national product. Until now emission and sinking of GHG from agriculture is a relatively minor element of climate policies and agricultural N-related policies. However more attention is needed now, in view of the tight coupling of agricultural nitrogen and carbon cycles (Oenema *et al.*, 2001) with respect to primary production, ranging from field scale to continental scale, and within the agro-complex from production of agricultural inputs (viz. fertilizer) to the consumption of agricultural outputs. It is recognized that climate change itself is an important factor for GHG emissions and future development of agricultural yield, nitrogen demand and nitrogen use efficiencies (Bresser *et al.*, 2005). Figure 3.1 shows an overview of the relevant drivers and interactions for the integrated assessment of nitrogen and climate related policies and impacts.

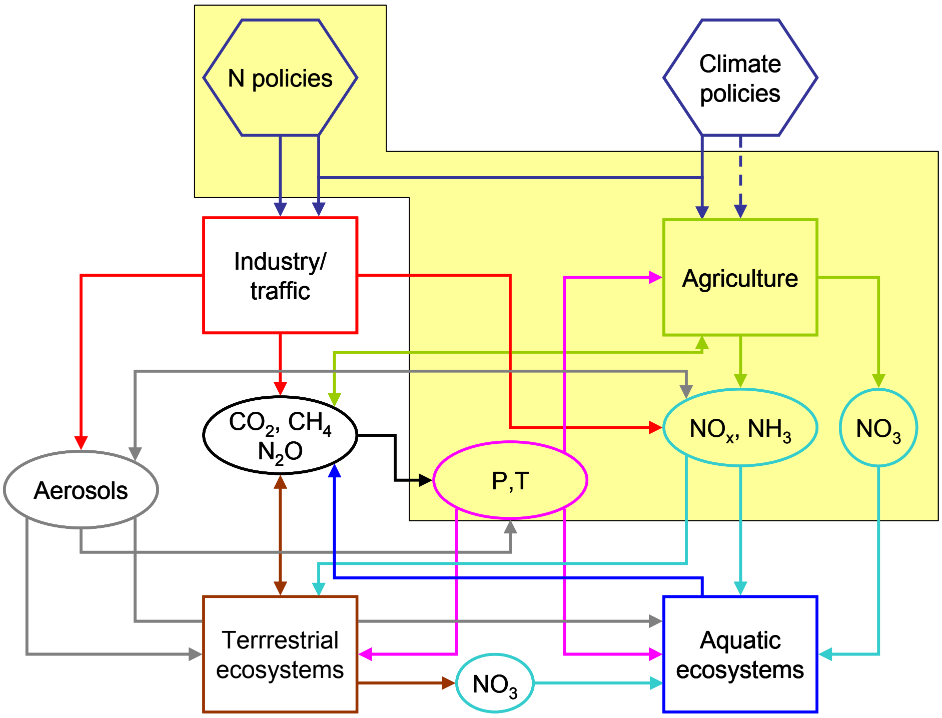


Figure 3.1 Overview of drivers and interactions considered for assessment of Impact of N-fertilization on GHG emissions in agricultural systems.

Outline analysis

For the purpose of identifying the major drivers and interactions for flows of nitrogen and greenhouse gasses in Dutch agriculture, the following topics are addressed:

- Effects of climate change, both temperature and precipitation, on agricultural emission of GHG and N.
- Effects of N-measures on GHG emission focusing on
  - N-measures following from implementation of the EU National Ceilings directive for ammonia and the EU Nitrates directive,
  - Production and use of chemical fertilizer for food, fodder and fibre
  - Application of manure digestion as a measure to implement the Nitrate directive.

Effects of climate measures on N-emissions in agriculture are not considered. Effects of increased cultivation of energy crops on nitrogen flows are in part reported in WAB Biomass Assessment (Lysen & Egmond eds., 2007)

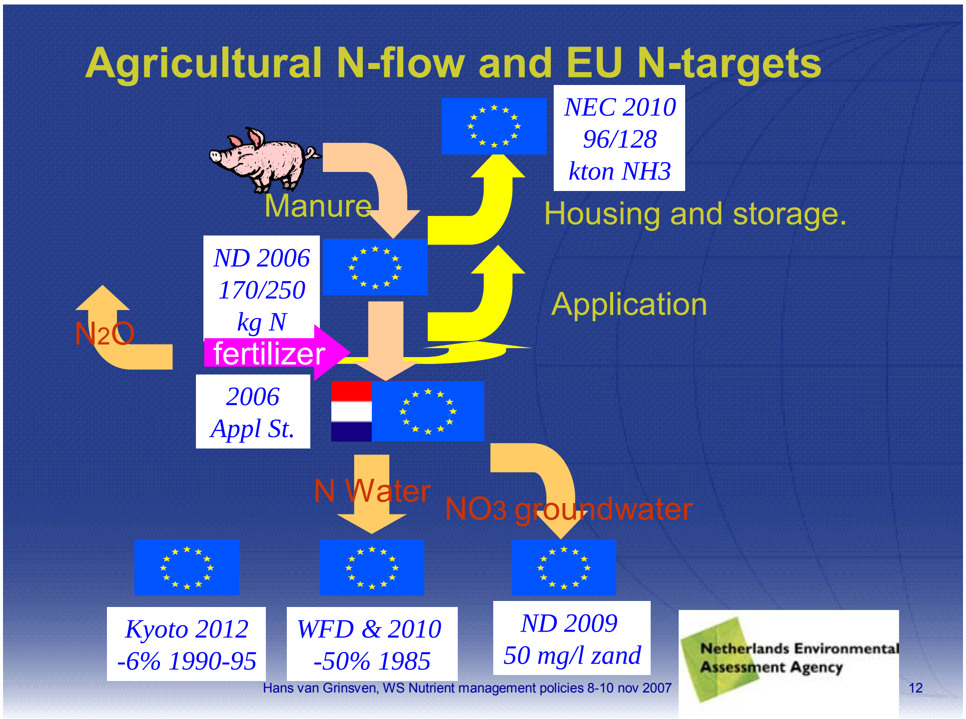
3.2 Nitrogen and climate policies for Dutch agriculture

Nitrates directive

The main objective of the Nitrate Directive is “to decrease water pollution caused or induced by nitrates from agricultural sources and prevent further such pollution”. For this, all member states have to take various measures (i.e.: designate vulnerable zones and establish action and monitoring programs and a code of good agricultural practices for these zones). Nitrate vulnerable zones must be designated on the basis of monitoring results which indicate that the groundwater and surface waters in these zones are or could be affected by nitrate pollution from agriculture. The action program must contain mandatory measures relating to: (i) periods when application of animal manure and fertilizers is prohibited; (ii) capacity of and facilities for storage of animal manure; and (iii) limits to the amounts of animal manure and fertilizers applied to land. In addition to these measures, Codes of Good Agricultural Practices (GAP) are defined for different agricultural systems which are expected to be required to meet the targets of the Nitrate Directive.

The main elements of the Dutch implementation of the Nitrates directive are:

- Legally determined fixed values for animal excretions, as calculated using an animal nutrient balance. The legal excretion values per dairy cow are 114.6 kg of nitrogen and 41.7 kg of phosphate, assuming an average milk production of 7482 litres of milk per cow. Excretion vales are modified for milk production per cow and urea content of milk.
- Application standards for the total use of nitrogen fertilisers based on a balance between the nutrient input (including manure) on the one hand and the nitrogen requirement of the crop on the other hand. Standards are gradually tightened in order to eventually attain nitrate levels in waters not exceeding 50 $mg.l^{-1}$. These standards are differentiated for grassland between cut and grazed. Standards are crop specific. Application standards are applied and enforced at farm scale.
- An application limit for use of nitrogen for manure of 170 $kg.ha^{-1}$ or 250 $kg.ha^{-1}$ for dairy farms with at least 70% of their acreage in use for grassland.
- A set of legal nitrogen efficiencies (in fertilizer equivalent fractions) for various types of manure and application methods.
- A set of technical measures regarding application of fertilizers aimed at reducing losses to water: e.g. bans and limitations of application in autumn and winter, compulsory catch crops after silage maize, regulation of application near water courses, and for ploughing of grassland



Agricultural N-flow and EU N-targets

NEC 2010
96/128
kton NH3

Manure

Housing and storage.

ND 2006
170/250
kg N

Application

N_2O

fertilizer

2006
Appl St.

N Water

NO_3 groundwater

Kyoto 2012
-6% 1990-95

WFD & 2010
-50% 1985

ND 2009
50 mg/l zand

Netherlands Environmental Assessment Agency

Hans van Grinsven, WS Nutrient management policies 8-10 nov 2007

12

Figure 3.2 Overview of EU and national policies and targets for nitrogen.

Water Framework Directive

The Water Framework Directive is the most substantial piece of EU water legislation. It requires all inland and coastal waters to reach good ecological status by 2015. It encompasses a large number of other directives. So far, most important for agriculture is the Nitrate Directive (91/676/EC), which has been agreed upon by all member states in 1991 and which must have been implemented by 2003.

EU Air Quality Directive and the Thematic Strategy on air pollution

The Framework Directive includes daughter directives, which set the numerical limit values for atmospheric pollutants. However, there is no concentration limit for NH_3.

- With respect to agriculture the following policy measures are relevant: The National Emission Ceilings Directive (NEC - Directive 2001/81/EC) sets upper limits for each Member State for the total emissions in 2010 of four pollutants among which NH_3. The emission ceiling for the Netherlands for 2010 is 128 Gg. New emission ceilings for ammonia will be developed before end 2006 as well as new guidelines for the national programs required under the NEC directive (see also below).
- The Integrated Pollution Prevention and Control (IPPC) Directive was issued in 1996. With respect to agriculture it applies to intensive animal production facilities (pig and poultry farms, with > 2000 fattening pigs; >750 sows; or > 40,000 head of poultry) are forced to apply control techniques for preventing NH_3 emissions. Measures that can be applied (e.g. storage, improved housing systems, air purification, manure handling and treatment, manure application) are described in detail in Reference documents (BREF) for Best Available Techniques (BAT), including their emission factor (kg per animal place and year), and an assessment of economic aspects (costs/benefits), animal welfare aspects etc. A possible future extension of the Integrated Prevention and Pollution Control (IPPC) directive is to include installations for intensive cattle rearing and a possible revision of the current thresholds for installations for the intensive rearing of pigs and poultry.

UNFCCC

The United Nations Framework Convention on Climate Change is the main international agreement addressing the issue of climate change. It took effect in 1994. In 1997 the Kyoto Protocol was established. UNFCCC requires parties to use the Revised 1996 IPCC

(Intergovernmental Panel on Climate Change) Guidelines for National Greenhouse Gas Inventories. In this perspective, the EU has adopted Council Decision 93/389/EEC, and later the amended Council Decision 99/296/EC. The UNFCCC sets no specific targets to agriculture. The overall Kyoto targets of 6% reduction of greenhouse gas emission in 2008-2012 as compared tot the reference period 1990-1995 has already been achieved in the agricultural sector mainly by reducing gross nitrogen use in feed and fertilizer. However, the Kyoto target is not sector specific and there are options in agriculture to go beyond the 6% reduction in particular by reducing emission of methane (main source enteric fermentation by ruminants) and nitrous oxide (main source indirect emission from soils and waters. Assessment and implementation of these options is part of the Dutch Climate policy (ROB; Reductieplan Overige Broeikasgassen).

Furthermore the agricultural sector may have the potential to contribute additionally to implement Dutch policy targets to increase the share of biofuels in the transport sector to 5,75% in 2010 and the share of sustainable generation of electro power to 9% in 2010 (Milieubalans 2006).

3.3 Linkage of N and C-cycles.

Nitrogen increases photosynthetic yields

There is a strong linkage between the cycles of nitrogen and carbon in agriculture ecosystems. Nitrogen is an essential element for plant growth, being a component of chlorophyll, amino acids, proteins and enzymes (see e.g. Olson & Kurtz, 1982). Sufficient supply of nitrogen is required for plant metabolism, and addition of N will essentially increase the efficiency of photosynthesis to produce carbohydrates. Photosynthesis only converts a minor part of solar energy (wavelength photons 400nm<λ<700nm) into carbohydrates (approx. 50-200 Gj.ha^{-1}, as compared to total irradiation of 35 Mj.ha^{-1} in the Netherlands).

Yara claims that an addition of 170 kg.ha^{-1} N-fertiliser increases the energy yield of a grain crop from 60 to 120 Gj.ha^{-1}. The energy needed to produce 170 kg of N-fertiliser was estimated at around 8 Gj, resulting in an energy production efficiency of 700-1500% (Yara, 2006). Energy efficiencies will depend on crop type and fertilizer production process.

For wheat and oil seed rape the yield of non-grain (non-tradable) plant parts increase from 4 to 7 tons per ha with increasing N-input from 20 to 110 kg N.ha^{-1}, however the response of the (tradable) grain yield was stronger for wheat than oil seed (Dreccer *et al.*, 2000). Part of the carbon yield of crops will be produced in roots and non tradable parts. If left in the field these carbon yields will increase the soil organic matter content. This potential carbon sink is likely to be small for the Netherlands (see par 3.4.3).

Nitrogen surplus generates losses to environment

Higher inputs of nitrogen to increase yields will decrease the efficiency of the added nitrogen. In the experiments by Dreccer *et al.* (2000) the amount of glucose in grain per kg of added nitrogen decreased from more than 100 kg/kg N for N-inputs of 20 kg.ha^{-1}, to around 50 at inputs of 160 kg.ha^{-1}. The surplus of N will be stored in the soil or will be lost to the environment. For the Netherlands (Figure 3.3) between 1995 and 2000 the average application of fertilizer was 400 Gg N (200 kg.ha^{-1}) and net application of manure was 435 Gg (220 kg.ha^{-1}). About 180 Gg of nitrogen in the gross animal excretion is lost to the air as ammonia and other gaseous N-compounds (25% of total excretion). Of the net nitrogen input of 860 Gg about 230 Gg is leached (25%, which is typical see e.g. Torstenson *et al.*, 2006) and eventually 70 Gg of N is lost to surface waters by leaching (7%). Leaching fractions strongly depend on soil type and soil depth. Leaching fraction in the Netherlands vary from 1% for peat soils to 50% for sandy soils. The leaching fraction at 13 m depth is 3% as compared to the 25% at the phreatic interface.

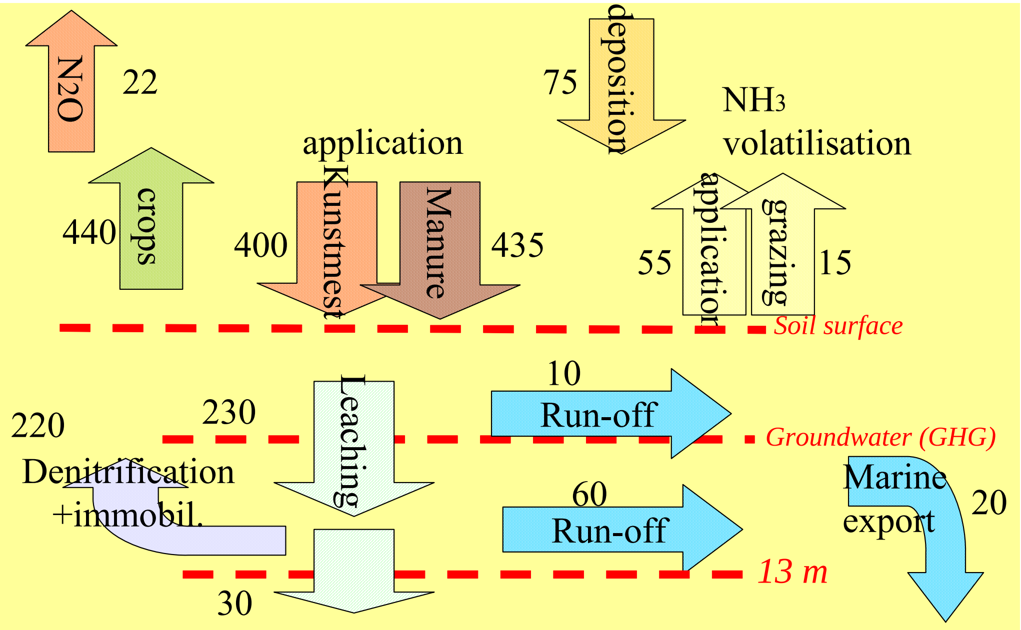


Figure 3.3 The approximate nitrogen balance (Gg) for Agricultural soils in the Netherlands 1995-2000 (based on STONE results van Grinsven et al., 2004, N2O according to MNP, 2006)

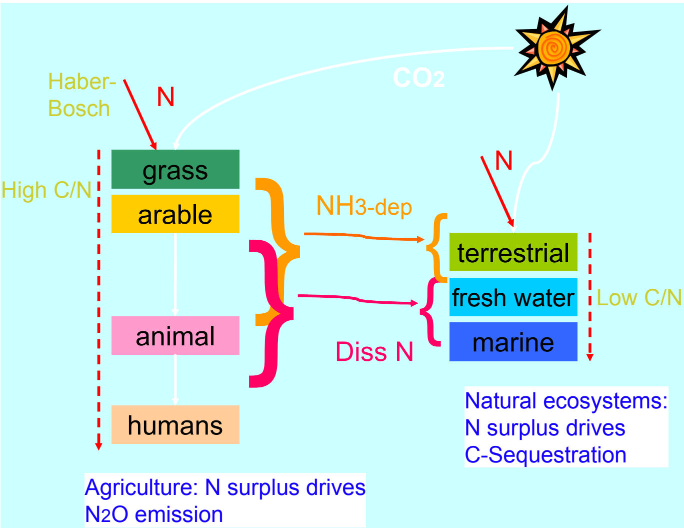


Figure 3.4 Linkage between N-loss from agro-ecosystems and N-loading of natural ecosystems.

These nitrogen losses are key drivers for biodiversity loss and biomass production in natural ecosystems (see Figure 3.4). Carbon sequestration in natural ecosystems due to N-losses from agro-ecosystems could be viewed as climate mitigation, if biodiversity losses are manageable or not considered detrimental. The implications of N-losses from agriculture for greenhouse gas emission in natural ecosystems are analysed in Chapter 5 and 6.

Nitrogen input and nitrous oxide emission

The part of the N-input to agricultural soils that is lost to the environment in the form of nitrous oxides is very small compared to other losses; in 2004 it was about 22 Gg, being 2% of N-input. Figure 3.5 illustrates the main loss routes for nitrous oxides in agriculture being emission from storage of manure, emission from soil application of manure and fertiliser, and indirect emission from groundwater and surface. In 2004 the N_2O emission from agriculture was estimated at 30.4 Gg N_2O, of which 2.3 Gg was from manure management, 15.6 Gg from application of manure and synthetic fertilizer, 10.4 Gg from leaching and run-off and 2.1 from grazing. The major contribution to N_2O emission during application comes from soil incorporation of manure, which is mandatory in the Netherlands to reduce ammonia emission. The currently used emission factor is 2%, but the scientific foundation of this value is weak.

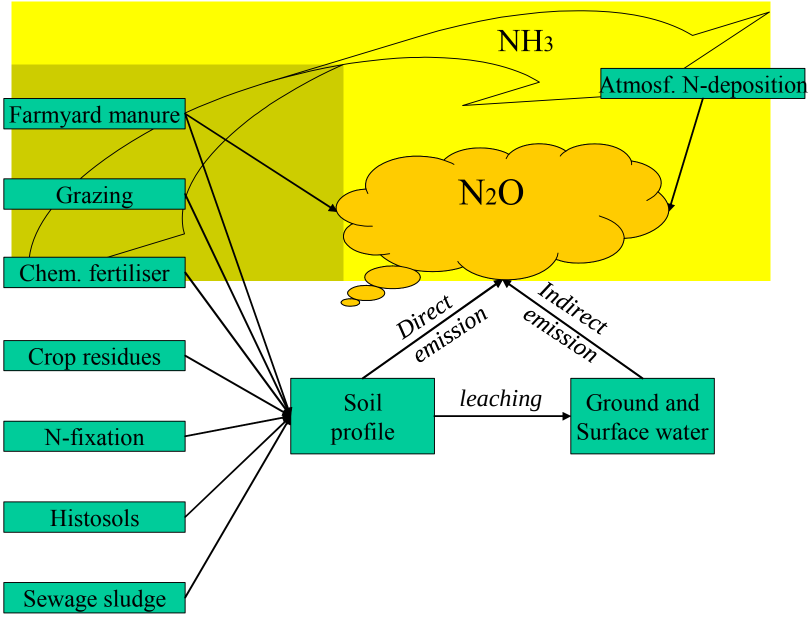


Figure 3.5 Main loss routes for nitrous oxide related to manure and fertilizer use in the agricultural sector following the IPCC-protocol.

The Dutch agricultural sector contributes about 14% to the total greenhouse gas emission of the Netherlands, including energy use by horticulture. Relative contributions in CO_2-eq are comparable for all three gasses (Figure 3.6, Table 3.1). The share of CH_4 and N_2O is decreasing due to decreasing livestock numbers Figure 3.7).

The allocation of CO_2-eq from green house heating in horticulture is arbitrary and not in accordance with FCCC procedures. The GHG emission related to fertilizer production is allocated to the manufacturing industry. The Netherlands is an important manufacturer and exporter of chemical fertilizer.

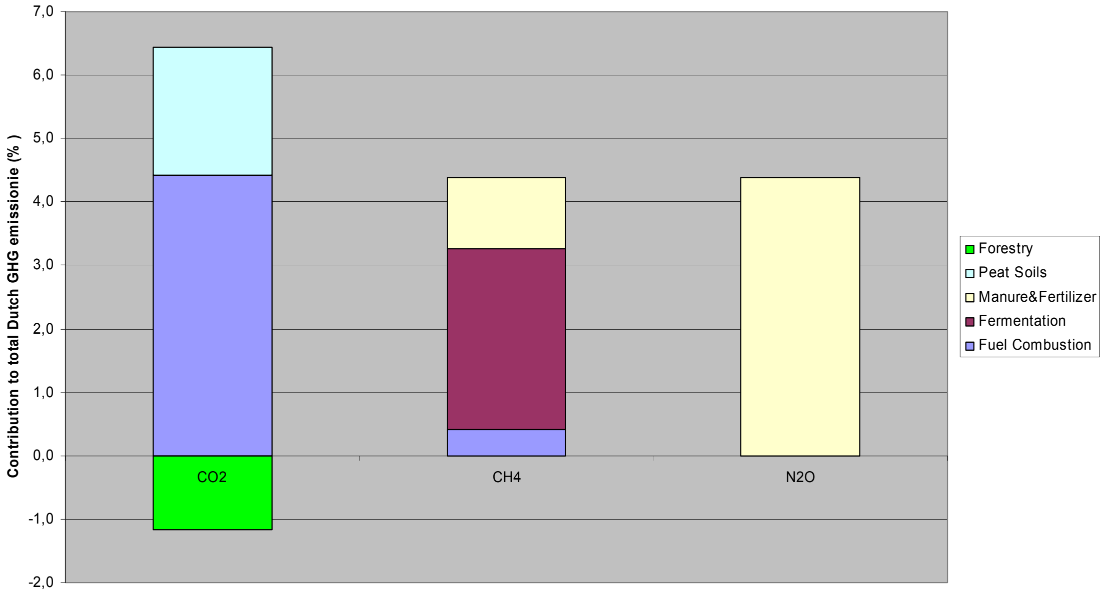


Figure 3.6 *Relative contributions of emissions of CO_2, N_2O and CH_4 from various agricultural activities and processes to the total greenhouse gas emission of the Netherlands in 2003, including the CO_2 emission from heating in horticulture (total contribution 14%).*

Table 3.1 *Summary report for CO_2 equivalent emissions for the Netherlands IPCCC inventory 2004.*

Greenhouse gas source and sink categories	CO_2 (1)	CH_4	N_2O	Total
		CO_2-eq (Gg)		
Total (Net Emissions) (1)	183,030.33	17,293.09	17,738.04	220,151.15
4. Agriculture		8,813.70	9,414.70	18,228.40
A. Enteric Fermentation		6,348.09		6,348.09
B. Manure Management		2,465.61	706.80	3,172.41
C. Rice Cultivation		NO		NO
D. Agricultural Soils(3)		NE,NO	8,707.90	8,707.90
E. Prescribed Burning of Savannas		NO	NO	NO
F. Field Burning of Agricultural Residues		NO	NO	NO
G. Other		NO	NO	NO
5. Land Use, Land-Use Change and Forestry(1)	2,355.65	NA,NE,NO	NA,NE,NO	2,355.65
A. Forest Land	-2,448.05	NO	NE,NO	-2,448.05
B. Cropland	-35.57	NA,NE	NA,NE	-35.57
C. Grassland	4,194.85	NE	NE	4,194.85
D. Wetlands	NE	NE	NE	NE
E. Settlements	-151.54	NE	NE	-151.54
F. Other Land	716.98	NE	NE	716.98
G. Other		NE	NE	78.98

Summary report for CO_2 equivalent emissions for the Netherlands IPCCC inventory 2004

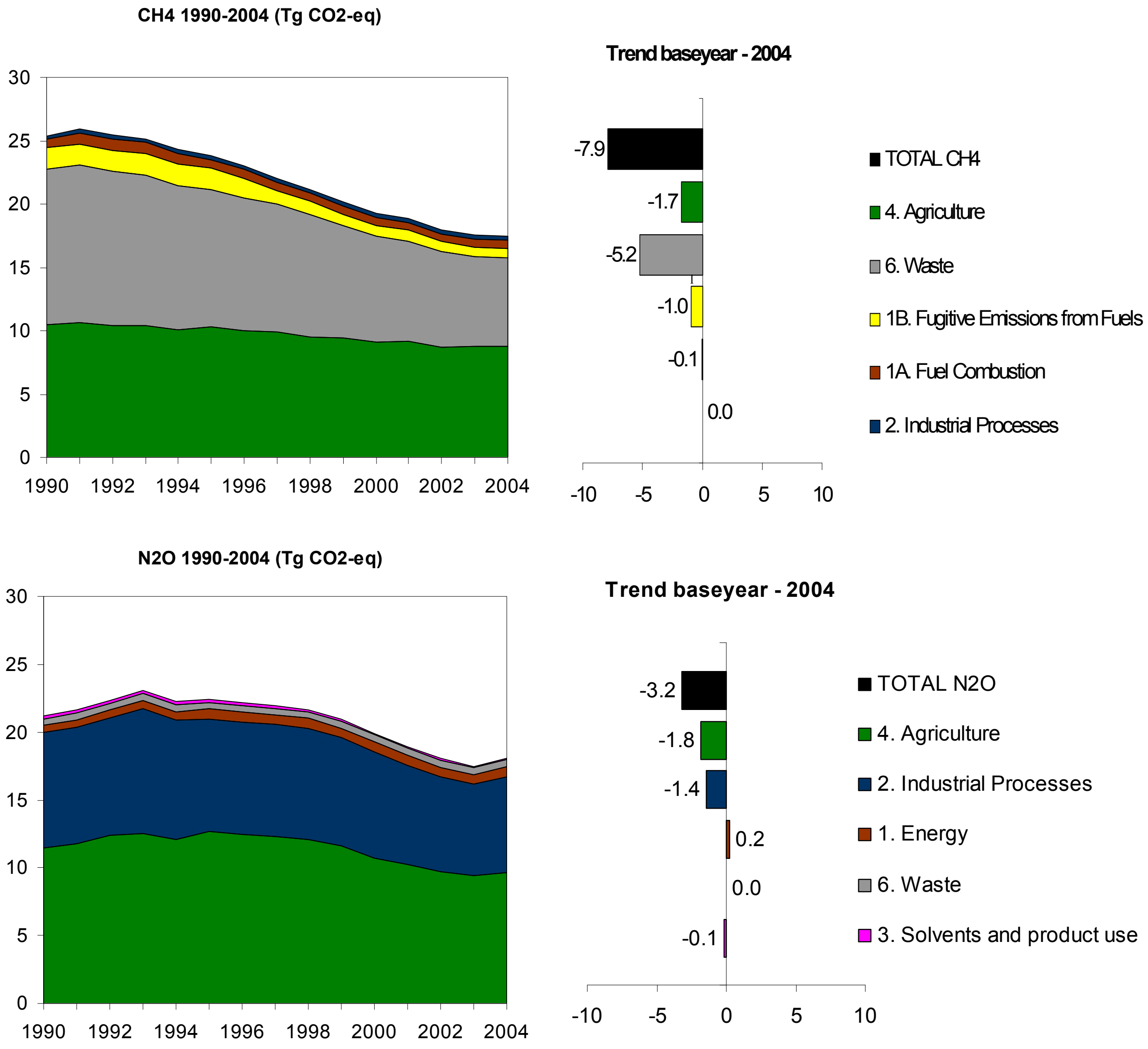


Figure 3.7 Trends and shares of greenhouse gas emissions by the Dutch agricultural (in green) sector.

3.4 Qualitative assessment of linkages between agricultural nitrogen policies, greenhouse gas emission and climate change

Opportunities for synergy

- Reduced inputs of nitrogen to agricultural soils to meet environmental targets for nitrate in groundwater and N in surface water, will decrease N-surpluses and by that all N-losses to the environment, including nitrous oxide. Between 1995 and 2005 fertilizer and manure regulation reduced the total N-input to the soil by 30%
  - o If reduced nitrogen inputs lead to a reduction of crop yields, and crop yields are a limiting factor for farm economy (internal fodder demand or trade) or for national or European supply, then reduction of GHG emission should be expressed per kg of tradable product in order to assess net effect. For Dutch agricultural there is no evidence yet for structural yield losses.
  - Reduced N feeding of cattle. Dutch policy stimulates reduction of N-excretion per animal as this allows higher stocking densities under the ceiling of 170 kg.ha^{-1} of N from manure imposed by the EU-nitrates directive (or 250 kg.ha^{-1} for dairy farms with derogation). However, there is no clear relation between N in feed and methane emission from enteric

fermentation. In general an increase of milk production per cow, will increase the methane emission per cow or ha, but decrease the emission per kg of milk.

Risks for antagonism

- Combat of acidification and eutrophication of ecosystems has had highest priority in Dutch environmental policy in the past decades. Consequently measures to reduce ammonia were imposed in the mid-nineties. Incorporation of manure in soils is mandatory since 1995 and has reduced ammonia emissions during application by about 60%. However, at the same time the N_2O emission has almost doubled from 5.6 Gg in 1990 tot 10.6 Gg in 1995. This trade-off has not yet been evaluated, and by lack of compatible effect indicators is not possible.
- Manure processing is a voluntary option for live stock farmers to meet the application ceiling of 170 $kg.ha^{-1}$ N in manure, or to reduce costs of disposing animal manure. If processed manure is disposed outside Dutch agriculture it is exempted from the ceiling of 170 $kg.ha^{-1}$ N application as manure. Present policies stimulate manure processing in pig and poultry operations, by extending livestock holding rights (quota) for individual farmers in case they process all their 100% manure. The environmental impacts and GHG effect of manure processing are complicated and depend on used techniques and raw material inputs.
  - If manure processing decreases the volume and duration of manure storage, it will reduce the associated methane (about 120 Gg CH_4; 2.5 Tg CO_2-eq in 2004) and nitrous oxide emissions (about 2.3 Gg N_2; 0.7 Tg CO_2-eq in 2004).
  - Energy from combustion and (co-) digestion of manure is considered CO_2-neutral. Building of (co-) digestion installations is fiscally stimulated but energy output is not structurally subsidized. Combustion will lead to a total loss of all N and probably also P, assuming that ashes will not be recycled to agriculture. If this loss of N and P would be compensated by increasing use of chemical N and P-fertiliser, it is questionable if there is net GHG-emission reduction.
  - Combustion and (co-) digestion will lower the organic carbon flux to the soil, in particular in the case of co-digestion of manure, when adding crops as co-substrates. If the soil organic matter content is a critical factor for water and nutrient use efficiency, this carbon cost should be weighted against the carbon gain for climate mitigation.

Table 3.2 N-measures and relative N_2O effect as compared to situation with no measures. Source: Velthof et al. (2007).

	N_2O (%)
NEC-directive related	
Low ammonia emission application of manure	+12
Low emission housing	+8
Low protein feed	-2
Nitrate directive related	
Balanced fertilization	-14
Limitation N-application on sloping grounds	-2
Prohibition of manure application in winter	-2

3.4.1 Climate effect on N-cycle

In this century substantial changes are expected of both temperature (Figure 3.8) and precipitation including changes of seasonal patterns (Table 3.3). The overall impact of these changes in combination with the increased ambient CO_2 levels on the N and C cycles in agriculture, both in crop and soil, are not yet systematically investigated. Overall an increase of agricultural production is expected (Bresser *et al.*, 2005).

Table 3.3 Climate change in 2100 (ranges in between brackets) for the middle climate scenario Netherlands 2100, Source MNP 2005 (after Kors et al., 2000).

Temperature	+2 °C	(1-6)
Average summer rainfall	+2%	(1-4)
Summer evaporation	+8%	(4-16)
Average winter precipitation	+12%	(6-25)
Annual max 10-d precipitation sum	+20%	(10-40)
Repeat time max 10-d precipitation sum	25 yrs	(47-9)
Sea level rise	60	(20-110)

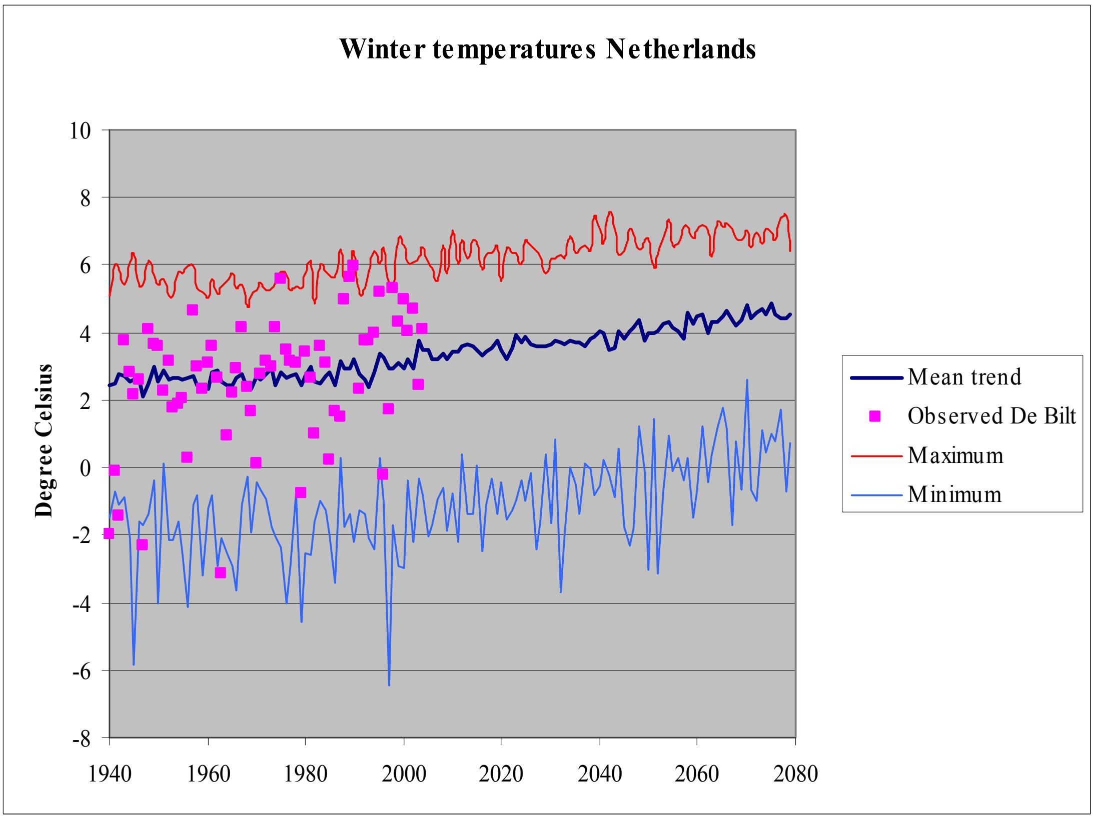


Figure 3.8 Future average winter temperature the Netherlands (Source: http://www.knmi.nl/onderzk/CKO/Challenge_live/).

In view of the multiple feedbacks there is considerable controversy on the net impact of climate change itself on both GHG emissions and losses and efficiencies of nitrogen in Agricultural systems (see e.g. Emmett *et al.*, 2004).

In principal all processes including (micro)-biological processes are accelerated by higher temperatures. A typical effect is a 15% increase of denitrification per 3°C (Linn & Doran, 1984). Most ecosystems studies apply to natural ecosystem; Mellillo *et al.*, (2002) found a net increase of N-mineralization by 60-120% after heating (+5°C) hardwood forest soils between 1990 and 2000. However, no effects were found on net C flux and net gaseous and leaching losses of nitrogen. Apparently internal feedbacks can counterbalance doubling of rates for individual processes.

Heuman & Bötcher (2004) looked at the effect temperature on N-mineralization in arable soils and found exponential increases up to 30°C but found classical Q10-model and Arrhenius model for temperature response to be poor predictors. Jansson & Karlberg (2006) suggest strongest temperature sensitivity just above the temperature where biological activity starts (see Figure 3.9). This effect could be relevant for the Nordic regions and the Netherlands, where there is long period were light availability limits plant growth and low nutrient conservation by crops in combination with predominant precipitation surpluses could increase risks of nitrogen leaching.

There is need to further investigate effects of changes of temperature and precipitation on N and C processes in Dutch agricultural en natural systems, as they may both enhance or reduce GHG emissions and N-losses.

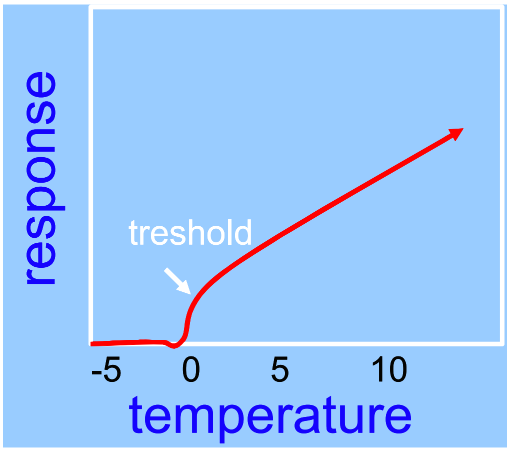


Figure 3.9 Enhanced temperature sensitivity at low temperatures (after Jansson & Karlberg, 2006).

Schlesinger & Andrews (2000) found that potentially huge amounts of CO_2 could be released from soils due to global warming, as temperature is limiting decomposition in many areas of the world. This provided a major explanation for variation of carbon storage on the globe, e.g. when comparing rain forests and tundra's. However, model calculations indicate only modest carbon sequestration effects upon an increase of global temperature of 1 °C and 650 ppm in atmospheric CO_2.

Scholze *et al*. (2006) studied the effect of temperature on the net CO_2 exchange of forest ecosystems, by a meta-analysis on results of 52 Global Climate Models. The net effect is the cumulative results of temperature effects photosynthesis, respiration and decomposition. They concluded that at a temperature rises of less than 2°C or more than 3°C, (forest) soil may become net CO_2 sources, but in the intermediate range the net CO_2 effect is smaller or absent (see Table 3.4).

Table 3.4 The modelled effect of global temperature increase on the net CO_2 exchange of forest ecosystems (Source: Scholze et al., 2006)

	ΔT<2	2<ΔT<3	ΔT>3
2035-2065	19%	0%	13%
2071-2100	13%	10%	44%

3.4.2 N-fertiliser use and CO_2 effect

A generally accepted view is that application of N-fertiliser is very cost-efficient from the perspective of the farmer, as the monetary return is a multiple of the costs. The return can be 500% when comparing a situation with and without fertilizer. (see e.g. Jenkinson, 2001). The question rises if this convincing return also holds when externalities of N-surplus, including GHG emission are included, and at what level of fertilizer input the marginal costs match the marginal profits (see e.g. Pretty *et al.*, 2002 and Blottnitz *et al.*, 2004).

Table 3.5 The gross GHG-emission balance for a arable field with 15 ton dry matter yield and 500 kg input of N-fertilizer (in CO_2-eq/kg N).

Source		**Sink (temporary)**	
Fertiliser production		**Crop**	
CO_2 emission	**1-4 *(Wood and Cowie,2004)***	**C**	**15-50**
N_2O emission	**0-8**	**Soil**	
		C	**3-6 (grassland)**
Fertiliser application			
N_2O emission	**7-9 *(NIR 3% of fertiliser dose*)**		

Table 3.5 gives gross GHG effects of adding 1 kg of N-fertilizer including the emission during production of fertilizer. CO_2-eq emissions from N_2O en CO_2 during production and application of fertilizer depend on the energy source for production, the fertiliser production process, fertilizer type, soil and application method.

Biomass production is not a net permanent carbon sink, but in particular below ground C-sequestration can be vast and could sink, globally perhaps one-third of the annual CO_2-eq emission. As photosynthesis in agriculture is mainly aimed at food production net GHG-emission is an inevitable trade-off for food production (Lal, 2004).

A fundamental debate in agricultural research is identification of the optimal level of N-fertiliser, at what level do the marginal costs of purchase and application of additional fertilizer, match the additional profit from additional crop yield.

Presently cost of N-fertiliser are increasing due to the increased price of energy, and has risen to about 0.8 euro.kg^{-1} (see Figure 3.10). Damage costs of nitrogen fertiliser were estimated at 0.3 euro.kg^{-1} N (Blottnitz *et al.*, 2004). The estimate of these external costs of fertilizer use is incomplete, as most damage items can not be quantified in monetary units. The two dominant items are CO_2-eq emission during production and application of fertilizer. Since 2004, the price of CO_2-emission right has risen to about 30 euro/ton, which would increase Blottnitz estimate to about 0.5 euro.kg^{-1} N-fertiliser.

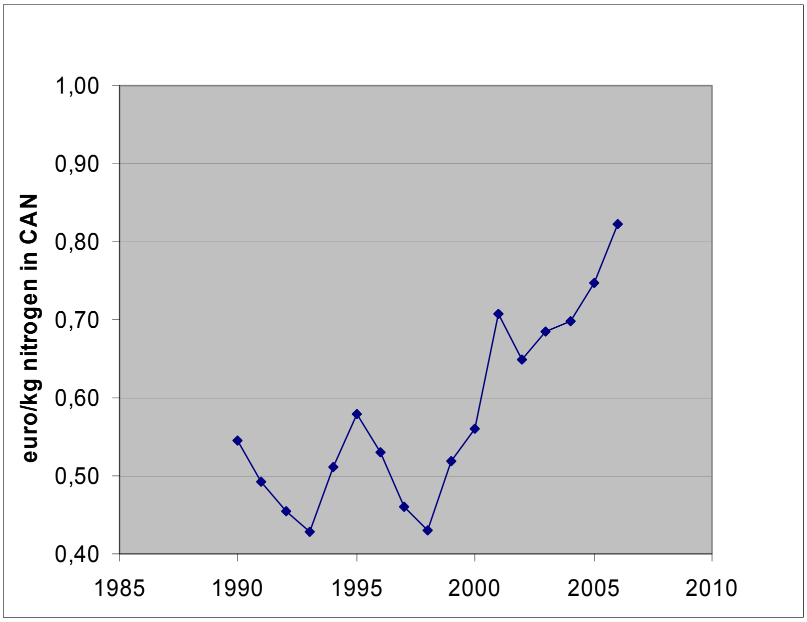


Figure 3.10 Development of prices of single nitrogen fertilizer (CAN).

Another possible bias in Blottniz *et al.* (2004) is the relation between N-surplus and damage costs. Particularly indirect N_2O emissions are a function of the leaching loss, which will typically increase when the N-input exceeds the N-crop output (see Figure 3.11).

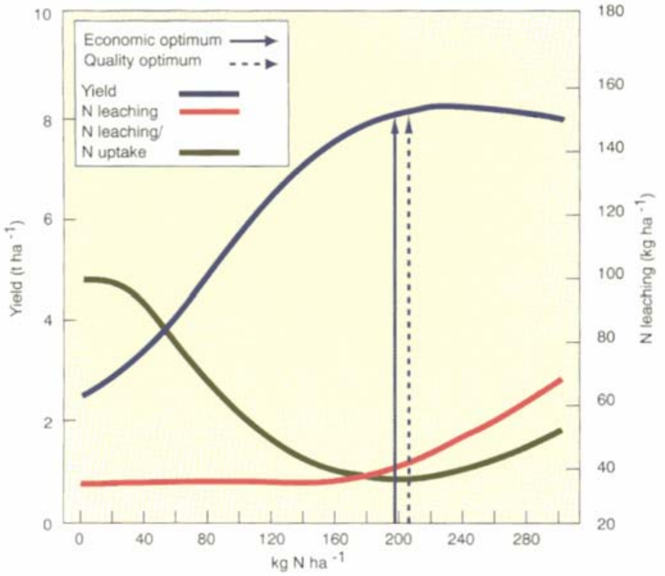


Figure 3.11 Example relationship between wheat yield and nitrate loss at various N-application rates (Source: Laegreid et al, 1999).

If the non-linear relation between N-input on the one hand, and N-damage cost and farm profit at the other hand, would be considered in the identification of the optimal N-fertiliser input, it would probably would be substantially lower than present fertiliser recommendations. However, these lower doses are optimal only from a national perspective, and could require measures to compensate individual farmers for (temporary) yield reductions. This alternative approach for determining optimal fertiliser dosage needs further attention.

A special case of optimal fertilizer input is agriculture for production of energy crops, as in that case not euros are leading but net energy gain. In the example of yield curve of Dam Kofoed

(1983) the net energy gain per kg N is 0 when the marginal energy yield of the crops is equal to the marginal energy need for the fertiliser input. Using the Yara data (section 3.1) this would mean that the optimal N-application is reached when slope of ΔYield/ΔN-input is about 9; that is at an N-application level of about 140 kg.ha^{-1}. This level is far lower than required for attaining the maximum dry matter yield (around 240 kg N.ha^{-1}).

A final point of discussion is the choice and representativeness of the yield curves used for identifying the optimal N-input. These yield curves are often ambiguous and subject to changes due to changes in crop and soil management. If data on the historical development of the relation between dry matter yield and fertilizer input for one crop would be plotted in one diagram highly non-linear relations would result (Figure 3.12).

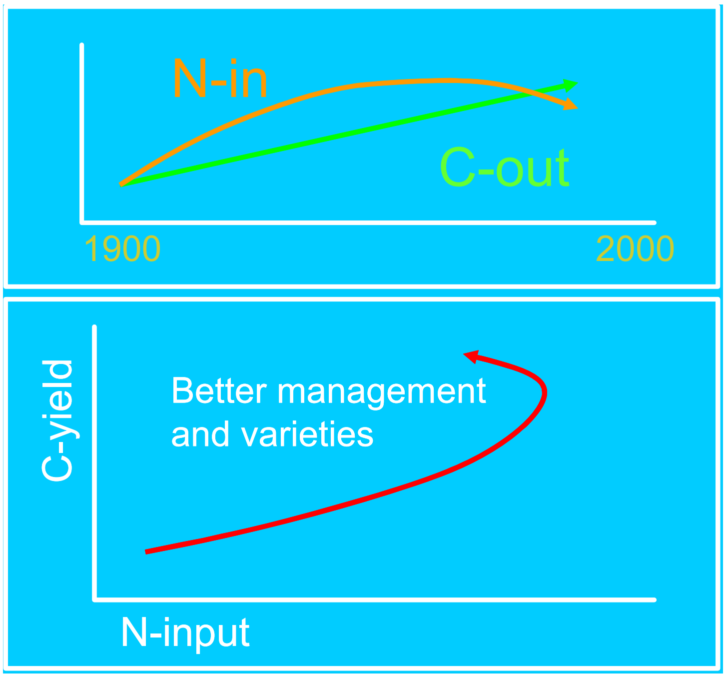


Figure 3.12 Conceptual and hypothetical relation ships between carbon yield, nitrogen input and time.

3.4.3 Greenhouse gas emission related to manure (co-) digestion

Manure (co-) digestion

In the past years some farmers have invested in research and application of manure (co-) digestion. Manure (co-) digestion is a process where bacteria produce biogas from the controlled digestion of manure with or without co-substrate. The biogas consists of methane (45-75%) en CO_2. By combustion of the methane in an installation co-generating electricity and heat (warmte-krachtkoppeling). Another product of the manure (co-) digestion is the digestate that can be utilized as fertilizer. Application of the digestate is easier than of non digested manure. Moreover, nitrogen availability for crops is better. As a consequence less synthetic fertilizer has to be applied. Little knowledge is available on the effects of application of the digestate on gaseous losses (e.g. ammonia and nitrous oxide).

Driving force manure (co-) digestion: climate policy

Investments by farmers in manure (co-) digestion were stimulated primarily by climate policy, to meet targets for sustainable electricity generation and biogas, in order to reduce CO_2 emission form fossil fuel combustion. An important policy instrument was the so-called MEP-subsidy of 9.7 Eurocent.kWh^{-1} that was abolished in September 2006. In addition power companies often pay for local electricity supply to the public network that can amount to ca 5 eurocent.kWh^{-1}. Finally the VAMIL-regulation allows partly tax deduction of investment in digester installations.

Although the maximum tax deduction is 44%, deduction in practice amounts to about half this rate, because deduction depends on the financial position of farming operations.

Dutch climate policy has also instigated reduction of emission of non-CO_2 greenhouse gasses. (e.g. CH_4 and N_2O) by stimulating cost-effective measures in the ROB-programme (Reduction Plan Other Greenhouse Gases). In the so-called ROB-Agro programme Manure (co-) digestion is classified as a favourable option to reduce CH_4-emission from manure storage. In case of frequent (daily for farm scale digesters, to weekly for larger digesters) long-term storage of manure is prevented which will reduce C methane emission up to 95%.

Implementation
At present less than 1% of the total Dutch manure production is digested. Since the MEP-subsidy was ended there is no incentive to further increase manure digestion, as digestion is not profitable anymore for farmers. In Germany and Denmark much larger shares of manure are digested (respectively 23% and 7%) as financial terms are better (Ecofys, 2003). In Germany manure digestion primarily is implemented at farm scale while in Denmark larger community scale digesters are more common.

Relation between manure policy on manure (co-) digestion
The Dutch manure policy aims at reducing nitrate leaching towards ground and surface water. Since the digestate has a better nitrogen availability for crops this leads to a lower leaching of nitrogen. Moreover, it may leads to a lower input of synthetic fertilizer if digestate is accepted by farmers and legislators as an alternative/equivalent of synthetic fertilizer (synergy between climate and nitrogen policy).

The new Dutch manure policy stimulates manure processing. In case a farmer processes all manure and brings it on the market outside the Dutch agriculture, this manure is not regarded as manure. It allows livestock farmers to save on manure disposal costs, and maintain higher stocking densities without violating the 170/250 ceiling for N-application from manure. Moreover, the POR (regulation on derogation production rights manure legislation) gives a farmer 50% extra production (bonus) rights. Within the manure policy manure (co-) digestion is only an option when it falls under the POR and/or when the digestate can be considered as synthetic fertilizer. Then the use of synthetic fertilizer can be reduced.

Digestion of manure: with or without co-substrate?
In many publications it is stated that manure digestion is not economically feasible without the use of co-substrate. The benefits are much higher because of the greatly improved biogas production. However, also costs will be higher. First of all investment costs are higher: a farmer needs a bigger fermentor and also an installation to mix the manure with the co-substrate before entering the fermentor (Ecofys, 2003). Often also a sanitation step has to be carried out if the digestate has to be applied on crop land. Besides, there are costs for the buying or cultivation of co-substrate and to get rid of the mineral surplus. The problem is that many studies do not take into account all costs. Also by using a co-substrate, part of the CO_2 reduction will be counteracted (by the CO_2 production linked up with cultivation and transportation of the co-substrate. Also the availability of co-substrate (or of agricultural land to produce it) could become a limiting factor.

A study carried out by ECN and MNP concludes that manure co-digestion is not always more cost effective than manure digestion without co-substrate (Daniels & Farla, 2006). Cost effectiveness depends on many factors. Especially for pig farms co-digestion can be much less cost-effective if a sanitation step is needed to make it possible to apply the digestate as fertilizer for crop production. For larger cattle farms co-digestion could be more cost effective, because in most cases no sanitation step will be needed (the digestate will be applied to the grassland of the farmer). Another factor influencing cost effectiveness is the possibility to find a market for the heat produced which is not used for own use.

Effect of (co-) digestion on greenhouse gas emissions

Table 3.6 presents the effect of manure (co-) digestion on greenhouse gases (optiedocument potentieelverkenning klimaat, ECN en MNP, 2006). Total GHG emission reduction potential is approximately 5 Tg CO_2-eq which is equivalent to 2% of the total greenhouse gas emission of the Netherlands.

Table 3.6 Effect of manure (co-) digestion on greenhouse gases.

	Reduction CH_4 (Tg CO_2-eq)	**Reduction CO_2 (Tg ref fossile)**	
		Without co-substrate	**With co-substrate**
All farms	2.5	1.6	2.7
Only bigger farms[1]	1.1	0.6	1.1

[1] > 125 cows; > 1250 pigs (for meat production); > 600 sows

Source: optiedocument potentieelverkenning klimaat, ECN 2006

Energy balance of (co-) digestion

Recently, a study was published on the negative energy balance of digestion of manure without co-substrate (Zwart *et al*., 2007). This was not in line with most other publications on the energy balance of manure (co-) digestion.

A major difference between the study of Zwart *et al*. and other studies is the assumption on the heat consumption by the digester and the heat production by utilizing the biogas. For small (farmer) scale digesters Zwart *et al*. considered the heat consumption to be twice as high as for large scale digesters. In case the digesters thermal efficiency would be 55% this would lead to a net zero heat production. However, another assumption of the study by Zwart *et al*. was a thermal efficiency of 35%, leading to a lower heat production. Overall a negative thermal energy balance was reached.

If we assume that a thermal efficiency of 55% could be reached by taking the appropriate technical measures, then the higher heat consumption could be counteracted and lead to a net zero heat production. As a consequence this would lead to a net zero CO_2-reduction as far as the thermal energy is considered. Overall this would still lead to a net positive CO_2-reduction potential, because of the effect of the electrical energy produced. Furthermore, a future development towards use of fuel cells for energy production from biogas could lead to a much higher electrical energy efficiency and as a consequence to a higher net positive CO_2-reduction potential.

Some other assumptions made in the study by Zwart *et al*. also contribute to a negative energy effect. One assumption is that the energy use for the transport of manure and the application of the residue is taken into account. However, if we make a comparison between the energy balance of manure with digestion and manure without digestion, the transport and application of the manure should not be taken into account, because its energy effect is approximately the same in both cases. Another assumption was that the biogas yield was lower for pig manure in Zwart *et al*. than in ECN and MNP, 2006.

Summarizing, it is clear that many factors influence the energy balance of manure (co-) digestion. Especially on farm scale level, the net heat production could be close to zero. However, in the future the effect on the total GHG emission reduction potential of manure (co-) digestion could disappear if fuel cells would be used instead of co-generation of electricity and heat.

Conclusions related to manure (co-) digestion

- The total GHG emission reduction potential of manure (co-) digestion is 4-5 Tg CO_2-eq ((2% of total NL-GHG emission)
- Total methane emission reduction potential is 2.5 Tg CH_4 ((28% agriculture, 14% national CH_4 emission)
- Without MEP subsidy (co-) digestion is only cost-effective for very big farms

- There are large uncertainties about the availability and side-effects in case of large scale use of co-substrate. If large scale implementation of (co-) digestion of manure leads to an increase of use of synthetic fertiliser or imports of feed and organic product it is likely that digestion is not sustainable.
- The acceptance of digestate by arable farmers as an alternative for synthetic fertilizer and by the legislator as an equivalent of synthetic fertiliser is still uncertain. In particular in the case of large scale co-digestion there is the risk of untradeable stocks of digestate.

3.4.4 Relation between N_2O-emission and soil and crop management

The production of nitrous oxide is a complex process depending both on the nature of the substrate and a suite of environmental factors. Nitrous oxide is a by-product of two common soil processes, nitrification and denitrification. The most common substrates are plant residues and animal manures, coming from various sources and are applied to the soil in various ways. Nitrification and nitrification involving these substrates always require looking at the interaction of carbon and nitrogen at the level of chemical reaction and at the level of the GWP-effect.
Three general principles are

- N- addition stimulates C-storage
- N- addition stimulates N_2O production
- C- addition stimulates denitrification

The net effect of carbon addition via plant residues or manure on N_2O production is therefore ambiguous as the effects on denitrification rate and N_2O/N_2 ratio are opposite (Table 3.7). Also the effect of temperature is not always clear.

Table 3.7 Environmental factors controlling indirect N_2O production during denitrification.

	Denitrification	**N_2O/N_2 ratio**
Increasing nitrate content	+	+
Increasing oxygen content	-	+
Increasing available organic carbon	+	-
Increasing temperature	+	-
Decreasing pH	-	+

The ambiguous effect of carbon addition is further illustrated by the effect of the C/N ratio of pig slurry on the N_2O emission flux. Although lower C/N ratios enhance the denitrification rate and indirectly the N_2O emission rate, this effect is overwhelmed by the effect of the higher C/N ratio on the N_2O/N_2 ratio (Velthof *et al*, 2005).

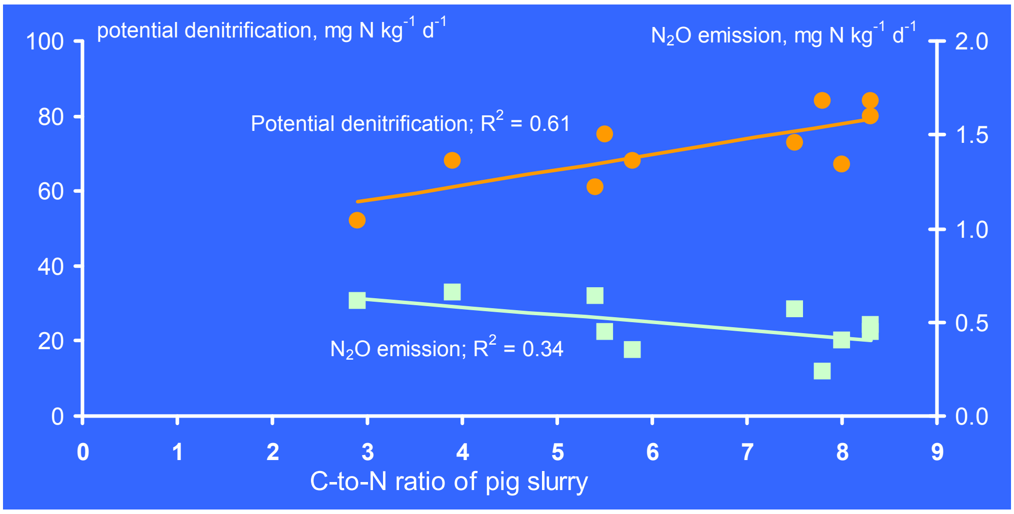


Figure 3.13 Effect of C/N ratio in pig slurry on the rates on denitrification and nitrous oxide emission (Source: Velthof et al., 2005).

A consequence of the generally present impact of environmental factors is that:

- N_2O emission per kg of added N in arable land is lower than on grassland
- N_2O emission per kg of added N in sandy soils is less than for clay and peat soils

The N_2O emission per kg of synthetic N-fertiliser is lower than per kg N from manure on arable land in view of lower availability of easily decomposable carbon (Figure 3.14), but higher on wet grassland (Velthof, 1997).

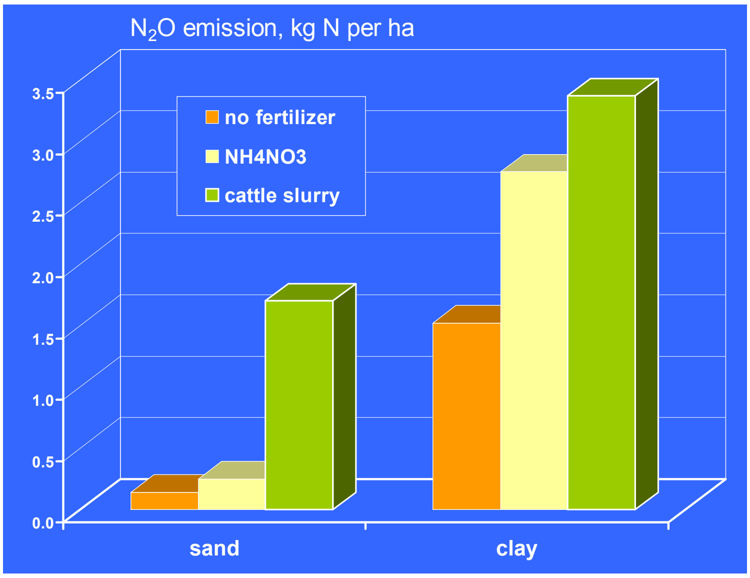


Figure 3.14 Effect of soil type and N-source on the emission of nitrous oxide (Source: Van Groeningen et al., 2004).

The complex relations between nitrous oxide emission rates and substrate characteristics and environmental factors makes it very difficult to establish generic emission factors for calculating the national GHG emission or evaluating policies and measures, as required for UNFCCC.

3.5 Conclusions

- Agriculture has substantial but partly evitable net GHG effects both through the arable sector (fertilizer), and the animal sector (enteric fermentation and manure storage).
- NEC ammonia measures enhance N_2O emission through mandatory manure incorporation, but we lack the knowledge to weigh the ammonia-ecology benefits against the GHG-effect of additional nitrous oxide emission.
- The nitrate directive has lead to 30% reduction of total N inputs to Dutch agricultural soils and has direct co-benefits for climate via reduction of N_2O emission. Further reduction to increase nitrogen efficiencies is possible.
- Including GHG-effects in damage effects and using the approach of marginal financial gain including externalities like damage by eutrophication and climate change should lead to substantially lower N-fertilizer recommendation for both food and energy crops.
- (Co-) digestion of manure can be a worthwhile measure to reduce methane and nitrous oxide emissions from manure storage, and to increase the fertilizing value of manure as compared to chemical fertiliser. Focusing too much on biogas and energy generation may lead to unsustainability if this would reduce replacement of chemical fertilizer by manure and reduce the organic matter supply to the soil.
- N_2O emission from agricultural soils is very complex which is major reason for frequent revision of emission factors.
- Effect of climate change (temperature and precipitation) on N and C-cycle in relation to GHG emission and N-losses for agro-ecosystems can be substantial but are hardly quantified for the Dutch situation and therefore need further attention.

4 Impact of biofuel use and land use change on GHG emissions in the energy sector and agriculture

4.1 Introduction

Energy use is one of the main drivers of developments in our society. The availability and use of energy strongly influences transportation, food and water, industrial development, economic growth, etc. Biomass is the oldest resource of energy used by mankind and has been the main source of energy until no less than a century ago. Biomass is storage of (solar) energy and can be committed as and when required. In principle biomass can replace the current fossil fuels without changing the infrastructure. Biomass can be used to produce synthetic natural gas (SNG), transportation fuels as diesel, ethanol and even gasoline type of fuels and fuels that can be used in coal power plants. This is a main advantage over the other sustainable options. As with fossil fuels, however, the need to produce sustainable energy requires significant societal and technological developments.

Because of the inherently low efficiency of the photosynthetic process, no form of energy supply has such low power densities, and hence such high land demands, as does the production of phytomass. Recent estimates of the global terrestrial net primary productivity (NPP) average approximately 55 billion metric tons (tonnes) (Gt) of carbon per year, that is, approximately 120 Gt of dry biomass that contains (assuming the mean energy density of 15 billion joules per metric ton [$GJ.ton^{-1}$]) some 1800 10^{18} joules (EJ). This productivity prorates to less than 0.5 watts per square meter ($W.m^{-2}$) of ice-free land. Rates for forests are naturally higher, but they do not surpass 1.1 $W.m^{-2}$ even in the richest tropical ecosystems (Smil, 2004). In all natural ecosystems, a large part of the NPP is consumed by heterotrophs (ranging in size from bacteria to megaherbivores); hence, the phytomass that is actually available for energy conversion is only a fraction of the originally produced biomass. In principal there is globally enough annual growth of new biomass to cover up to 4 times the human annual energy use. However, in order to grow, collect and use biomass in a sustainable way to still the human energy hunger, a well regulated and optimized process is needed.

Recently the EU adopted new targets for sustainable energy and GHG emission reductions: 20% reductions and 20% contribution of sustainable sources in 2020. The Dutch government has agreed to reach these targets and set an ambition for the national GHG emission reduction of 30% in 2020. The way to reach these targets is being studied now and proposals for policies and measures are expected in the course of 2007. From the drafts it is clear that biomass as transport fuel, electricity and heat production and Substitute natural gas will be a major component necessary to reach the targets. It is estimated that a sustainable energy production of 750 PJ will be necessary in 2020, of which 500 PJ must be from biomass. The current share of biomass in the Dutch energy use is only 60 PJ (total Dutch use is 3250 PJ, www.energie.nl). Estimates of the current available biomass (mainly waste and agricultural rest products) range from 130-180 PJ (Rabou *et al*., 2006). There is a major challenge to reach the targets in a sustainable way.

There has been a large increase in the volume of biofuels for transport produced mainly with sugar cane in Brazil and corn in the USA. The motivation for producing biofuels is the lower production price, especially ethanol from sugar cane, and the fact of being a recycled fuel not contributing to increase the CO_2 concentration in the atmosphere. In this way biofuels can be judged as an appealing environment-friendly fuel. However, the environmental and social costs of sugar cane, corn, or any other crop has not been fully discussed and taking into account in the whole budget of the biofuel production. Furthermore, the greenhouse gas balance focuses mainly on the replacement of current fossil fuel use without taking all GHG emissions during the whole chain of alternative fuel production. The purpose of this chapter is to discuss the GHG and nitrogen issues of using biomass as an energy source and transportation fuel and to use our knowledge to provide opportunities and criteria for sustainable use of biomass to replace fossil fuels. An overview of the most important drivers and interactions considered in this chapter is shown in Figure 4..

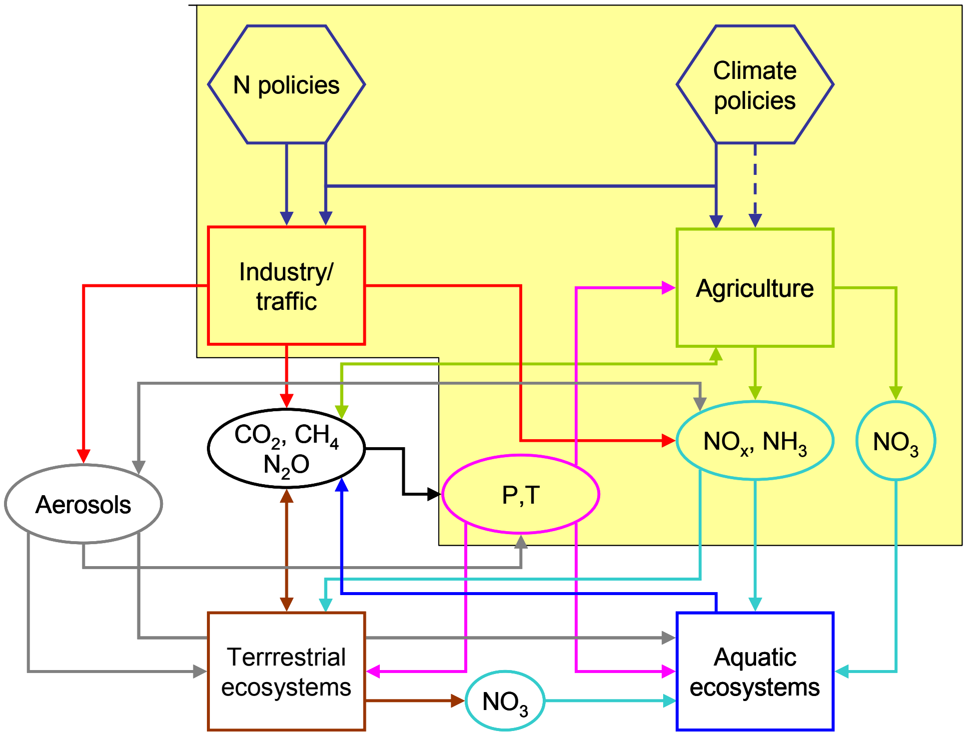


Figure 4.1 Overview of drivers and interactions considered for assessment of impact of biofuel use and land use change on GHG emissions in the energy sector and agriculture.

4.2 Specific overview of linkages

Biomass can be used to produce a wide range of products for energy use in several ways. Figure 4.2 provides an overview of the different technological and product routes. In general the following products can be distinguished for the Dutch and European markets:

- Solid fuels (torrefaction) and liquid fuels (pyrolysis) as pre-treatment options
- Electricity and heat (firing, co-firing, gasification)
- Transportation fuels:
  - Biodiesel (fatty acid methyl ester, FAME, or fatty acid ethyl ester, FAEE) from rapeseed (RME), soybeans (SME), sunflowers, coconuts, recycled cooking oil
  - Pure plant oil (rapeseed)
  - Bio-ethanol (E100, E85, E10, ETBE) from grains or seeds (corn, wheat, potato), sugar crops (sugar beets, sugarcane) or lignocellulosic biomass (wheat straw, switch grass, short rotation woody crops)
  - Fischer-Tropsch diesel and Dimethyl ether (DME) from lignocellulosic waste wood, short-rotation woody crops (poplar, willow), switch grass
- Synthetic (or Substitute) natural gas (SNG) and/or biogas (from digestion)
- Chemicals

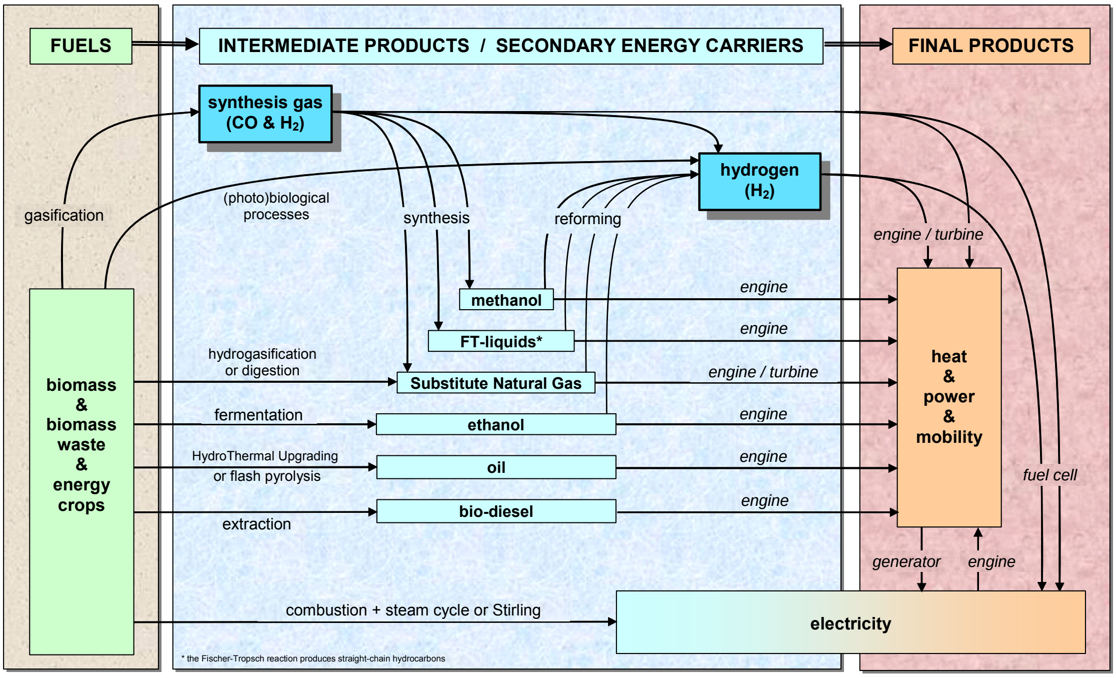


Figure 4.2 Scheme showing the biomass technological and product routes.

There are many possible combinations of type of biomass, conversion technology and end products, which all have their own environmental pressures and interlinkages. These will not be treated separately in this assessment. Currently first and second generation transport fuels are considered. The basis for this difference is the competition with food (seeds, beans pure plant oil) and the improved GHG balance (second generation). One could also distinguish between:

Simple:
- Combustion (stoves, ovens) for heat and power
- co-combustion and co-gasification in coal fired power plants for heat and power
- Digestion for heat and power
- First generation transport fuels

Advanced:
- Small-scale gasification for heat and power
- Large-scale gasification for Fischer-Tropsch diesel, Syngas production, Substitute Natural gas together with heat and power

Intelligent:
- Biorefinery for chemicals and transport fuels, together with heat and power

The efficiency with which the biomass is used and with which the products are made, increases from simple to intelligent application. For advanced and especially intelligent applications technology development is necessary.

The interactions between nitrogen and land use and GHG emissions can be regarded similar to those for food in agriculture when energy crops or crop residues are considered. The only exception is when biogas or energy from manure is produced, because then the additional cycle of animal food and manure plays a role in the interlinkages, making it more complex. The way the biomass is converted into energy makes the difference in the GHG balance. Figure 4.3 shows the relationships between nitrogen, landuse and GHG. The main interactions are:

1. Biomass cultivation and fertilizer use
2. Energy for harvesting and transport
3. Carbon emission from the land (deforestation, land use change) and sequestration
4. Pre-treatment, transport
5. Production of fuel: directly from biomass, or indirectly through the food/animal/manure chain

6. Use of fuel

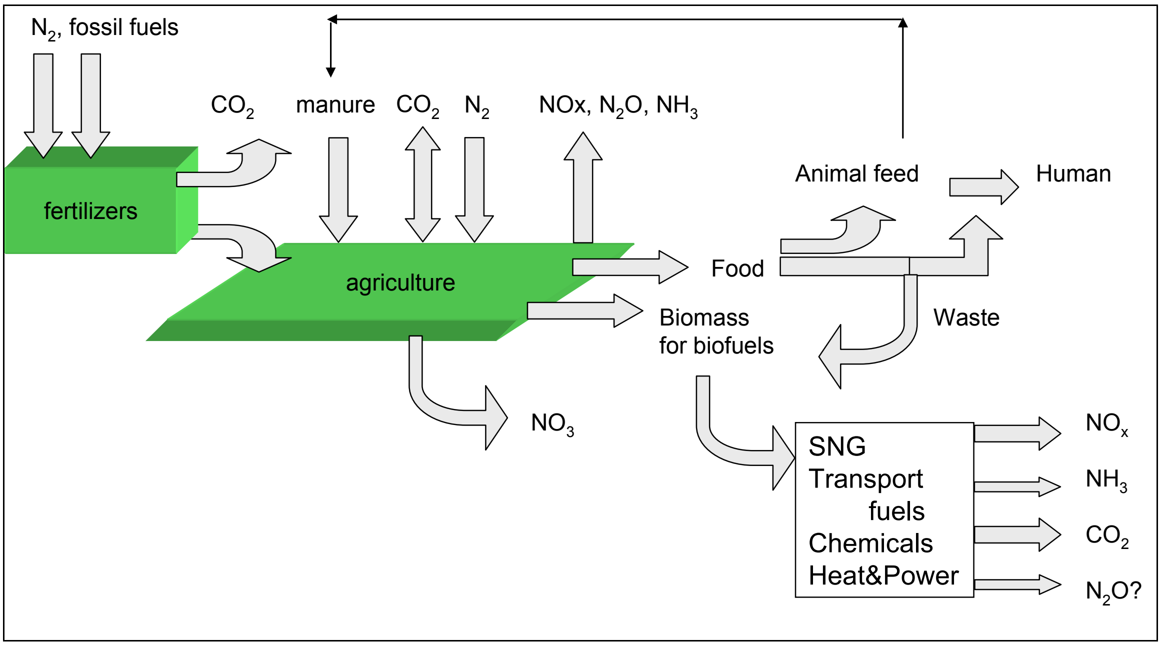


Figure 4.3 Scheme showing the interaction between inputs and outputs.

Overall the literature shows that from these 6 issues generally the emissions of GHG from distribution and transport are small (<10%) compared to the cultivation and production and use of fuel (Delucchi, 2006; Quirin *et al*., 2004). For the assessment we will therefore focus on No. 1 and 5-6. In the next paragraphs we will quantify the interlinkages for the fertilizer production and application, fuel production and use. The conclusions will provide the most important interlinkages and the means to quantify them.

4.2.1 Fertilizer production, application and biomass yield

Fertilizer production
Fertilizer production was in the past mainly based on gasification of coal. Currently natural gas is used to produce the ammonia, which is the main basis for fertilizers. To-day, fertilizer production consumes approximately 1.2% of the world's energy (5% of the natural gas use) and is responsible for approximately 1.2% of the total emission of the Greenhouse gases in the world, consisting of 0.3% of pure CO_2, 0.3% as N_2O and 0.6% as flue gas CO_2 (Kongshaug,1998). Increased focus on energy issues during the last 25-30 years has already caused a positive downward trend both for energy consumption and Greenhouse gas emissions (Smil, 2001). An example of a modern ammonia plant is given in Table 4.1. The current CO_2 emission from fertilizer production world wide that is used for producing animal feed is 41 Tg CO_2 eq for 14 Tg of NH_3 produced (Seinfeld *et al*., 2006). In 2000 the energy needed to produce one ton of NH_3 in the most efficient plants amounted 26 GJ (Smil, 2001), FAO ranged from 40 (natural gas) to 50 GJ.ton^{-1} (coal). The amount of ammonia produced in 2004 amounted to 142 Tg for which about 5700 PJ was needed. If a carbon efficiency of 100% is assumed (literature states 98-99%), about 17 ton C.TJ^{-1} is formed and then the CO_2 produced for global ammonia synthesis amounted to 350 Tg CO_2. 82% is for fertilizer use: 287 Tg CO_2 (or 2.5 kg CO_2/kg fertilizer).

Table 4.1 Energy Balance for Modern Ammonia Plant (Kongshaug, 1998).

	GJ / t NH_3	GJ / t N	t CO_2 / t N
Feed	25	30.5	1.74
Fuel	5.6	6.8	0.39
Gas Consumption	30.6	37.3	2.13
Electricity	0.2	0.2	0.01
Steam Export	-2.5	-3	-0.17
Net Balance	**28.3**	**34.5**	**1.97**

Approximately 82% of the natural gas is used as feedstock, while 18% is used as fuel. Including the energy credit, 88% of the net energy consumption is used as feed. The energy loss for production of electrical energy is not included (50% for Combined Cycle and 65% for Steam Turbine). Average net consumption for European plants is assumed to be 39 GJ/t N (28 GJ/t N representing the feed and 11 GJ/t N as net fuel). 30 years ago, the best plants operated with approximately 47 GJ/t N (28 GJ/t N as feed and 19 GJ/t N as net fuel). The energy improvement has consequently also reduced the total CO_2 emission. A modern ammonia plant, given credit for energy export should be charged by a net emission of ~2.0 t CO_2/t NH_3-N, of which ~1.75 t CO_2/t N is pure CO_2 gas generated from feedstock. The average European CO_2 formation in ammonia plants is 2.2 t CO_2/t N, while 30 years ago the net CO_2 emission was around 2.7 t CO_2/t N (Kongshaug, NorskHydro, 1998).

Oxidation of ammonia generates the Greenhouse gas nitrogen oxide (N_2O), giving around 700-1300 ppm in the tail gas. Increases in combustion pressure from 1 to 5 bar has slightly increased the N_2O emission. The global N_2O emission for nitric acid plants is estimated to be 70 Tg CO_2 eq. (EDGAR database). A good average for the European plants is 0.03 t N_2O/t N, corresponding to 9.3 t CO_2-eq/t N (Kongshaug, NorskHydro, 1998). Table 4.2 shows an overview of different estimates of CO_2 and N_2O emissions of fertilizer production. The largest share of GHG emission with fertilizer production is from N_2O. Currently there are secondary and tertiary catalytic converters available that can reduce the N_2O emission with more than 90% (e.g. www.ecn.nl). Under the CDM mechanism several plants are being equipped with these converters. Therefore it is expected that the N_2O emissions will decrease strongly the coming years.

Table 4.2 Overview of emissions of fertilizer production in the literature.

Fertilizer	kg CO_2/kg N in product	kg N_2O/kg N in product	Total kg CO_2 eq/kg N in product	Reference
AN	1.5 – 2.8	0.013 – 0.017	3.0 – 7.1	Wood&Cowie (2004)
CAN	2.6 – 3.2	0.013 – 0.020	3.0 – 9.6	Wood&Cowie (2004)
Urea	0.9 – 4.0		0.9 – 4.0	Wood&Cowie (2004)
UAN	1.3 – 3.4	0.0073 – 0.0075	2.0 – 5.7	Wood&Cowie (2004)
N	2.0 – 2.7	0.03	11.3 – 12.0	Kongshaug, 1998; Smil, 2000
N	3.02	0.00964	6.07	Concawe, 2004
N	3.5	0.0164	8.6	Steinfeld e.a. (2006)
N	3.9			Elsayed M.A. et al. (2003).
N	3.2	0.018	8.8	Börjesson and Berghund (2007)

Fertilizer application

Fertilizer application is the start of the cascade of reactive nitrogen in the system and to the environment. Because of the limited nitrogen efficiency especially at increasing rates of fertilizer application, losses become larger. Losses can be in the form of nitrates to the groundwater, nitrogen oxide emissions to the air, ammonia emissions and N_2O emissions. The CO_2 uptake by the plant increases because of the increase in biomass and becomes lower if more nitrogen is applied and subsequently taken up by the plant (see e.g. Figure 4.4). On the positive side, CO_2 is sequestered in the soil through the addition of fertilizer. The net GHG emission balance for fertilizer application is difficult to quantify because of the large variations in soil, crops, climatic and environmental conditions and management of the fields. It becomes even more complex if the animal manure cycle is introduced in a system (e.g. to eventually produce biogas, heat and power).

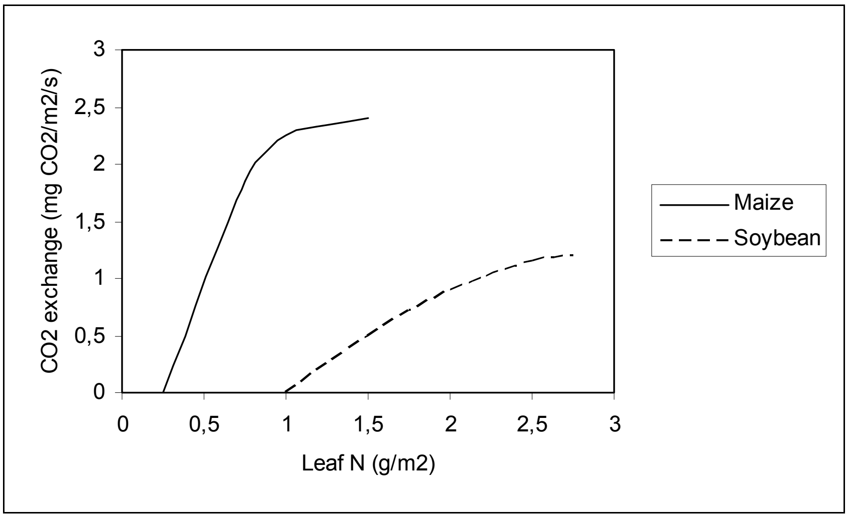


Figure 4.4 Leaf exchange rates at light saturation as a function of leaf N content per unit area (Sinclair and Horie, 1989).

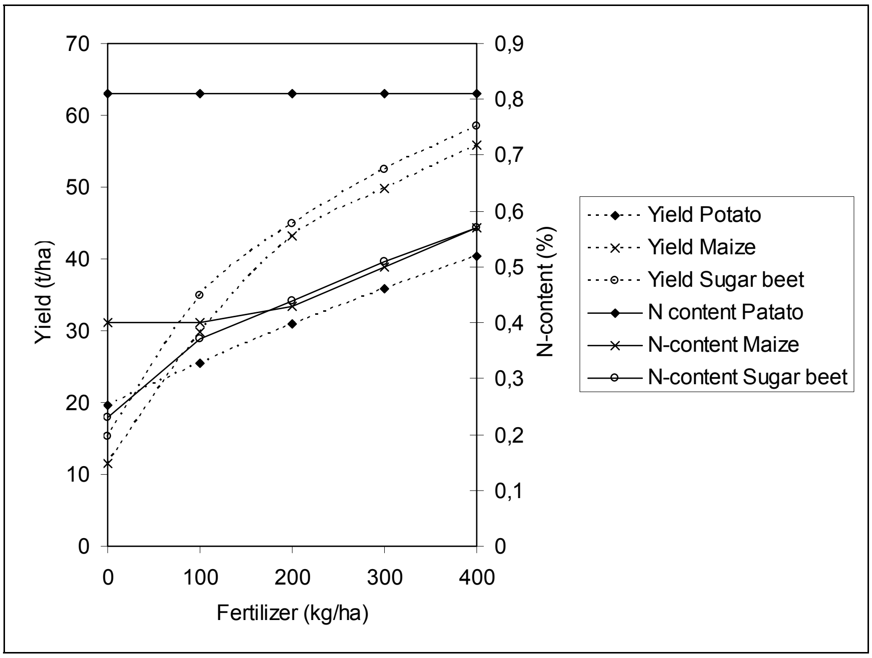


Figure 4.5 Yield and nitrogen content in the crop as function of fertilizer input, calculated with the Nitrogen Crop Response Model for Dutch conditions (http://www.qpais.co.uk/nable/ nitrogen.hm#info).

The application of fertilizer yields higher biomass and thus food and biofuels. At the optimum nitrogen rate of 192 kg N.ha^{-1} (winter wheat in Europe), it is possible to produce 9.3 tonnes of grain per hectare. When no nitrogen fertilizer was added the yield would be: 2.07 t grain.ha^{-1}, a factor 4.5 lower. Several models exist to determine the yield and N content (see e.g. Figure 4.5). In some cases also the nitrogen losses are quantified, but very limited studies regard all the multi-media nitrogen losses. A compilation of the literature is shown in Figure 4.6. Although there is a clear increase of emissions with increase fertilizer/manure application rates, the relation is not consistent. The emissions of NH_3 are the largest part of the airborne emissions and generally (depending on the type of management) larger than NO_3 emissions to the groundwater. The emission factors for NH_3 for fertilizer are about 2% of the input, whereas the N_2O emissions generally range between 1 and 3% of the Kg N added, see Figure 4.7 (Refs, IPCC). Recently, Crutzen *et al.* (2007) proposed that the current N_2O emission estimates are probably underestimated. Based on the annual atmospheric concentration growth they estimate that the total N_2O emission should be about 3-5% of the fertilizer production (and global use). Their estimates are much higher than those used by the IPCC. Nitrogen oxide emissions are much lower from field applications and in the Netherlands and elsewhere in Europe irrelevant.

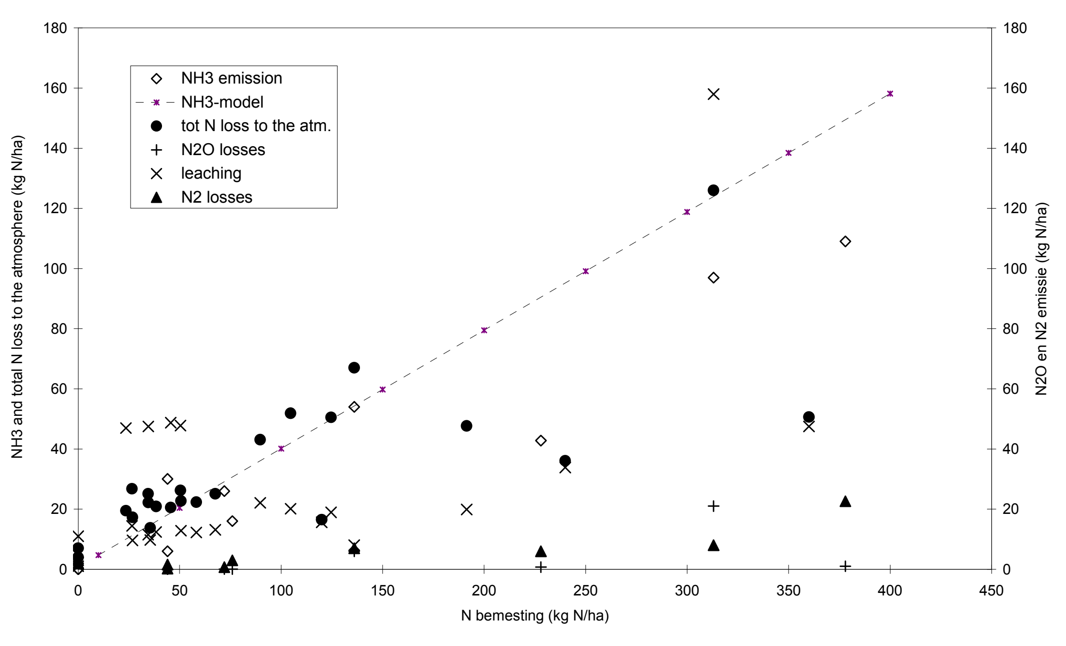


Figure 4.6 Compilation of N losses as function of N-input from the literature (Ruck et al., 1997; Jarvis&Pain, 1997; Sapek, 1984; Menzi et al., 2000).

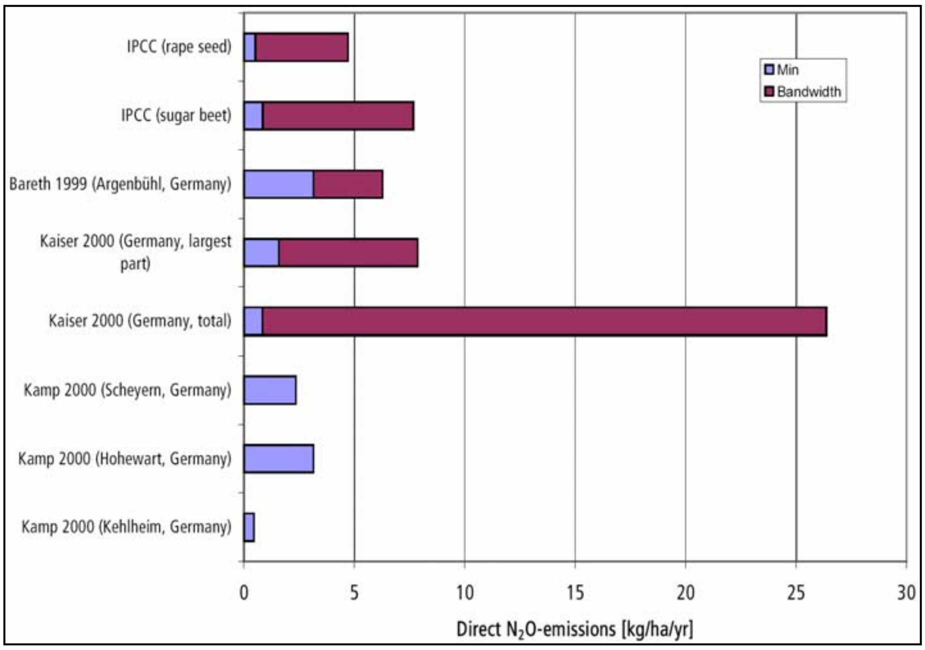


Figure 4.7 Direct N_2O from annual crops, Germany (GM et al., 2002).

Crop production for food or biofuels leads to a different net-exchange of CO_2 if fertilizer is applied and when land use changes. The net exchange depends on the original carbon stock (see e.g. Figure 4.8), the nitrogen stock, fertilizer application rate, the climatic and environmental conditions, the management of the field and the return of crop residues, etc. Soil carbon will eventually saturate and re-release is possible. Furthermore, if wood is used, deforestation might lead to initial CO_2 release from soil carbon. Finally, as shown in Figure 4.4 there is a relation between crop N-content and CO_2 uptake from the atmosphere. Most LCA studies assume no soil/plant carbon contribution (+ or -) to GHG emissions as the relations are too complex. Furthermore, it is assumed that effect is small compared to other GHG and N interactions.

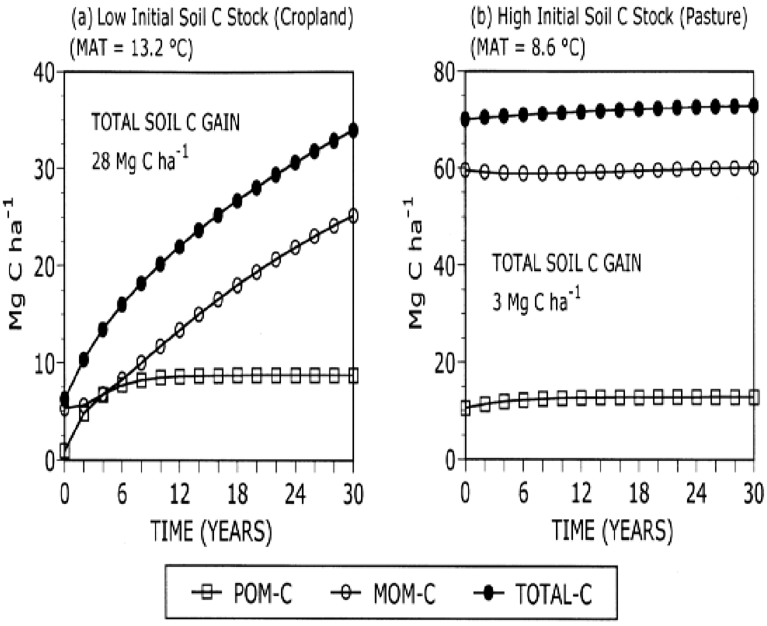


Figure 4.8 Soil carbon storage calculated for low and high initial carbon stock (McLaughlin et al., 2002).

4.2.2 Fuel production and use

Apart from the biomass production in relation to fertilizer use the next main issue for biomass is the fuel production and use efficiency. The efficiency determines the net-gain of GHG emissions compared to conventional fuels. Additional emissions of NO_x might be expected because the fuel-N is higher and/or no de-NO_x installations are used. There are some very detailed LCA studies and overview studies that can be used to provide an assessment of the current and future situation. Weisser (2007) recently made an overview of LCA's of different electric supply technologies, as shown in Figure 4.9. It shows that renewables are much lower in GHG emissions than conventional technologies based of fossil fuels. About 8 studies on biomass were included, showing an average reduction of more than 90% compared to coal and up to 9-% for natural gas.

The direct nitrogen emissions from different options to produce heat and power are given by Pehnt (2006), one of the very few studies providing these data. The data are plotted in Figure 4.10. The pattern for eutrophication is rather different: electricity generating systems excluding biomass are considerably better than the reference mix, but biomass systems are well above the reference mix (exception: systems with co-combustion of forest wood). This is due in particular to the fact that the NO_x emissions of small systems are higher. A special case is the biogas system, which is above the reference mix owing to the ammonia emissions of the agricultural system.

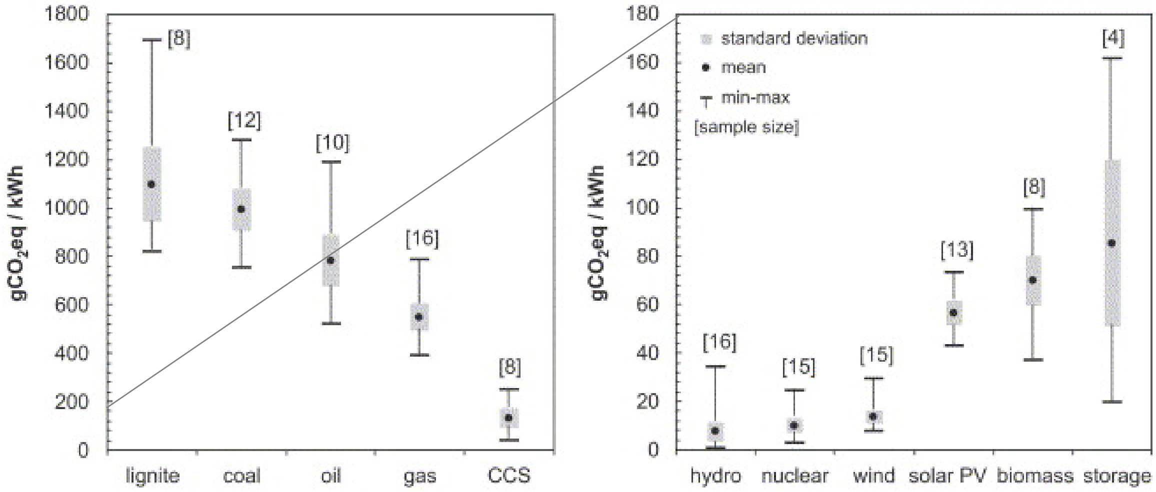


Figure 4.9 Summary of LCA GHG emissions for selected power plants (BWeisser, 2007).

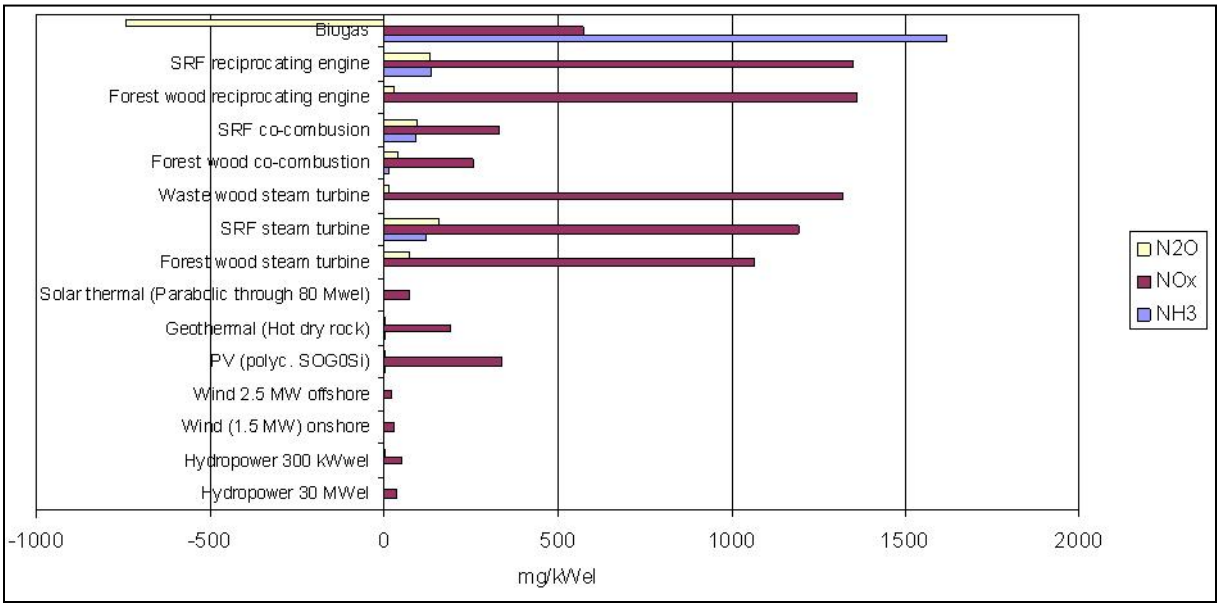


Figure 4.10 LCA nitrogen emissions from biomass options to produce heat and power (Pehnt, 2006).

Most LCA studies consider the whole cycle from biomass production until use, with the exception of EUCAR (2004) who split the well to tank and the tank to wheel analysis for transport fuels. Figure 4.11 provides an overall view of the difference in GHG emissions for the two pathways. The analysis shows that the well to tank is for the biomass options generally smaller than the tank to wheel contribution. The net effect is still better than for conventional options.

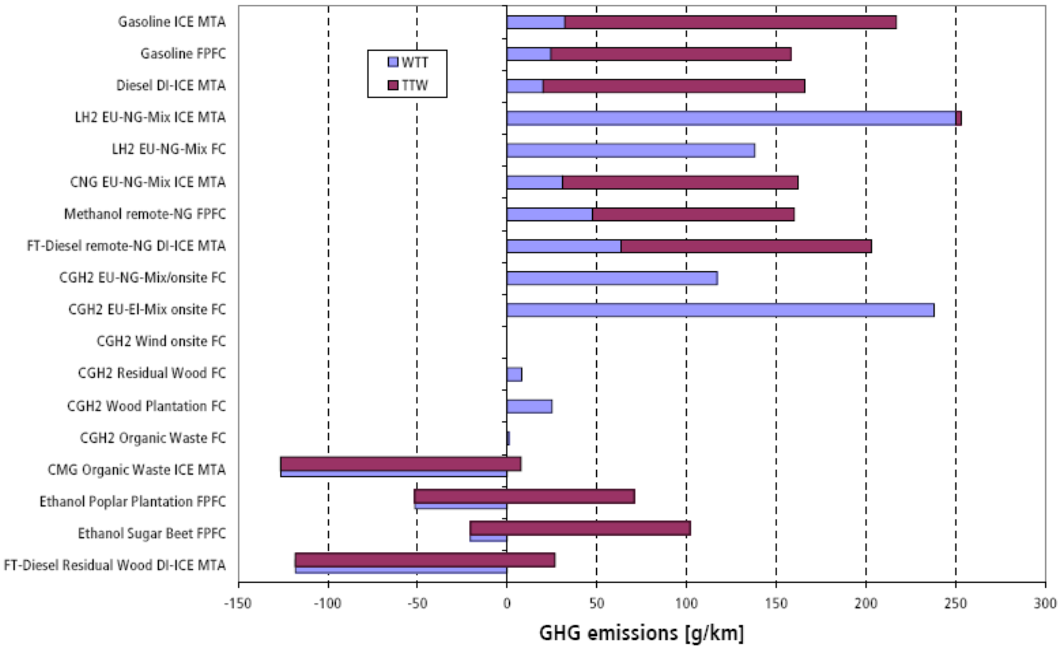


Figure 4.11 LCA GHG emissions from biomass options for transport fuels differentiated between WTT and WTW (EUCAR, 2004).

4.2.3 Overall assessment of interlinkages

The global biofuel system may influence the global environment in a variety of ways. The direct impacts of agriculture on the environment include modification of land for agricultural purposes and by-products of production such as methane and nitrous oxide. Activities such as biomass processing, distribution, and preparation use fossil or biofuels, fuel wood, refrigerants, and other inputs and generate wastes. Indirect impacts include the effects of energy, materials, and pollution entailed in constructing and maintaining equipment, transportation and storage facilities, and other infrastructure used in food production, fisheries, and related activities, and in supporting the populations involved in them. Of course, it is especially difficult to quantify such indirect impacts, to attribute them consistently to particular activities, and to ascertain whether alternative uses of resources would have resulted in greater or lesser impacts. The increased use of biofuels is likely to be a counterproductive approach to mitigate global warming because the fuel energy gained from different biofuel crops might be offset against the nitrogen inputs and associated nitrous oxide emissions from these crops. N_2O is a 300 times more effective greenhouse gas than CO_2 and therefore, a small increase in N_2O emissions resulting from additional fertilizer use can easily offset large CO_2 reductions through the replacement of fossil fuels by biofuels.

The overview studies by Quirin *et al*. (2004), Blottnitz *et al*. (2004), Dellucchi (2006) and Bergsma *et al*. (2007) provide the summary of all the LCA studies conducted so far. LCAs are almost universally set in European or North American context (crops, soil types, agronomic practices, etc.) LCAs are almost universally set in European or North American context (crops, soil types, agronomic practices, etc.). All studies are relatively narrow engineering analyses that assume one set of activities replaces another. The studies conclude that the different energy and GHG balances as well as their further environmental impacts and costs estimations vary greatly as the result of the different assumptions made regarding cultivation, the conversion and valuation of co-products. In general it is concluded that the disadvantage of biofuels from energy crops are the higher level of eutrophication, acidification and ozone depletion associated with their use due to the nitrogen compounds from agricultural production. The difference with GHG is that these environmental impacts act locally or regionally, whereas GHG emissions contribute to the global issues. It is therefore important to consider the location of the emissions of NO_x, NH_3, particles, etc. Local impacts can be due to local high deposition of nitrogen to nature areas or exposure of humans to NO_x and/or particles. NO_x emissions for applications without SCR or de-NO_x are important to consider. These will be especially relevant for the decentral production of heat and power from biogas and the small scale production of biofuels (not cost effective to apply SCR).

The net GHG emissions are smaller than the fossil fuel emissions, especially when second generation crops and technology is used (up to 80% smaller emissions. For first generation the gain is much smaller: 20-40%. Figure 4.12 shows the GHG emissions as averaged over several studies. The largest part is CO_2 from the combustion engine, but N_2O from fertilizer production and application can be a considerable part.

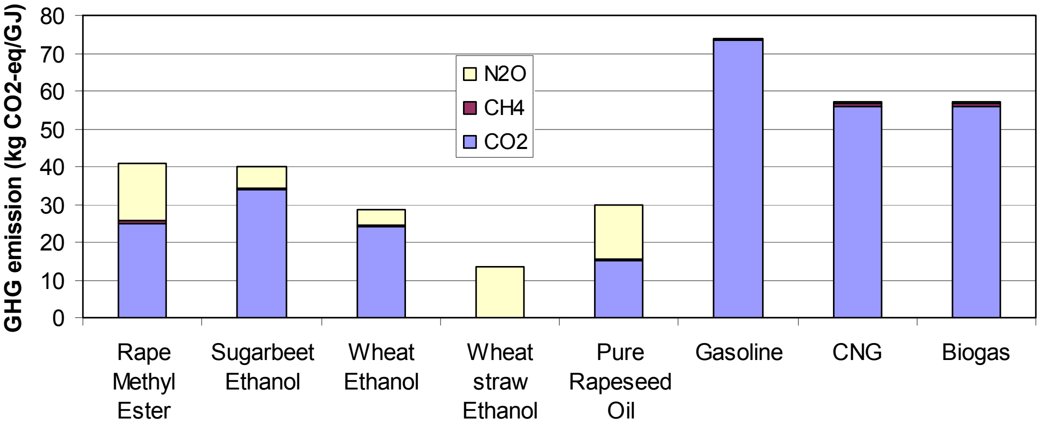


Figure 4.12 Overall estimates of GHG emissions for different options (Elsayed et al., 2003).

Based on the data presented in this chapter we made an estimate of the different emissions contributing to the net GHG emissions for rapeseed oil, replacing current fossil fuel diesel. Figure 4.13 shows the contribution of the different emissions as a function of fertilizer inputs, using relationships between inputs and yields, etc. At very high inputs of 400 kg.ha^{-1} the net reduction of GHG is only 10%, at lower fertilizer input it can even be up to 50% reduction. The largest part is due to the N_2O fertilizer production. Through the application of reduction technologies these can be reduced with 90% yielding better performances in terms of net GHG emission reductions.

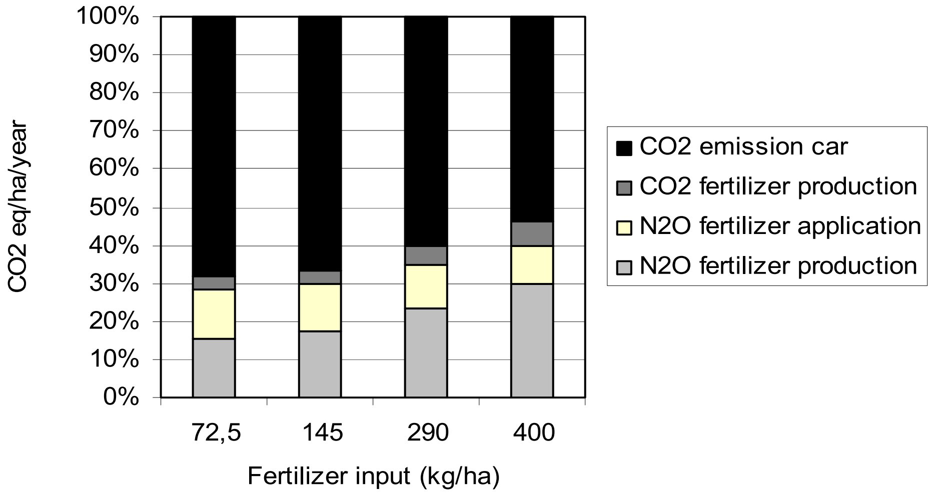


Figure 4.13 *Contribution of different GHG emissions in the production and use of rapeseed oil as diesel in the transport sector.*

We used the best estimate for N_2O and also the data estimated by Crutzen *et al*. (2007), as an extremes to see if there is a point where the net reduction og GHG becomes negative. Figure 4.14 shows the results. It shows that with the estimates of Crutzen *et al*. (2007) the GHG emissions are equal for fossil diesel and rapeseed at fertilizer inputs of 250 kg.ha^{-1}, whereas with the other estimate of N_2O emissions the break-even point is reached above 400 kg.ha^{-1}. This exercise shows the importance of taken N_2O into account in these studies.

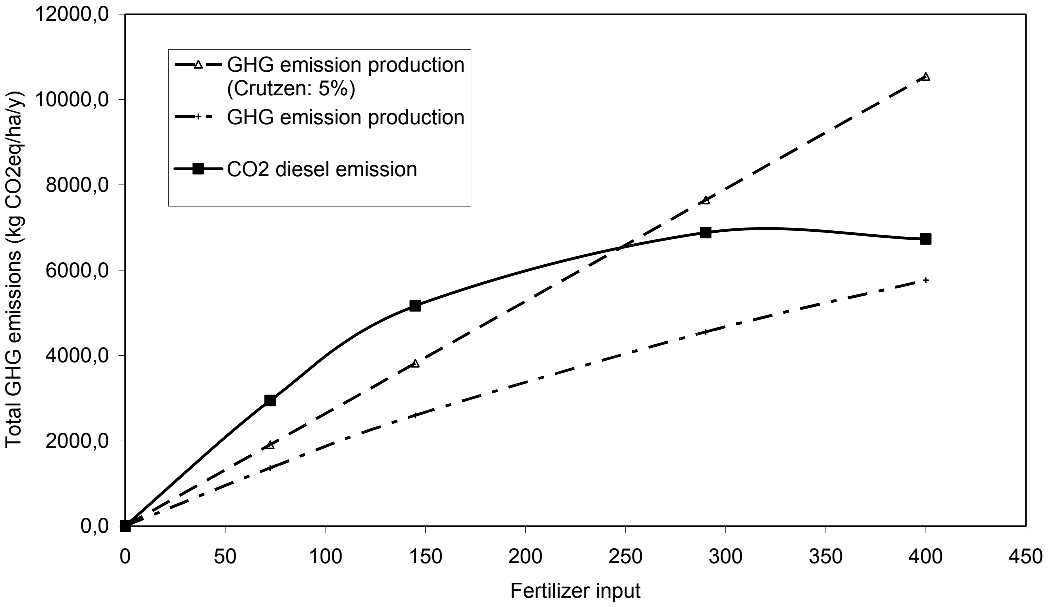


Figure 4.14 *Comparison of break-even points for two estimates of N_2O emissions.*

4.3 Conclusions

There is a broad range of LCA results for GHG mitigation for any given biofuel due to different input assumptions (corresponding to different actual practices) and methods, but some broad conclusions are possible:

- N_2O is a large contributor to the overall GHG balance for (food and) biofuels. N_2O is emitted at fertilizer production, application and further in the cascade (producing animal food, manure, losses to groundwater and atmosphere, etc.)
- The GHG emissions resulting from fertilizer production is dominated by N_2O. Furthermore, N_2O from fertilizer production is the largest N_2O source. Currently catalysts are available to reduce these emissions with more than 90% and are being implemented
- Nitrogen losses increase with increased fertilizer use, similar to the nitrogen-food cascade. Energy crops might use more fertilizer than food because they are produced on marginal grounds and production will be driven by higher ton biomass per ha with a low fertilizer price compared to a high (biofuel) yield and price
- Grain-based biofuels offer less GHG mitigation than lignocelluloses-based fuels due primarily to lower effective yields
- Among commercial biofuels today, sugarcane ethanol gives highest land use efficiency for GHG mitigation
- In longer term, land use efficiency for GHG mitigation is likely to be highest for lignocelluloses plantation biomass (FT or DME in 2010/2015 timeframe, ethanol in 2020/2030 time frame)

Biomass for biofuels vs. biomass for electricity

- Less GHG mitigation per hectare if biomass is used to make biofuels than if it is used to make electricity displacing coal power. (This is true with existing steam cycle biopower technology and more true with future bio-IGCC.)
- If bio-electricity is displacing NGCC electricity or electricity from any fossil-fuel combined heat and power, then biofuels (from sugarcane or from lignocelluloses crops) may give greater GHG mitigation per hectare.
- Cost of GHG mitigation (€/tC avoided) for stationary versus transport applications has not been examined, but likely would be lower for higher GHG mitigation options.

Risk and opportunities of climate policies and nitrogen policies

- Decentralised biofuel production leads to higher NO_x emissions; large scale production (de-NO_x SCR) and fuel use (catalytic converters) do not yield higher NO_x
- (Co-) digestion is a widely promoted option to produce biogas and/or heat and power. However, there is a competition between animal feed and energy/biofuels. Furthermore, it might be sustainable with respect to energy production, but when focussed on GHG reduction only, it is more effective to only use manure digestion without addition of mais.
- Increased biomass production yield higher N emissions in the existing cascade (similar to food production nitrogen cascade)
- 2nd generation biofuels is favourable in relation to energy/GHG balances. However, the competition with animal feed needs attention.
- NL depends on import to reach targets: opportunities to regulated imported ‘commodity’ fuels with N-certificates?

Overall: an increase of NUE and further development of 2nd generation biofuels are necessary.

5 Impact of nitrogen deposition on greenhouse gas emissions in terrestrial ecosystems

5.1 Overview of linkages

Qualitative linkages between the implementation of nitrogen and climate policies in industry/traffic and in agriculture and greenhouse gas emissions in terrestrial and aquatic ecosystems are presented in Figure 5.1. Major linkages for terrestrial ecosystems are:

1. Nitrogen policies affecting NO_x and NH_3 emissions by industry/traffic and in agriculture and N use in agriculture cause a change in
    - N deposition on terrestrial ecosystems, which in turn affects GHG exchange from these systems. The same effect happens when climate policies focused on CO_2 emissions, also affect NO_x and NH_3 emissions. An increase in N deposition leads to enhanced growth, which in turn implies an increased CO_2 sequestration, but it also leads to an enhanced N_2O emission and a reduced CH_4 sink (core aspect of this chapter).
    - N leaching from agriculture and re-emission of N_2O from terrestrial ecosystems. This linkage is specifically reviewed in Chapter 6, but it is also discussed at the end of this chapter.
2. Climate policies affecting GHG emissions by industry/traffic and in agriculture cause a change in the
    - CO_2 concentration in the atmosphere which in turn affects the GHG exchange from terrestrial ecosystems. An increase in CO_2 concentration leads to an increased growth, which in turn implies an increased CO_2 sequestration.
    - Temperature which in turn affects GHG exchange from these systems. An increase in temperature leads to an increased growth, which in turn implies an increased CO_2 sequestration, but it also enhances decomposition processes and through that of CO_2 soil emissions, specifically of peat soil. It also leads to an increase in nitrogen transformations and, through that, N_2O emissions.

Quantification of the latter two linkages requires a quantification of the relationships between (i) GHG exchange by industry/traffic and in agriculture and GHG (specifically CO_2) concentration in the atmosphere, (ii) GHG (specifically CO_2) concentration in the atmosphere and temperature and (iii) changes in CO_2 and temperature on GHG exchange by terrestrial ecosystems. In this chapter, only some indications are given of the impact of changes in CO_2 and temperature on GHG exchange by terrestrial ecosystems in the discussion section. This includes the indirect impact of GHG emissions by terrestrial ecosystems on CO_2 concentration and temperature and through that again on these GHG emissions.

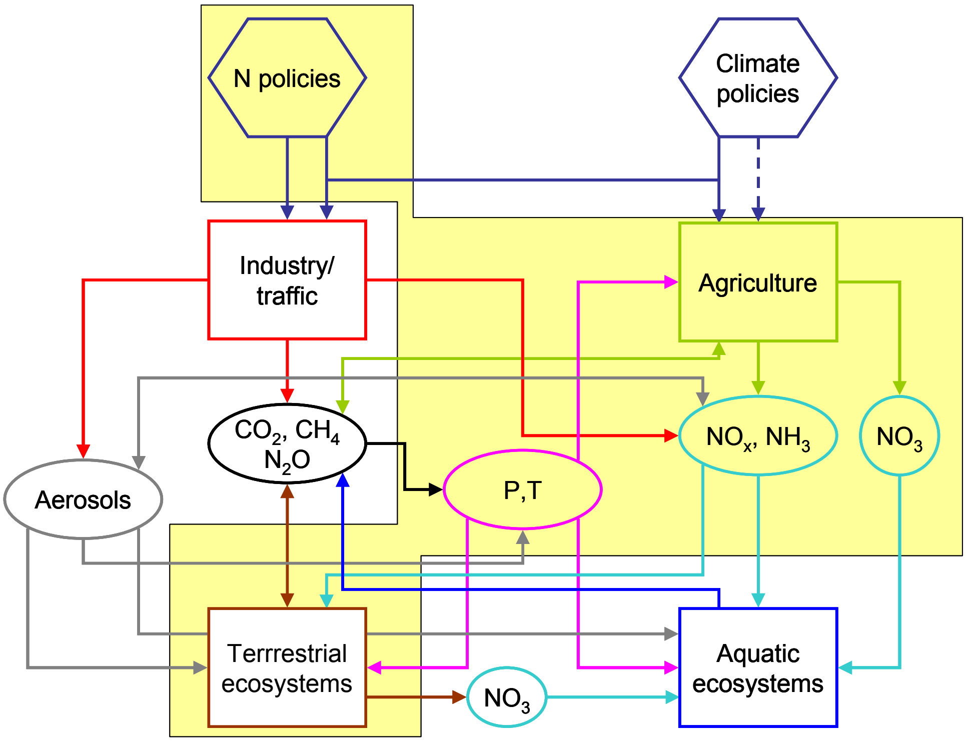


Figure 5.1 Overview of drivers and interactions considered for assessment of the impact of nitrogen deposition on GHG emissions in terrestrial ecosystems.

Production and consumption of N and C trace gases in soils are predominantly due to the microbial processes of mineralization, nitrification, denitrification, methanogenesis and methane oxidation. The interactions between carbon (C) and nitrogen (N) and the exchange of CO_2, N_2O and CH_4 between biosphere and atmosphere are largely controlled by external drivers such as climate (radiation, rainfall, temperature), land use and management (forest type and its management), soil type and deposition of atmospheric N (Conrad, 1996; Groffman *et al.*, 2000). This section summarizes the effects of atmospheric N deposition on the exchanges of CO_2, N_2O and CH_4 between biosphere and atmosphere, using the relational diagram depicted in Figure 2, while briefly describing the effect of other factors controlling the exchange of CO_2, N_2O and CH_4 between terrestrial ecosystems and atmosphere. The focus of the description is on forests, which are the largest non agricultural sink of CO_2 and source of N_2O.

Carbon dioxide exchange

Net primary production of forests greatly depends on climate, forest type, age, nutrient availability and management. Apart from changes in forest management, recent changes in net primary production of forest in Europe (Spiecker *et al.*, 1996) have been attributed to fertilization effects caused by enhanced atmospheric CO_2 concentrations (e.g. Friedlingstein *et al.*, 1995) and N deposition (Holland *et al.*, 1997; Nadelhoffer *et al.*, 1999) and by increased temperatures or by a combination of those effects, increasing the growing season (e.g. Hasenauer *et al.*, 1999).

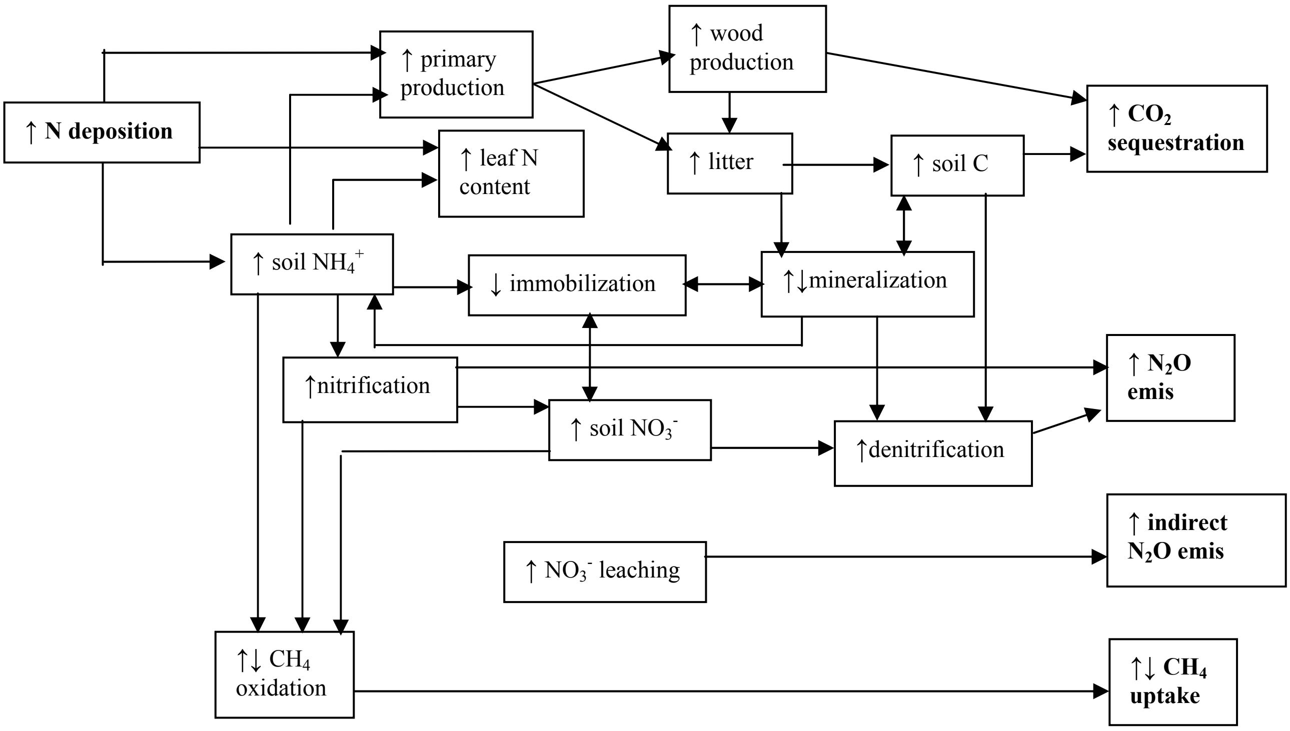


Figure 5.2 Diagram showing the potential effects of increases in atmospheric N deposition on CO_2 sequestration, N_2O emissions (direct and indirect) and CH_4 uptake.

The productivity of many temperate ecosystems is nitrogen limited. Adding N via deposition thus has the potential to increase growth, and therefore to sequester CO_2 from the atmosphere. The increase in N deposition on forests may increase C sequestration by increased growth and increased accumulation of soil organic matter (Figure 5.2). Current hypotheses suggest that increased N deposition causes an increased rate of soil organic matter accumulation through (i) stimulation of growth and subsequent increased leaf/needle biomass and litter production (e.g. Vitousek and Howarth, 1991; Townsend *et al.*, 1996; Schulze *et al.*, 2000), and (ii) or via increased recalcitrance of N-enriched litter, leading to reduced long-term decomposition rates of organic matter, depending on the stage of humus formation (Berg and Matzner, 1997; Harrison *et al.*, 2000; Hagedoorn *et al.*, 2003). Although fertilization by atmospheric N deposition is generally thought to have increased C storage in forest biomass and soils, estimates of the magnitude of this sink vary widely (Peterson and Melillo, 1985; Schindler and Bayley, 1993; Townsend *et al.*, 1996; Holland *et al.*, 1997).

Chronic N additions to temperate forest soils in the US have been shown to cause an initial increase in soil respiration, but continued additions for more than a decade resulted in a reduction in soil respiration of more than 40% (Bowden *et al.*, 2004). The N-content of forest litter and humus might thus be an important indicator of the soil C sequestration. Understanding the N cycle in semi-natural ecosystems is therefore the key to understanding the long-term source or sink strength of soils for carbon. Insight in the soil C sequestration is crucial since the soil is the ultimate sink or source of CO_2 for forest ecosystems over the long term. By far the largest amount of C stored in forests in the northern hemisphere is stored in the soil. More information on the soil CO_2 sink is given in Smith *et al.* (2007).

Nitrous oxide exchange

The production of nitrous oxide (N_2O) in (forest) soils is predominantly due to the microbial processes of nitrification and denitrification (Conrad, 1996). Understanding the relative importance of nitrifiers and denitrifiers in producing N_2O is the key to understanding the mechanism of N_2O production and to accurately up-scaling and quantifying the N_2O source strength of EU forests. Environmental factors influence N_2O production in three different ways; by influencing the nitrification rate, the denitrification rate and the fraction of N products produced that is present as N_2O. These influences are described in literature and can be summarized in qualitative terms (Table 5.1) The process rates of nitrification and denitrification and the N_2O fraction are often inversely related. For more information, we refer to Enzler (2006).

Table 5.1 The influence of environmental factors on nitrification, denitrification and N_2O fractions (fN_2O), based on literature (see text). When an environmental factor is increased, this may have a positive (+) or negative (-) influence on nitrification, denitrification and N_2O production by both processes.

Increase of environmental factor	**Nitrification**	**N_2O fraction nitrification**	**Denitrification**	**N_2O fraction denitrification**
Soil moisture	-	+	+	-
Temperature	+	+	+	-
pH	+ autotrophic	+		
	- heterotrophic	-	+	-
C availability	0 autotrophic	0 autotrophic		
	+ heterotrophic	- heterotrophic	+	-
NH_4 availability	+	+	+	+
NO_3 availability	+	+	+	+

+ = increase- = decrease 0 = no effect

Atmospheric N-deposition is a key factor influencing N_2O emissions. The positive relationship between N deposition and N_2O emissions from forest soils has mainly been attributed to the increased availability of N (NH_4^+ and NO_3^-) for the microbial processes of nitrification and denitrification (Rennenberg *et al.*, 1998). Various studies showed that atmospheric N-deposition is a key site parameter influencing N_2O emissions (see De Vries *et al.*, 2007) and references therein). These studies have shown that temperate forests can function as significant sources for N_2O, especially if these forests are affected over decades by high rates of atmospheric N-deposition. In addition to atmospheric N deposition, many commercial forests receive applications of fertilizer N in the form of urea or ammonium nitrate. Such applications have been shown to double N_2O emission rates from forest soils (Regina *et al.*, 1998). Furthermore, an increased input of N deposition affects the N leaching/runoff from forests (Dise *et al.*, 1998a; Gundersen *et al.*, 1998; De Vries *et al.*, 2003a), causing an elevated indirect N_2O emission from surface waters (Figure 5.1). Finally, atmospheric N deposition may also increase the emission of the secondary radiatively active trace gas NO from forest soils (see De Vries *et al.,* 2007, and references therein).

The increase in N deposition on forests may increase C sequestration and lower the pH through soil acidification. Increases in C sequestration in soils may increase N_2O emissions because of an increased denitrification potential. These increases in N_2O emissions, converted into CO_2-equivalent emissions, may largely offset the C sequestered (Li *et al.*, 2005). More information on the impact of soil C store on N_2O emissions and on GHG emissions is given by Lemke and Janzen (2006). Inversely, a decrease in soil pH may lower the denitrification, but it increases the ratio of N_2O to N_2, thus making the impact of soil pH on N_2O emission less predictable.

Methane exchange

The production and consumption of CH_4 within the soil profile is controlled by several factors, of which oxygen pressure (pO_2), methane partial pressure (pCH_4), and temperature are the most important. Increased atmospheric N deposition increases [NH_4^+] in the soil and usually decreases CH_4 uptake by well-drained soils (Van den Pol-van Dasselaar *et al.*, 1999; Le Mer and Roger, 2001). Three mechanisms have been postulated for the partial inhibition (slow down) of methane uptake by well-drained soils in response to increased N input; a) competitive inhibition of the methane mono-oxygenase by ammonia, b) inhibition of methane consumption by toxic intermediates and end products of methanotrophic ammonia oxidation such as hydroxylamine and nitrite, or c) osmotic stress due to high concentrations of nitrate and/or ammonium (Bodelier and Laanbroek, 2004). In addition to the potential for chronic atmospheric N deposition to alter the size of this sink, other sources of N input to soils have been found to have a significant effect on the forest soil methane sink. For instance, Reay *et al.* (2005) reported vastly differing methane oxidation potentials in soils under different vegetation types, with soils under alder having almost no capacity for methane oxidation even under optimal conditions. This was apparently due to inhibition of methane oxidation by the elevated N concentrations in the soils that result from the N-fixing *Frankia* sp. in the alder root nodules. Similarly, the practice of N fertilization, as used in many commercial forests to increase productivity, has been shown to significantly reduce methane consumption rates (Chan *et al.*, 2005). Summarizing, the mechanisms of increased N availability through increased atmospheric N deposition on CH_4 oxidation is still poorly understood. The mechanisms may differ from site to site, tipping the balance from inhibition to no effect or even increase of CH_4 oxidation (see Figure 5.2).

5.2 Quantification of nitrogen deposition impacts on net green house gas exchange

The quantification of nitrogen deposition impacts on net green house gas exchange is focused on European forest ecosystems. This is because most of the research into the processes of N saturation to which C dynamics is related, has been targeted on forested ecosystems (e.g. Dise *et al.*, 1998a; Gundersen *et al.*, 1998; Aber *et al.*, 2003; De Vries *et al.*, 2003a) and because a large European monitoring network exists (e.g. De Vries *et al.*, 2003b). For unforested areas, such as heathlands, moorlands and acid grassland, understanding of N dynamics remains comparatively poor (NEGTAP, 2001). We thus focus on quantification for forest ecosystems (section 5.2.1), and evaluate the uncertainty in the estimates and the applicability for terrestrial ecosystems in general, by comparing the results with available information for unforested areas including heatlands and grasslands (section 5.2.2). More information on the quantification of nitrogen deposition impacts on net green house gas exchange of European forest ecosystems is given in De Vries *et al.* (2007a). The role of N deposition on CO_2 and CH_4 exchange in wetlands (peatlands) has not been explored in this study due to the limited time frame.

5.2.1 Quantification for European forest ecosystems

Approach

Quantification of the effects of atmospheric N deposition on the net exchange of CO_2, N_2O and CH_4 from forest ecosystems is based on experimental field data and simple empirical model approaches as summarized below

Carbon: The effects of atmospheric N deposition on the net carbon exchange (by CO_2) is assessed by the extent to which reactive N inputs cause increased carbon (C) accumulation from either increased litter production and/or suppressed litter decomposition. This ratio is based on experimental field data, including ^{15}N tracer experiments, and model studies upscaling the results. The question is what the ratio of C accumulation to N accumulation, C/Nseq, is. Where C/Nseq is high, more of the N added to the system will be associated with increased C storage, change in organic soil C/N will be small, and N saturation will occur slowly. Where C/Nseq is low, C pool changes will be smaller, soil C/N will decrease more rapidly, and N saturation will occur faster (Evans *et al.*, 2006). Both the magnitude of N-stimulated C sequestration and the development of N saturation in time is thus related to the value of C/Nseq, being the major uncertainty in the estimation of current and future terrestrial C sinks of both forested and unforested ecosystems.

Nitrous oxides and methane: The effects of atmospheric N deposition on the net exchange by N_2O and CH_4 are based on empirical field data, allowing simple regression relationships, that are upscaled to the European level (by CO_2). In case of N_2O the regression relationships are also based on results of a process based model application (derivation of a metamodel).

Upscaling: The upscaling of relationships for forests and forest soils to a European level is based on data for precipitation, temperature, atmospheric N deposition and soil chemistry (C/N and pH data) at: (i) approximately 500 Intensive Forest Monitoring Plots (level II Monitoring programme with measured data except for temperature that is partly estimated and (ii) approximately 6000 plots at a systematic grid throughout the whole of Europe (level I Monitoring

Programme with interpolated precipitation and temperature data, modelled N deposition data and measured soil chemistry (C/N and pH data). The data at level II plots were used for deriving relationships between N deposition and N accumulation while the available data for all level I plots, representing a total area of 162 million ha, were used for the upscaling

Impact of nitrogen deposition on carbon dioxide exchange

The methodology used to calculate the impact of increased N deposition on C sequestration by European forests is inspired by the approach of Nadelhoffer *et al.* (1999). These authors assessed the additional C sequestration on a global scale from additional N uptake by trees and N immobilisation in soils in response to N deposition, according to:

$$\Delta\text{C sequestration} = \Delta\text{N deposition x } (\text{frN}_{\text{uptake}} \text{ x C/N}_{\text{stemwood}} + \text{frN}_{\text{immobilisation}} \text{ x C/N}_{\text{soil}}) \quad (1)$$

The basic assumption in this approach is that the additional N uptake or immobilisation is reflected in carbon pool changes due to tree growth and organic matter accumulation according to the C/N ratios of the tree and the soil, respectively (Nadelhoffer *et al.*, 1999; De Vries *et al.*, 2006). Nadelhoffer *et al.* (1999) calculated the C sequestration due to increased N uptake by trees and N immobilization in soils on a global scale, assuming (i) a constant N uptake fraction of 0.05 and a constant N immobilization fraction of 0.70, based on short-term (1-3 year) ^{15}N labelled tracer experiments in 9 temperate forests, and (ii) an average C/N ratio in stem wood of 500 and in forest soils of 30. Using this approach, an additional deposition of 1 kg N.ha^{-1}.yr^{-1} leads to a sequestration of 46 kg C.ha^{-1}.yr^{-1}, of which 25 kg C.ha^{-1}.yr^{-1} is retained in stemwood (0.05 x 500) and 21 kg C.ha^{-1}.yr^{-1} in soil (0.7 x 30). These results suggest that C sequestration in forest trees and in forest soils in response to additional N deposition is of equal magnitude.

De Vries *et al.* (2006) adapted the approach by Nadelhoffer *et al.* (1999), using measured and estimated data at the 6000 level I plots. They included the spatial differences in N deposition for the individual plots (EMEP model estimates). The N uptake fraction ranged between 0.05 and 0.10 with high values in low deposition areas, because of N deficiencies, and low values in high deposition areas. Similarly, the C/N ratios in trees were assumed to range from 250 to 500, with low values in high deposition areas, because of the assumed luxury consumption. The N immobilisation fraction was assumed to be a function of the C/N ratio of the soil organic matter layer and not a constant of 70%. The relationship between the N immobilisation fraction and the C/N ratio was based on a cross-European surveys of forests, (Dise *et al.*, 1998a; Gundersen *et al.*, 1998; De Vries *et al.*, 2003a) showing a significant relationships between the input flux of inorganic N and the N concentration of the forest floor, but no relationship between N input and the C/N ratio, supporting the hypothesis of carbon accumulation due to N fertilisation on a regional scale. For the C/N ratio in the organic layer and mineral layer, we used the measured values at all Level I plots, instead of using a constant value of 30 (De Vries *et al.*, 2006).

De Vries *et al.* (2006) used 1960 as the reference for N deposition (this leads to ‘normal’ or background growth) and calculated the additional N deposition for the period 1961-2000 relative to the reference year 1960, to assess the contribution of increased N deposition on the increase in C pools in trees and soil. Results for the period 1960-2000 indicated an average additional C sequestration in stem wood of 15 Tg.yr^{-1} in response to an additional N input (above the reference N deposition of 1960) of 0.45 Tg.yr^{-1}, which translates to 33.3 kg C/kg N deposited (De Vries *et al.*, 2006). This is close to the 25 kg C/kg N used by Nadelhoffer *et al.* (1999). For soil the C sequestration was 6.7 Tg.yr^{-1}, which translates to 14.8 kg C/kg N deposited. Overall the impact of N deposition on the total C sequestration by trees and soils is estimated near 48 kg C/kg N. Assuming that the long term C sequestration in tree wood, corrected for CO_2 emissions due to harvest and forest fires, is 33% of the carbon pool changes, being an average ratio for Europe (Nabuurs and Schelhaas, 2003), the net carbon sequestration in stem wood is only 11.1 kg C/kg N and the total C sequestration by trees and soils is estimated near 26 kg C/kg N.

Impact of nitrogen deposition on nitrous oxide exchange

There are a large number of controlling variables and complex interactions that influence the net N_2O emission, which would suggest applying a detailed mechanistic model for calculating the effect of atmospheric N deposition on N_2O emission. The problem however is that the application of such a model on a European scale is limited by the large number of data requirements. Consequently, we used simple, transparent and empirical approaches, based on process based model approaches and empirical data sets, because these are currently the most feasible to quantify the effect of anthropogenic N deposition on N_2O emission.

Regression model based on process based model
We derived a regression model, predicting N_2O emissions as a function of stand and site characteristics and environmental factors, including total N deposition, based on the predicted N_2O emissions (in kg N_2O-N.ha^{-1}.yr^{-1}) for European forest soils with the PnET-N-DNDC model for a geographic resolution of 50 km by 50 km (Kesik *et al.*, 2005). Total N deposition was calculated according to De Vries et al (2007a). The best results obtained from the regression analyses, while distinguishing between tree species, was (with N_2O-N emission and N deposition both in kg N.ha^{-1}.yr^{-1}):

$$\begin{aligned} N_2O - N_{emission} = & \, 1.3211 + a \cdot tree\ species + 0.019925 \cdot clay - 0.01329 \cdot C_{\min} \\ & - 0.05877 \cdot T + 0.0006640 \cdot C_{\min} \cdot T - 0.0004592 \cdot \Pr \\ & + 0.000002804 \cdot C_{\min} \cdot \Pr - 0.01006 \cdot pH + 0.018029 \cdot N_{dep,tot} \end{aligned} \qquad (2)$$

with clay = clay content (%), C_{min} is mean value for the C pool in the mineral topsoil (0-30 cm; ton C.ha^{-1}), *pH* = pH-H_2O, *Pr* = annual precipitation (mm.yr^{-1}), *T* = mean annual temperature (^{o}C) and $N_{dep,tot}$ = total N deposition (kg N.ha^{-1}.yr^{-1}). The percentage variance accounted for by this model (r^2_{adj}) is 0.42 and the standard error of observations is estimated to be 0.280 kg N_2O-N.ha^{-1}.yr^{-1}. The regression model (Eq. 2) shows that a change in N deposition of 1 kg N.ha^{-1}.yr^{-1} leads to an increase of approximately 0.018 kg N_2O-N.ha^{-1}.yr^{-1}. This is *1.8% of the N input*, which is almost a factor two higher than the default N_2O emission factor of 1% used by IPCC (IPCC, 1996; Mosier *et al.*, 1998). More information on the model features and an explanation for it is given in De Vries *et al.* (2007a).

As with C sequestration, the effect of elevated atmospheric N deposition on N_2O emissions from European forest soils was assessed by comparing the calculated average N_2O emission in the period 1960-2000 to the emission in the reference year 1960 (reference N deposition rates), using available data for all level I plots. The estimated N_2O-N emission for the year 1960 was estimated at 66 kt N_2O-N, corresponding to an average N_2O emission of 0.41 kg N_2O-N.ha^{-1}.yr^{-1}. Using an average additional N deposition of 2.8 kg N.ha^{-1}.yr^{-1} in the period 1960-2000, this leads to an average increase of 0.05 kg N_2O-N.ha^{-1}.yr^{-1}. Using 162 million ha of forests, the average additional N_2O emission can be estimated at 8.1 kt N_2O-N.yr^{-1}. Comparing this value with the emission in the year 1960, it follows that the impact of N deposition on N_2O emission in the last 40 years is approximately 12%.

Regression models based on field measurements
In the IPCC methodology for accounting N_2O emissions from agriculture (IPCC, 1996), the N_2O released from atmospheric N following its deposition on for example forest soils is simply calculated as a fraction of the amount of NH_3-N lost from agriculture. The emission factor (1% of N lost) for these so-called indirect N_2O emissions from agriculture, multiplied with the total amount of NH_3-N lost from agriculture, provides a rough estimate of the indirect N_2O emissions. Empirical data, relating N_2O emissions to N deposition (Denier van der Gon and Bleeker, 2005) show, however, that the current IPCC default value of 1% for indirect emissions is underestimating the N_2O emission from forests (Figure 5.3).

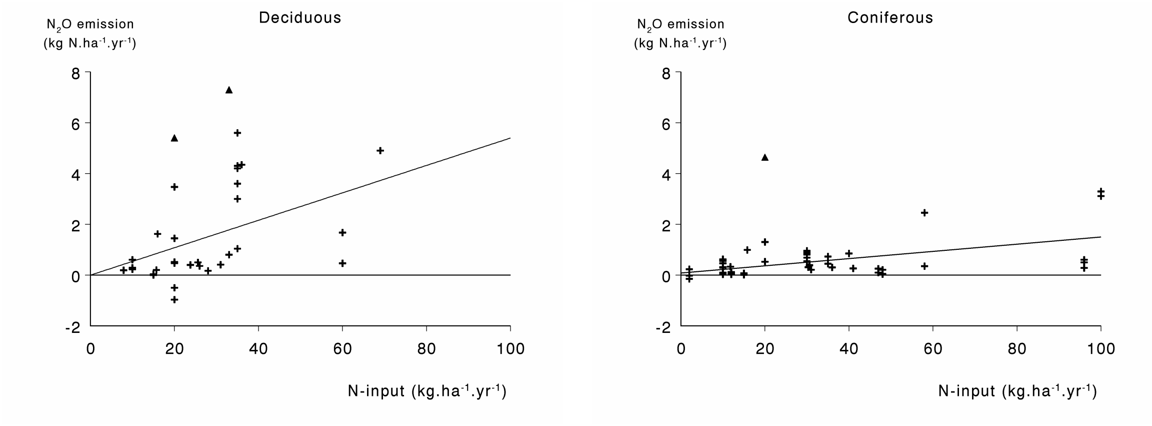


Figure 5.3 N_2O-N emission as a function of N-input for deciduous forests (left) and coniferous forests (right), based on the literature (Denier van der Gon & Bleeker, 2005).

Derivation of an average N_2O emission factor from field measurements depend on the way average emissions are calculated (see Table 5.2). Median values or average values, either weighted by the number of observations or unweighted, which assume no N_2O emission when N deposition is negligible, vary between 2.0-3.7% for coniferous forests and between 4.4-5.7% for deciduous forests. An averaging approach accounting for a certain N_2O emission when N deposition is negligible is linear regression analysis. Results from such an analysis indicate average emission factors of 1.4% for coniferous forests and 5.4% for deciduous forests (Table 5.2).

Table 5.2 Estimated N_2O emission factors for deciduous and coniferous forests based on data by Denier van der Gon & Bleeker (2005).

Type of forests	**N_2O emission factor (%)**			
	Mean[a]	**Weighted mean**[a]	**Median**	**Regression**
Coniferous forests	3.7	2.6	2.0	1.4[b]
Deciduous forests	5.7	5.6	4.4	5.4

a) In calculating mean emission each location and/or study is weighted equally, whereas the weighted mean weighs the average by the number of observations.

b) Intercept of 0.088 kg N_2O-N.ha^{-1}.yr^{-1}

The N_2O emission factor for coniferous forests (1.4%) is close to the value of 1.8% derived by the PnET-DNDC model (Eq. 2). Applying the N deposition– N_2O emission relationships using a regression relation leads to an estimated average annual N_2O-N emission of 0.3 kg N_2O-N.ha^{-1}.yr^{-1} for the year 1960, corresponding to 48.6 kt N_2O-N for the whole of Europe. Using an average additional N deposition of 2.8 kg N.ha^{-1}.yr^{-1} in the period 1960-2000, leads to an average increase of 0.098 kg N_2O-N.ha^{-1}.yr^{-1}, being equal to 15.9 kt N_2O-N.yr^{-1}. Comparing this additional emission to the estimated emission in the reference year 1960, suggests that the impact of N deposition on N_2O emission in the last 40 years has been approximately 33%. This is three times larger than the estimated contribution of N deposition on N_2O emission increase using Eq (2). The impact of 33% is, however, likely to be an overestimation since the regression for deciduous forests implies that the background emission is zero, which is not true. In general, the difference in estimates on effects of N deposition on N_2O exchange is largely due to missing information on N_2O fluxes under unperturbed, pre-industrial and natural conditions, which would allow us to estimate the magnitude of background emissions.

Impact of nitrogen deposition on methane exchange

Increased atmospheric deposition of N increases [NH_4^+] in the soil and usually decreases CH_4 uptake by well-drained soils although reverse effects can also be observed in N deficient forest. For the Höglwald (spruce) in Germany (Butterbach-Bahl *et al.*, 1998; Butterbach-Bahl *et al.*, 2002) found the following relationship:

$$CH_{4,up}\ (\text{kg } CH_4.\text{ha}^{-1}.\text{yr}^{-1}) = 3.04 - 0.041\ NH_4\text{-}N_{in}\ (\text{kg } NH_4\text{-N.ha}^{-1}.\text{yr}^{-1}) \qquad (3)$$

where $CH_{4,up}$ is the methane uptake by oxidation (kg $CH_4.ha^{-1}.yr^{-1}$) and NH_4-N_{in} is the NH_4-N input (kg NH_4-N.$ha^{-1}.yr^{-1}$). Equation (3) indicates that the uptake rate of atmospheric CH_4 is decreased by 4.1% by each kg of NH_4-N entering the system. As with the N_2O emission, a rather extreme impact of N deposition on CH_4 uptake was found for a pine forests in North-East Germany (Butterbach-Bahl *et al.*, 2002), but this result can not be considered representative. Assuming the Höglwald forest representative for Europe and considering an additional average NH_4 deposition of 1.2 kg N.$ha^{-1}.yr^{-1}$ in the period 1960-2000, this may thus have caused an average decrease in atmospheric CH_4 uptake of 0.05 kg $CH_4.ha^{-1}.yr^{-1}$ at this site. This amount equals 1.6% of the total uptake of 3.04 kg $CH_4.ha^{-1}.yr^{-1}$ at zero NH_4-N input (see Eq. 3).

Assuming that the effect of NH_4 input can be scaled to the CH_4 uptake at negligible NH_4-N input, the impact of NH4-N input can in general be derived as:

$$CH_{4,up} = CH_{4,up(NH4\text{-}Nin=0)} - 0.0136\ CH_{4,up(NH4\text{-}Nin=0)} \times NH_4\text{-}N_{in} \tag{4a}$$

where $CH_{4,up(NH4\text{-}Nin=0)}$ is the the methane uptake by oxidation at negligible NH_4-N_{in} input (kg $CH_4.ha^{-1}.yr^{-1}$). Using a European average ration NH_4-N to total N input of 0.43 kg NH_4–N for the period 1960-2000, implies that the impact of an additional N input can be derived as:

$$CH_{4,up} = CH_{4,up(Ndep=0)} - 0.0058\ CH_{4,up(Ndep=0)} \times N_{dep} \tag{4b}$$

Impact of N deposition on net emissions of CO_2, N_2O and CH_4 by European forests

The impact of N deposition on the net exchange of CO_2, N_2O and CH_4 by European forests is shown in Table 5.3. Table 5.3 also contains a comparison of the net exchange of the three investigated greenhouse gases by European forests in terms of their global warming potential (GWP). The GWP is an index defined as the cumulative radiative forcing between the present and a chosen future time horizon, by convention 100 year, caused by a unit mass of gas emitted now, by convention CO_2. Using this approach N_2O and CH_4 emissions are thus expressed in terms of CO_2 equivalents. In this study 1 kg N_2O is assumed to equal 296 kg CO_2 equivalents and 1 kg CH_4 equals 23 kg CO_2 equivalents (Ramaswamy, 2001). Furthermore 1 kgCO_2-C equals 44/12 kg CO_2 equivalents.

Table 5.3 Estimated ranges in long term annual average CO_2, N_2O and CH_4 emissions and the impact of N deposition on those emissions, including a comparison of their global warming potential (GWP) in CO_2 equivalents.

GHG	Emission change (kg.$ha^{-1}.yr^{-1}$) per kgN		GWP change (kg CO_2-$eq^{-1}.ha^{-1}.yr^{-1}$) per kgN	
CO_2-C	-21 to -33	(-26)	-78 to -118	(-100)
N_2O-N	+0.018 to +0.039	(0.029)	+ 7.1 to +17,9	(12.5)
CH_4-C	+0.001 to +0.018	(0.0093)	+ 0.036 to +0.39	(0.21)

1 The N deposition impacts are given for an estimated increase in total N deposition of 1 kg N.$ha^{-1}.yr^{-1}$ (and 0.43 kg NH_4-N.$ha^{-1}.yr^{-1}$)

The average estimated contribution of N deposition to the increase in CO_2 exchange (sink) is near 26 kg C.$ha^{-1}.yr^{-1}$/kg N of which 11.1 kg C.$ha^{-1}.yr^{-1}$ is sequestered in tree wood and 14.8 kg C.$ha^{-1}.yr^{-1}$ in soil. Multiplying this value with 44/12 gives a value near 100 kg CO_2 equivalents. The average contribution of N deposition to the increase in N_2O emission is estimated at 18-40 g $N_2O.ha^{-1}.yr^{-1}$/kg N, being the range found in DNDC model estimates for Europe (first estimate) and in empirical data for coniferous and deciduous forest (second estimate). Multiplying these values with 296 leads to a GWP that is on average 12% of the CO_2 sequestration.

The average contribution of N deposition to the reduction of the CH_4 sink is assessed by using Eq. (4b) and a range in estimated CH_4 sinks for European forest of 0.2 -3.0 kg $CH_4.ha^{-1}.yr^{-1}$r (De Vries *et al.*, 2007a). By doing so, the range in reduction of the CH_4 sink becomes 0.001-0.018 kg $CH_4.ha^{-1}.yr^{-1}$ (Table 5.3). Even though the effect of the increased atmospheric N deposition on methane uptake by forest soils is highly uncertain, given the small number of studies, the complexities involved and the simple upscaling procedures, small. it is clear that the impact on the GWP is negligible compared to the CO_2 sink or N_2O emission effect. In summary, Table 5.3

shows that the C sequestration in response to N deposition is clearly outweighing the increased N_2O emissions and that the effect on the CH_4 sink can be ignored.

5.2.2 Quantification for non forested ecosystems and uncertainty in the N deposition impacts

The uncertainty in the quantification of the impact of elevated N deposition on net GHG exchange by terrestrial ecosystems is mainly determined in the uncertainty in the N deposition impact on the carbon sequestration in biomass and soil. To a lesser extent, it is determined by the uncertainty in N_2O estimate in response to N deposition. In this section, we mention and partly quantify those uncertainties, taking also into account available literature information on non-forested ecosystems

5.2.2.1 Nitrogen deposition and carbon sequestration

A key uncertainty in predicting the impact of N deposition on both forested and non forested ecosystems relates to the extent to which reactive N inputs causes increased carbon (C) accumulation. In general, terrestrial ecosystems will only respond to elevated N inputs if they are N limited. In P-limited (e.g. tropical) ecosystems N additions most likely do not lead to additional CO_2 sequestration. Furthermore, with increasing N-enrichment, soil and vegetation C/N will decline, and less C will be sequestered per unit N deposition. If all incoming N is being leached as NO_3 to surface waters, this N is not contributing to CO_2 sequestration at all. So, N deposition will be most effective at sequestering C in regions of low N availability and thus most likely also low N deposition. Both the magnitude of N-stimulated C sequestration and the impact of N deposition on the development of N saturation are therefore of crucial importance and a major uncertainty in the estimation of future terrestrial C sinks. Both aspects are discussed below.

The magnitude of N-stimulated C sequestration in forests

Fertilization by atmospheric N deposition is generally thought to have increased C storage in forest biomass and soils, although estimates of the magnitude of this sink vary widely (Peterson and Melillo, 1985; Schindler and Bayley, 1993; Townsend *et al.*, 1996; Holland *et al.*, 1997). A range of studies have shown positive forest growth and C accumulation responses under low to moderate N additions (Vitousek and Howarth, 1991; Aber *et al.*, 1995; Bergh *et al.*, 1999; Franklin *et al.*, 2003).

Some studies (e.g Holland *et al.*, 1997) have suggested very large (up to 2.0 $Pg.yr^{-1}$) CO_2 sequestration in forests due to N deposition. These studies assumed that most (~80%) of the deposited N would be stored in woody biomass with a high C/N ratio (250-500). By combining various tracer experiments, Nadelhoffer *et al.* (1999) showed, however, that only a very small part of the added N (~5%) is stored in trees whereas most of the deposited N (~70%) is actually stored in soils with a much lower C/N ratio (10-30). Because of the different C/N ratios, a lot more N is required to lock up C in soils than in woody biomass. In non forested ecosystems the net impact of N deposition on carbon sequestration is even more limited to the soil, since the productivity of forests is generally higher. Nearly all available recent studies, based on both experimental data and modelling show similar results as those obtained by Nadelhoffer *et al.* (1999) as illustrated in Table 5.4.

Table 5.4 Estimated ranges in carbon sequestration per kg nitrogen addition in above and below ground biomass in forest at various scales.

Approach	**Carbon sequestration (kg C/kg N)**			**Scale of application**	**Author**
	Above ground	**Below ground**	**Total**		
Extrapolation of ^{15}N experimental data with world average data	25	21	46	Global average	Nadelhoffer et al. (1999)
Extrapolation of ^{15}N experimental data with site specific data at 6000 plots in Europe	33	15	48	European Average	De Vries et al. (2006)
Multivariate relation between measured growth increase of nearly 400 level 2 plots and influencing factors	15-38	-		Europe	Solberg et al. (2007) Laubhann et al. (2007)
Average results from 30 year low dose (34 kg N/ha/yr) fertilizer experiments	25	-		Forest in Sweden	Högberg et al. (2006)
Average results from 14-30 fertilizer experiments	25 (5-60)	11	36	Forest in Sweden and Finland	Hyvonen et al. (2007)
Range in results of model simulations of five process based models	15-25	-		Two forest sites	Rehfuess et al. (1999)
Range in results of model simulations of three process based models	-	-	10-30	One forest sites	Levy et al. (2004)
Range in results of model simulations of the process based model EFM	-	-	41	22 forest sites	Milne and Van Oijen (2005)
Range in results of model simulations of the process based model SUMO	20-30	-		Dutch average	Wamelink et al. (2007)

Nadelhoffer *et al.* (1999) based their estimate on 15N tracer experiments, showing that approximately 5% of the added N ends up in stem wood with an average C/N ratio of 500, leading to an assumed above ground accumulation of 25 kg C/kg N if the C/N ratio remains constant. Similarly, they assumed 70% N retention in the soil at an average C/N ratio of 30 leading to an assumed below ground accumulation of 21 kg C/kg N. The derivation of data on European scale by (De Vries *et al.*, 2006) was based on the Nadelhoffer *et al.* (1999) approach and has been described before in Section 6.2.1.

Solberg et al. (2007) and Laubhann et al. (2007) investigated the influence of both site and environmental factors on forest growth, which in turn is related to carbon sequestration. They both carried out a multi-factor analysis of measured forest growth data at nearly 400 Intensively monitored forest plots in Europe, including Norway spruce, Scots pine, common beech, European oak and sessile oak. In the study by Solberg et al. (2007), the influence of nitrogen and acid deposition was considered at stand level by using the deposition during the growth period (1993-2000), while the impacts of temperature, precipitation and drought were addressed by taking the deviation of these climatic parameters in the growth period (1993-2000) from the 30-year mean. They simultaneously accounted for site factors influencing measured tree growth, including site productivity, stand age and stand density, all at stand level. Laubhann et al. (2007) applied a multi-factor analysis at tree level, with measured basal-area-increment of each individual tree as responding factor and tree size (tree diameter at breast height, dbh), tree competition (basal area of larger trees, BAL, and stand density index, SDI), site factors (soil C/N ratio, 30-year average temperature) and environmental factors (temperature change compared to long-term average, nitrogen and sulphur deposition) as influencing parameters. The multivariate statistical approach at tree level indicated a 1.2-1.5 % increase in basal area increment (coefficients varying between 0.012 and 0.015 relative increase), depending on tree species in response to a fertilizing effect of N deposition of 1 kg of N.ha^{-1}.yr^{-1}. The approach at stand level indicated a stronger response in N sensitive sites (high soil C/N ratio), having roughly a 1.3-2.2 % increase in growth in response to a fertilizing effect of 1 kg of N.ha^{-1}.yr^{-1}. These responses were recalculated in terms of C sequestration, by multiplying the mean measured volume growth at each stand with the estimated growth increase and the mean wood density of each tree species, assuming a C content of 50%. Results implied an increase in carbon fixation in stem wood of between 15-26 kg C/kg N on all sites and between 20-38 kg C/kg N on N sensitive sites, on the basis of this wide European growth dataset in the period 1993-2000.

Experimental N fertilization results in Sweden and Finland showed an average response near 25 kg C/kg N depending on the site. Hyvonen et al. (2007) investigated the impact of long-term nitrogen addition on carbon stocks in trees and soils in northern Europe (Sweden and Finland). They quantified the effects of fertiliser N on C stocks in trees (stems, stumps, branches, needles, and coarse roots) and soils (organic layer +0–10 cm mineral soil) by analysing data from 15 long-term (14– 30 years) experiments in *Picea abies* and *Pinus sylvestris* stands in Sweden and Finland. Low application rates (30–50 kg N $ha^{-1}yr^{-1}$) were always more efficient per unit of N than high application rates (50–200 kg $N.ha^{-1}.yr^{-1}$). Addition of a cumulative amount of N of 600– 1800 kg $N.ha^{-1}$ resulted in a mean increase in 25 kg C/kg N in tree and 11 kg C/kg N in soil, respectively. The ‘‘N-use efficiency’’ for C sequestration in trees strongly depended on soil N status and increased from close to zero at C/N 25 in the humus layer up to 40 kg C/kg N at C/N 35 and decreased again to about 20 kg C/kg N at C/N 50 when N only was added. In contrast, addition of NPK resulted in higher N-use efficiencies, also at N rich (C/N 25) sites, reflecting a limitation of P and K for tree growth at these sites. N-use efficiency for soil organic carbon (SOC) sequestration was, on average, 3–4 times lower than for tree C sequestration. Högberg *et al.* (2006) reported effects of a long-term (30 years) N fertilization experiment, with annual N loading, on tree growth and soil chemistry in an unpolluted boreal forest. Ammonium nitrate was added to replicated 0.09 ha plots at two doses, of 34 and 68 kg $N.ha^{-1}.yr^{-1}$, respectively. A third treatment of 108 kg $N.ha^{-1}.yr^{-1}$ was terminated after 20 years, allowing assessment of recovery during 10 years. Tree growth initially responded positively to all N treatments, but the longer term response was highly rate dependent with no gain for the highest treatment and a gain of $100m^3.ha^{-1}$ stem wood in excess of the control for the lowest treatment. Assuming a tree wood density of 500 $kg.m^{-3}$ and a C content of 50%, this implies a net C gain of 25.000 kg C at an accumulated N input of 1020 kg (30 years x 34 kg $N.ha^{-1}.yr^{-1}$) implying an ‘‘N-use efficiency’’ =of 25 kg C/kg N. This result seems most appropriate in view of the level of N deposition Adding higher doses of N up to 108 kg $N.ha^{-1}.yr^{-1}$ do not mimic the long-term effects of N deposition at lower rates.

Simulations with process based models also indicate comparable results. For example, Rehfuess et al. (1999) presented simulation results of five process-based models on two forest sites showing a variation 15 - 25 kg C/kg N depending on the model used. Levy et al. (2004) presented a Monte Carlo approach to uncertainty and sensitivity analysis of three ecosystem models, Century, BGC and Hybrid. These models were applied to a coniferous forest ecosystem in Sweden. The best estimate of the change in total carbon content of the ecosystem with the cumulative change in nitrogen deposition over 100 years, was 20.1 kg C/kg N using with a standard deviation of 13.8 kg C/kg N. In an analysis with a complex forest growth model (EFM), parameterized for Norway spruce and Scots pine and tested against measurements from 22 forest locations across Europe, Milne and Van Oijen (2005) showed that the main driver of increased forest growth in the 20th century has been increased nitrogen deposition. A slope of the direct relation between NEP vs. N-deposition gave 150 kg C/kg N, but that slope is an artefact as it is the result of multiple variables acting in concert. The true impact of N-deposition, which was studied by repeating simulations with different values of N-deposition, gave on average a change in NEP of 41 kg C/kg N deposition, where NEP includes both above and below ground C sequestration. F

Wamelink *et al*. (2007) evaluated the impact of N deposition on forest growth by applying the succession model SUMO2 to Dutch forests, using a spatial resolution of 250m*250m grid cells (109374 and 38707 cells for coniferous and deciduous forests, respectively). They simulated an increase in average net carbon sequestration in living biomass, litter and dead wood from 0 to 1.1 $ton.ha^{-1}.yr^{-1}$ for coniferous forest and from 0.4- 2.2 $ton.ha^{-1}yr^{-1}$ for deciduous forest between the lowest (5 $kg.ha^{-1}.yr^{-1}$) and the highest nitrogen deposition level (70 $kg.ha^{-1}yr^{-1}$). The average simulated increase was 20-30 kg C/kg N deposition. The difference between deciduous and coniferous forest is caused in part by the difference in maximum growth rate. The various model results are quite consistent, all showing an average variation between 15 - 40 kg C/kg N depending on the model used and the forest compartment considered (only trees or trees and soil).

In general, all the figures are all well in agreement and show that the range in above ground accumulation of carbon is generally within 15-30 kg C/kg N. The below ground response is less certain although the results by Nadelhoffer *et al.* (1999) and (De Vries *et al.*, 2006) are quite consistent. Model simulations for two forest types (red pine and mixed hardwoods) at Harvard Forest, USA, that best fitted decadal field data for pools and fluxes of C, N and ^{15}N, presented by Currie et al. (2004), suggest however, a net sequestration of approximately 5 (4.8) kg C.ha^{-1}.yr^{-1}/kg N deposition, only. If their data would be representative for Europe, it would imply that the net impact of N deposition on the exchange in greenhouse gases in terms of GWP, is on average in the same order of magnitude as the N_2O estimate in response to N deposition.

The magnitude of N-stimulated C sequestration in non forested systems

Studies of unforested systems show a range of responses to N additions, for example enhanced C sequestration in an Arctic wet sedge system (Johnson *et al.*, 2000); no overall change in C storage in an alpine meadow system (Neff *et al.*, 2002) and decreased C sequestration in European peat bogs associated with the outcompeting of Sphagnum by vascular plants and Polytrichum moss (Berendse *et al.*, 2001). Available recent studies, based on both experimental data and modeling for heatland site in the UK show similar results for below ground soil C accumulation in response to N input as obtained by Nadelhoffer *et al.* (1999) and de Vries et al. (2006) for forests, as illustrated in Table 5.5. The above ground accumulation is, as expected, however less. (compare Table 5.4 and Table 5.5).

Table 5.5 Estimated ranges in carbon sequestration per kg nitrogen addition in above and below ground biomass in moorlands and heathlands at sites in the UK.

Heathland site	**Carbon sequestration (kg C/kg N)**		**Approach**	**Author**
	Above ground	**Below ground**		
Ruabon	15	34	Observed, 40 kg N.ha^{-1}.yr^{-1} addition	Evans et al. (2006)
(Upland heath)	5	23	Observed, 80 kg N.ha^{-1}.yr^{-1} addition	
	9	20	Observed, 120 kg N.ha^{-1}.yr^{-1} addition	
	-	28	Simulated	Evans et al. (2006)
Budworth (Sandy soil)	-	21	Simulated	Evans et al. (2006)
Thursley	9	33	Observed, 15 kg N.ha^{-1}.yr^{-1} addition	Evans, pers. comm.
	-	32	Simulated	

Evidence of carbon accumulation in response to N addition has been presented by Evans et al. (2006) for two heathland N manipulation sites. The first site, Ruabon, is an upland (470 m) heath in North Wales dominated by heather (*Calluna vulgaris*). The manipulation experiment, established in 1989, includes a control treatment plus three N addition treatments of 40, 80 and 120 kg N.ha^{-1}.yr^{-1}, added monthly as finely sprinkled NH_4NO_3 solution. Research at the site included amongst others the N dynamics of the system (Pilkington *et al.*, 2005b; 2005a).

Measurements of soil C pools allowed to calculate changes in the C pools at given N inputs during the 11 year experiment, thus allowing to calculate a C/N sequestration ratio, as presented in Table 5.6. Similarly the ratio between measured increases in vegetation C relative to control plots, and net N inputs under each treatment could be derived as presented earlier in Table 5.5. The system has shown remarkably clear responses to N addition, including increased biomass accumulation rates, and increased N storage in both vegetation and soil.

Table 5.6 Estimated soil carbon sequestration per kg nitrogen addition and soil C/N ratios at the Ruabon heathland N manipulation site (after Evans et al., 2006).

N input in 11 year	**Soil C pool**	**Change in soil C pool compared to ambient**	**dC/dN**	**C/Nsoil**
kg.ha^{-1}	***kg.ha^{-1}***	***kg.ha^{-1}***	***Kg C/kg N***	***Kg C/kg N***
0	105360	-	-	33.4
440	120360	15000	34.1	31.9
880	125640	20280	23.0	31.2
1320	131880	26520	20.1	30.7

The second site, Budworth, is a lowland heath located in Northwest England dominated by heather (*Calluna vulgaris*, with small amounts of *Deschampsia flexuosa*) on humo-ferric podzol soils. Treatments at the Budworth began in 1996, with an experimental design similar to that at Ruabon, and NH_4NO_3 additions of 20, 60 and 120 kg N.ha^{-1}.yr^{-1}. At Budworth, there were no measurements of soil C pools, allowing to calculate a C/N sequestration ratio. Instead Evans *et al*. (2006) applied the model MAGIC at the Budworth and this model best reproduced observed C/N changes at Ruabon at a calibrated value of C/N sequestration of 21 kg C/kg N. Similarly Evans *et al.* (2006) also applied MAGIC at the Ruabon site, which best reproduced observed treatment C pool and C/N changes at a C/N sequestration near 28 kg C/kg N (see Table 5.5). Finally, data for two levels of N addition in a lowland heath land in southern England (Thursley Common), as presented in Power *et al* (2006) were used by the MAGIC model to estimate a C/N sequestration of 32 kg C/kg N (see Table 5.5; Evans, pers. comm.).

Compared to forests, the above ground sequestration is much less varying between 5-15 kg C/kg N. Furthermore, unlike managed forests, where tree removal causes a continuous C sink, in non forest ecosystems the net C sequestration is ultimately negligible with the exception of from managed ecosystems, such as mown grasslands or heath lands with sod cutting.

The impact of N deposition on N saturation in terms of soil C/N ratios

A short-term increase in ecosystem carbon stock may not translate into stable long-term storage. The effects of N deposition on soil organic matter turnover is less clear than effects on production, but in general it may increase decomposition rates in reactive soils/soil pools) and decrease decomposition rates in unreactive soils/soil pools. As a result, the greatest increases in C stock are likely in C-rich, N-poor systems, but not if N deposition triggers species change (e.g. replacement of sphagnum by higher plants, Berendse *et al.*, 2001).

Whether the impact of N deposition on soil C sequestration remains comparable in the future depends on the impact of N deposition on the N saturation. There is ample field evidence that this impact is not as large as previously thought. First of all, a clear increase in soil solution or surface water inorganic N concentrations, due to long-term N saturation has hardly been detected within the limited timescale (1-2 decades) of most available monitoring data. This is partly because the significant variation in runoff NO_3 in response to short-term factors such as climate variability (e.g. Monteith *et al*., 2000; Aber *et al*., 2002) make it difficult to discern slow underlying increases in N leaching within a time scale of 1-2 decades. The absence of clear evidence of long-term rising trends in inorganic N leaching may, however, also be due to a limited effect of N deposition on the C/N ratio, which is an important indicator for N leaching and N accumulation. A number of studies in European forests receiving intermediate to high levels of N deposition demonstrate an inverse relationship between the leaching flux of inorganic N and the soil C:N ratio (e.g. Matzner and Grosholz, 1997; Dise *et al.*, 1998b; 1998a; Gundersen *et al.*, 1998; MacDonald *et al.*, 2002; De Vries *et al.*, 2007b; Van der Salm *et al.*, 2007). Furthermore, observed soil and soil solution responses support the use of a conceptual model in which soil C/N ratios provide a predictive index of site N-enrichment, levels of N immobilisation, and inorganic N leaching, such as the MAGIC model applications presented before (Evans *et al.*, 2006).

Where N deposition does lead to C accumulation, N-enrichment of the soil (expressed in terms of C/N ratio) will be slowed, or potentially halted. In a cross-European survey of forests, Dise *et al*. (1998b; 1998a) found significant relationships between the input flux of inorganic N and the

N concentration of the forest floor, but no relationship between N input and the C/N ratio, supporting the hypothesis of carbon accumulation due to N fertilization on a regional scale. In studies in which a weak negative relationship between C/N and N deposition was found, this effect was predominantly attributed to the confounding influence of a parallel climate gradient, with low decomposition rates, and thus high forest floor C/N, in northern Scandinavian sites (Kristensen *et al*., 2004).

In peaty soils, the impact of N deposition on N saturation is likely to be even lower. In these systems decomposition rates are low due to cold, wet and/or acid conditions. Any N-induced increase in litter production, or decrease in long-term litter degradability, will therefore lead to increased organic matter build-up. Since many organic-rich soils (such as ombotrophic bogs) are nutrient-poor, growth stimulation in response to N addition may also be amplified. At the extreme, C accumulation may keep pace with N accumulation such that no C/N ratio decrease occurs. The presented experimental results for heathlands on moorlands presented before also show that the impact of N deposition on soil C/N ratio is quite limited due to C accumulation. In the Ruabon site, the calculated values of C/N sequestration at the low N treatment (still 40kg $N.ha^{-1}.yr^{-1}$) were nearly equal to the observed organic soil C/N ratio (34.1 compared to 33.4; see Table 5.6). This value implies a high level of increased C storage under elevated N, which strongly slows the onset of N saturation. Only at very high N inputs, a progressive reduction in the C/N ratio of the soil of 33.4 in the control to 30.7 was observed In Buadon since the C/N sequestration drops below the original soil C/N ratio (see Table 5.5). However, recent search in European peat bogs showed that also at low N deposition a negative correlation between N deposition and C/N ration exists (Bragazza *et al*., 2006). In addition, Bragazza *et al*., (2006) found a positive relationship between N deposition and C sequestration for these European peat bogs.

By contrast, at warmer and dryer sites it is likely that much of any additional organic matter produced will be removed due to higher decomposition rates. Thus, the C/N ratio of the small organic pool may be expected to decrease relatively quickly in response to N inputs, leading to the rapid onset of N saturation. Furthermore forest disturbances, either natural as a consequence of e.g. fire, wind, pest or diseases, or managed such as forest logging, may decrease photosynthetic capacity, completely in the case of logging, and converts a forest from a carbon sink into a carbon source (Hymus & Valentini, 2007). The time required for a stand to become C neutral and ultimately sequester C after the initial disturbance depends on the type and intensity of the disturbance and on post disturbance management for evergreen forests (Thornton *et al*., 2002). Periods generally vary between 10 - 12 years but periods near 25 have also been found (Hymus & Valentini, 2007).

5.2.2.2 Nitrogen deposition and nitrous oxide emission

There are large uncertainties in the greenhouse gas emission estimates and in the estimated effects of N deposition on N_2O. The range in values presented in Table 6 may in reality be even larger due to other aspects not included in the quantification, such as the occurrence of forest disturbances, neglecting indirect N_2O emissions and the occurrence of lag times between changes in N deposition and green house gas emissions.

Forest disturbances affect the potential of a forest to sequester carbon as described above, but N_2O emissions can also substantially increase in the period following clear-cutting. In an overview article, Bowden (1986) estimated an increase up to 0.5 kg N_2O-N/ha/yr, but this might be much higher in N saturated forests. First results for such forests indicate huge emissions of even more than 3 kg N_2O-N/ha/yr in the first 4-7 years after clear-cutting (Butterbach Bahl, pers. comm). Including those estimates over the rotation time of a forest could increase the average N_2O emission by approximately 0.2-0.5 kg $N_2O\text{-}N.ha^{-1}.yr^{-1}$, being very substantial compared to the estimates presented before (see also Table 5.3). Similarly, the N leaching is substantially increased in this period. This was already shown in an overview of numerous studies from the 1960s and 1970s in USA (Vitousek and Melillo, 1979) and more recently substantiated by an additional over view of numerous studies since the 1980s in Europe and also partly USA (Gundersen *et al.*, 2006). Their results show that the nitrate concentration in soil and stream waters increase directly after the clear-cut with peak concentrations within 2-3 years after clear-cut. Gundersen *et al.* (2006) found that highest responses (the difference in nitrate concentration between cut and reference stands) were observed in Central Europe (5 mgN.l^{-1} in stream or seepage water as a mean over the region). Assuming an average precipitation excess of 100-300 mm.yr^{-1}, this would imply an increase in N leaching rate of 5-15 kg.ha^{-1}.yr^{-1}. Using the standard fraction of 1.5% for indirect N_2O emission (see below) for each kg of N leached (Mosier *et al.*, 1998), this would cause an estimated increase in indirect N2O emissions of approximately 0.075-0.225 kg N_2O-N.ha^{-1}.yr^{-1}. The nitrate concentration, however, often returns to pre-cutting levels within 5 years.

Indirect N_2O emissions from surface water in response to additional N deposition are induced by additional N leaching/runoff and this aspect has not been included in the calculation because of its extreme uncertainties. In general, N leaching is negligible below an atmospheric input of 10 kg N.ha^{-1}.yr^{-1} (Dise *et al.*, 1998a; Gundersen *et al.*, 1998; De Vries *et al.*, 2003a). At higher N inputs, N leaching is clearly higher in "N enriched" sites (C/N ratio in the organic layer below 23) than in "C enriched" sites (C/N ratio in the organic layer above 23). For the first case, a linear relationship has been derived according to N leaching= - 4.3 + 0.67 x N deposition in kg N.ha^{-1}.yr^{-1} (Gundersen *et al.*, 2006a). Using an average N deposition of 12.3 kg N.ha^{-1}.yr^{-1} for forests in Europe (De Vries *et al.*, 2006), this would imply an average N leaching rate of 5.3 kg N.ha^{-1}.yr^{-1}, being close to 40% of the N deposition. However, in most cases the C/N ratio of the organic layer is above 23 and the N leaching is then generally lower. On average, the N leaching rate at 121 intensively Monitored plots was only near 1 kg N.ha^{-1}.yr^{-1}, being approximately 7% of the N input (near 13 kg N.ha^{-1}.yr^{-1}, De Vries *et al.*, 2001). In the IPCC approach the N leaching is estimated at 30% the N input (Mosier *et al.*, 1998). Using the standard fraction of 2.5% for indirect N_2O emission for each kg of N leached (Mosier *et al.*, 1998) and a leaching fraction varying between 0.1 and 0.4 implies a net N_2O emission ranging from approximately 0.25- 1.0% for each additional kg N deposition. Compared to the average estimated value of 1.8%-4.0% for direct N_2O emission, this implies an average increase of approximately 10-20%, but the uncertainty in this value is high and the contribution may be high at certain plots.

Lag times in N deposition effects occur, which most likely is due to a slow response of the soil C/N ratio that largely affects the availability of N and thereby N_2O emissions in forest soils (Pilegaard *et al.*, 2006). As with carbon, decreases or increases in atmospheric N deposition may thus not cause direct changes in N_2O emissions, since the C/N ratio only changes slowly in soils with a large reservoir of C and N. This is illustrated by Borken *et al.* (2002), describing application of normal "polluted" versus natural, unpolluted precipitation to soil under roofed plots of a 70-year old Norway spruce plantation in Germany. No significant differences in N_2O emission rates were found after 7 years of treatment (Borken *et al.*, 2002). It should be noted that the N_2O emissions from the spruce forest were low (~0.3 kg N_2O-N.ha^{-1}.yr^{-1}) and thus the mitigation potential was limited. Borken et al. (2002) also monitored the CH_4 oxidation rate and, like N_2O emission, no changes in CH_4 oxidation upon 7-year reduction of N deposition were observed. Clearly the results cannot be extrapolated to "high N_2O out" forest systems, neither to forests that are largely N limited like many boreal forests. Further N input manipulations – N_2O response studies in various systems would be an important addition to quantify the mitigation potential of reduced N inputs to reverse N_2O emission levels and CH_4 oxidation capacities of forests exposed to elevated N deposition levels.

5.2.2.3 Nitrogen deposition and greenhouse gas emissions in forested and unforested ecosystems

In general, the difference between forested and unforested ecosystems with respect to the response of N deposition is likely to be small. This is already illustrated before regarding the soil carbon accumulation in response to N input, being the major effect. With respect to impacts on N_2O and CH_4 emissions, effects are also likely to be comparable. This is illustrated in Figure 5.4 showing the net greenhouse gas budget (in CO_2 equivalents) per unit N added by including CO_2, N_2O and CH_4 for European forests and the previously mentioned lowland heath Thursley Common in southern England. The major difference is the impact on biomass accumulation, which is likely to be much larger in forests, specifically on a longer timeframe (e.g 100 years). In that case, it is best to neglect the net biomass increase of unforested sites, whereas a temporary carbon sink in forest can certainly occur for a 100 year period

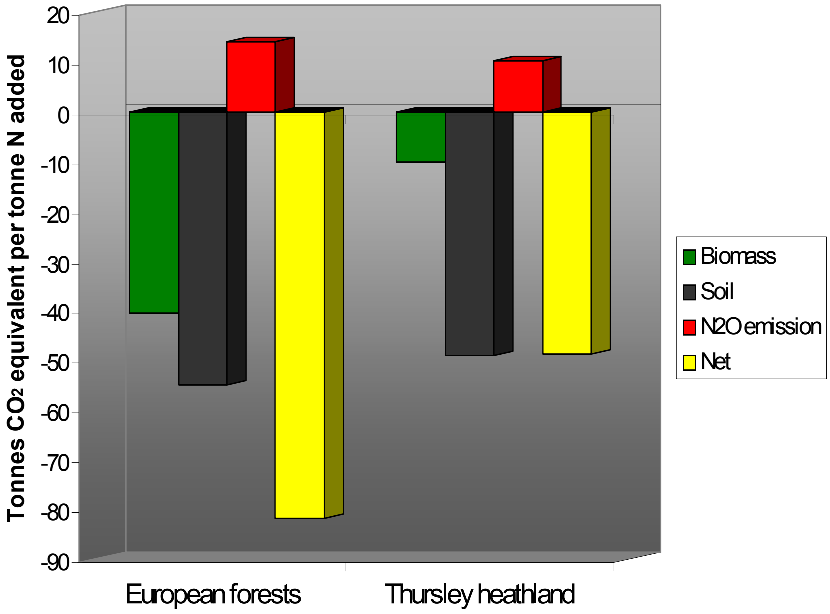


Figure 5.4 Net greenhouse gas budget (in CO_2 equivalents) per unit N added for European forests and for a lowland heath land in southern England (Thursley Common). The forest results are based on De Vries et al. (2006) and the heath land results on Power et al. (2006). (Evans , pers. comm.).

5.3 Discussion and conclusion

5.3.1 Discussion

The efficiency of existing or future measures related to reduced N use requires a full quantification of its effects on the exchange CO_2, N_2O and CH_4 by complete life cycle analyses. The aim of this section is to illustrate this aspect by he quantification of changes of N use in agriculture on GHG emissions from agricultural and non agricultural ecosystems, including aquatic ecosystems. The measures are related to a reduced input of N by either animal manure (by reduction of livestock or reduction of N contents in feed) or N fertilizer. The quantification is done by taking most relevant effects into account, namely:

1. Release of CO_2 and N_2O related to the production of N fertilizer
2. C sequestration in agricultural soils in response to a change in N input
3. C sequestration in terrestrial ecosystems (forests) in response to a change in N (NH_3) input induced by the change in N use, including on-site and off-site sequestration.
4. N_2O exchange from agricultural soils, non agricultural (forest) soils and re-emission from aquatic systems.

The impact of N input on CH_4 exchange from agricultural soils and terrestrial ecosystems (forests) has been considered negligible (see before). An impact not considered in this evaluation include the effect of N content in feed on CH_4 exchange from animals (fermentation) and release or oxidation of stored manure. Up to a certain limit, an increase in N content increases the efficiency (digestibility) of feed implying that more energy (C, N) is directed for the production of useful products (meat, milk etc) so that CH_4 emissions per unit product are decreased. It also implies a reduction of the herd size per unit of product level and thereby another decrease in the CH_4 emission by fermentation. At too high N inputs, however, the digestibility will not be further increased and may even go down. Inversely, lowering the N content in feed, implying higher C/N ratios in feed, lower the methane emissions from manure storage. In performing the quantification, we make use of the results presented in Chapter 3 regarding the CO_2-eq emissions related to the production of N fertilizer and C sequestration in agricultural soils and of Chapter 6 regarding the re-emission of N_2O from aquatic systems due to N leaching from agricultural and terrestrial ecosystems

1 Release of CO_2 and N_2O related to the production of N fertilizer
The gross GHG effects in CO_2-eq emissions of producing 1 kg of N-fertilizer due to N_2O and CO_2 emissions during production depend on the energy source for production, the fertilizer production process and the fertilizer type. In Table 3.5 and Table 4.2 of Chapter 3 and 4, respectively, a range of approximately 1-10 kg CO_2-eq emissions is given for both CO_2 and N_2O emission, mainly based on results by Wood and Cowie (2004). In this exercise we used a mean value of 5 kg CO_2-eq per kg N-fertilizer production.

2 C sequestration in agricultural soils in response a change in N input in agriculture
A decrease in N input may lead to a decrease in C sequestration due to a decrease in growth and related crop residue input. In Table 3.5 in chapter 3, the soil carbon sequestration is estimated at 3-6 kg CO_2-eq/kg N for grassland. This estimate is used in our quantification, while neglecting the effect on crop C sequestration, being a very temporary effect.

3 C sequestration in terrestrial ecosystems in response to a change in NH_3-N input
A change in N input in agriculture leads to a change in N emission, which in turn cause a change in N deposition, affecting the C sequestration by terrestrial ecosystems. The change in carbon sequestration in response to N use is affected by the N deposition which further depends on the ratio between the area of non agricultural versus agricultural areas (including forests). Assuming that all NH_3 emitted by Dutch agriculture is deposited on Dutch agricultural and non agricultural areas, the CO_2 sequestration per kg N use can be calculated as:

$$CO_{2,seq}/N_{use} = (A_{\text{forests}}/A_{\text{total}}) \times frNH_{3\ \text{emission, agriculture}} \times kgCO_2\text{-eq/ kg N}_{\text{forests}} +$$

$$(A_{\text{other terrestrial ecosystems}}/A_{\text{total}}) \times frNH_{3\ \text{emission, agriculture}} \times kgCO_2\text{-eq/ kg N}_{\text{other terrestrial ecosystems}}$$

where $A_{\text{total}} = A_{\text{agriculture + forests+ other terrestrial ecosystems}}$

Using an agricultural area in the Netherlands of 2 million hectare, a forested area of 0.2 million hectare and an area of other terrestrial ecosystems 0.2 million hectare, considering also other land use (mainly infrastructure) an average value of approximately 0.1 seems reasonable for the ratio of both $A_{\text{forests}}/A_{\text{total}}$ and $A_{\text{other terrestrial ecosystems}}/A_{\text{total}}$

To assess the overall impact of N use in agriculture on C sequestration in non agricultural systems, it is needed to assess the NH_3 emission per kg change in N use. This depends on the type of N used (N fertilizer or type of animal manure). On average, the following values can be used for the Netherlands:

Animal manure:	NH_3 emission on average 23.6% or 0.236 kg NH_3-N.
Fertilizer:	NH_3 emission on average 2.6% or 0.026 kg NH_3-N.
Total:	NH_3 emission on average 14.2 % or 0.142 kg NH_3-N.

The basis for these NH_3 emissions fractions are INITIATOR2 model calculation for 2000, as summarized in Table 5.7, which are based on country accepted emission fractions for NH_3 emission form housing systems and manure application. When applying low emission housing, the effects of N used in agriculture would on average be halved regarding its indirect effects on terrestrial ecosystems assuming the following parameterization (Kros *et al.*, 2003): Pigs 0.3 times poultry 0.1 times and dairy cattle 0.75 times present emissions; Application by injection: grass land 1.15% and arable land 10.35 % (Van der Hoek, 2002). This leads to an average emission reduction from 14.2 % to 7.2% as given in Table 5.8. However, in the calculations, we used the present NH_3 emission factor.

Table 5.7 Estimated NH_3 emission fractions for animal manure, fertilizer and organic products and a country averaged fraction for the Netherlands

Type of manure		**N use (Gg N)**	**N emission (Gg NH_3- N)**	**Emission fraction (%)**
Animal manure	Housing	384	70.4	18.3
	Application	314	37.0	11.8
	Housing + application	384	107.4	27.9
	Grazing	108	8.6	8.0
	Total	492	116	23.6
Fertilizer		306	7.8	2.6
Organic products		11	0.3	3.0
Total		873	124.3	14.2

Table 5.8 Estimated NH_3 emission fractions for animal manure, fertilizer and organic products and a country averaged fraction for the Netherlands assuming low emission housing

Type of manure		**N use (Gg N)**	**N emission (Gg NH_3- N)**	**Emission fraction (%)**
Animal manure	Housing	384	31.6	8.2
	Application	352	17.0	4.8
	Housing + application	384	48.6	13
	Grazing	108	8.6	8.0
	Total	492	57.2	11.7
Fertilizer		306	7.8	2.6
Organic products		11	0.3	3.0
Total		912	65.4	7.2

The above ground carbon sequestration in forests is approximately 15-30 kg C per kg N deposition, as shown in Table 5.4. In the previous estimates for forests, we assumed that wood which is harvested and removed from a site is ultimately released as CO_2 into the atmosphere. We thus only account for the C sequestered in standing biomass, being only 1/3 of the total amount (2/3 is removed). However, often harvested wood can reside in solid wood products, recycled products or landfills for centuries. A sometimes large fraction of harvested wood is also used for energy production. Carbon sequestration is often dealt with using this type of full accounting and results show that increases in off-site C can be sizable, perhaps matching increases in on-site C (e.g. Pacala *et al*., 2001). If N deposition accelerates forest growth, the potential for the off-site C sequestration (storage in products or in landfills, bio-energy offsets of fossil fuel emissions) is thus increased. In performing the overall calculation, we assumed that above ground biomass C sequestration in response to N deposition is:

- 15-30 kg C/kg N deposition for forests (see Table 5.4) of which 33% is sequestered on-site and 17% off-site, leading to an assumed sequestration of 7.5-15 kg C/kg N, being equal to 27- 55 kg CO_2-eq/ kg N (multiplied by 44/12).
- negligible for unforested ecosystems, assuming that the sequestered C is released again within a short time period.

For both forested and unforested ecosystems, we assumed that below ground soil C sequestration in response to N deposition is 20-30 kg C/kg N deposition, as shown in Table 5.4 and Table 5.5. This is equal to 73- 110 kg CO_2-eq/kg N.

Applying the country average NH_3 emission fraction of 14.2% and the above mentioned C sequestration rates per kg N deposition gives: $CO_{2,seq}/N_{use}$ = 0.1 x 0.142 x (27-55 + 73-110 kg) + 0.1 x 0.142 x (73-110) = 2.45-3.91 kg CO2-eq/kg N use or approximately 2.5-4.0 kg CO2-eq/kg N.

4 N_2O exchange from agricultural, terrestrial and aquatic systems.

Agriculture

The increase in GWP due to N_2O-N emission from agriculture can be calculated as:

$CO_{2,seq}/N_{use} = frN_2O\text{-}N_{\text{ emission, agriculture}} \times 44/28 \times 296$

Where 44/28 is the ratio relation N_2O to N_2O-N and 296 is the GWP of N_2O. Using the IPCC default value of 1% implies that a change of 1 kg N by animal manure or fertilizer implies an N_2O emission in agriculture of 0.01 kg N_2O-N per kg N. The values however are higher for organic soils and for urine and also the application of injection leads to higher emission values. Assuming an overall range of 1-1.5% leads to the following impact on GWP: 0.01-0.015 x 44/28 x 296 = 4.65-6.98 kg CO_2 eq/kg N use by either animal manure or fertilizer or approximately 4.5-7.0 kg CO_2 eq/kg N

Terrestrial systems
The related N_2O-N emission from terrestrial ecosystems, assuming a similar response of both forested and unforested systems can be calculated as:

$CO_{2,seq}/N_{use} = A_{\text{terrestrial ecosystems}}/A_{\text{total}} \times frNH_{3\text{ emission, agriculture}} \times frN_2O_{\text{ emission, terrestrial ecosystems}} \times 44/28 \times 296$

Assuming an average N_2O emission fraction form terrestrial ecosystems of 1.8% (based on forests) and an uncertainty range of 50% leads to an emission 0.009-0.027 kg N_2O-N/kg N deposition, which implies the following GWP impact, when applying the NL average emission fraction: 0.2 x 0.142 x 0.009-0.027 x 44/28 x 296 = 0.12-0.36 kg CO_2-eq/kg N or 0.1-0.4 kg CO_2-eq/kg N.

Aquatic systems
The related N_2O-N emission from terrestrial ecosystems, assuming a similar response of both forested and unforested systems can be calculated as:

$CO_{2,seq}/N_{use} = frN_{\text{leaching, agriculture}} \times frN_2O_{\text{ emission, aquatic ecosystems}} \times 44/28 \times 296$

In Dutch agriculture, the overall average nitrate leaching is approximately 15% being twice as low as the standard IPCC value of 30% ($Frac_L = 0.30$). In the previous IPCC estimates it was assumed that 2.5 % of the N leached is re-emitted according to: $EF_5 = 0.025$ (=EF_5-groundwater (0.015) + EF_5-rivers (0.0075) + EF_5-estuaries (0.0025). An EF_5 of 0.025 is however, presently considered as an overestimation, as discussed in Chapter 6 (Well *et al.*, 2005). Therefore, in the IPCC 2006 guideline the EF_5 is set to 0.015. Assuming a range of 0.01-0.02 implies the following GWP effect for agricultural N leaching:

0.15 x 0.01-0.02 x 44/28 x 296 = 0.70-1.40 kg CO_2-eq/kg N.

The indirect effect due to N leaching from terrestrial ecosystems is very low. Assuming the same N leaching fraction of 15% from these systems leads to: 0.2 x 0.142 x 0.15 x 0.01-0.02 x 44/28 x 296 = 0.02-0.04 kg CO_2-eq/kg N, which can be neglected.

The overall impact of N use in agriculture on CO_2 exchange is given in Table 5.9. Results show that the CO_2 sequestration in agricultural and terrestrial ecosystems (5.5-10 kg CO_2-eq/kg N) is slightly lower than the overall release of CO_2 due to N fertilizer production and N_2O release (6.3-19.2 kg CO_2-eq/kg N). It is specifically the uncertainty in the release of CO_2 due to N fertilizer production that determines whether there is a positive or a negative impact of N use on the GWP. When this is near the lower end of the range, both positive and negative effects of N use on the CO_2 balance are comparable and the net effect on the GWP would be negligible.

Table 5.9 Estimated NH_3 emission fractions for animal manure, fertilizer and organic products and a country averaged fraction for the Netherlands assuming low emission housing

Aspect	**System**	**CO_2 exchange kg CO_2-eq/kg N**
N fertilizer production	Housing	1-10 (release)
C sequestration	Agricultural	-3 -6 (sequestration)
	Terrestrial	-2.5-4 (sequestration)
N_2O	Agricultural	4.5-7
	Terrestrial	0.1-0.4
	Aquatic	0.7-1.4

5.3.2 Conclusions

The following conclusions (main messages) can be drawn from this study:

- Nitrogen deposition has overall a positive effect on the global warming potential. The net effect of carbon sequestration by terrestrial ecosystems, induced by an increased growth and a reduced decomposition largely offsets the increased N_2O emission induced by elevated N deposition. The impact of N deposition on the soil CH_4 sink is generally negligible.
- On average the above ground carbon sequestration in forests is 15-30 kg C/kg N deposition, of which 33% is sequestered on-site and half of that amount off-site, leading to an assumed sequestration of 7.5-15 kg C/kg N, being equal to 27- 55 kg CO_2-eq/kg N . For other terrestrial ecosystems, the effect of N deposition on C sequestration can be neglected since the sequestered C is released again within a short time period.
- For both forested and unforested ecosystems, a reasonable range in below ground soil C sequestration in response to N deposition seems 20-30 kg C/kg N deposition N deposition, being equal to 73- 110 kg CO_2-eq/kg N.
- Assessing the efficiency of existing or future measures related to reduced N use requires a full quantification of its effects on the exchange CO_2, N_2O and CH_4 by complete life cycle analyses, including: (i) release of CO_2 and N_2O related to the production of N fertilizer, (ii) C sequestration in agricultural soils in response to a change in N input, (iii) C sequestration in terrestrial ecosystems (forests) in response to a change in N (NH_3) input induced by the change in N use, including on-site and off-site sequestration and (iv) N_2O exchange from agricultural soils, non agricultural (forest) soils and re-emission from aquatic systems.
- Quantification of all above pathways shows that the overall impact of 1 kg N used in agriculture on the global warming potential (in kg CO_2-eq) may be negligible when the N is applied in the form of animal manure. When it is applied in the form of N fertilizer, it is likely to have a negative impact on the GWP due to the CO_2 and N_2O emissions during the production of fertilizer, which are, however, highly uncertain.

6 Impact of N-inputs on GHG emissions in aquatic ecosystems

6.1 Introduction

Aquatic systems may play an important role within the context of exchange of greenhouse gasses (IPCC, TAR en SAR). However, their role has not been investigated as thoroughly as compared to terrestrial systems (see e.g Chapter 5). An overview of the different drivers and interactions considered in this chapter is shown in Figure 6.1.

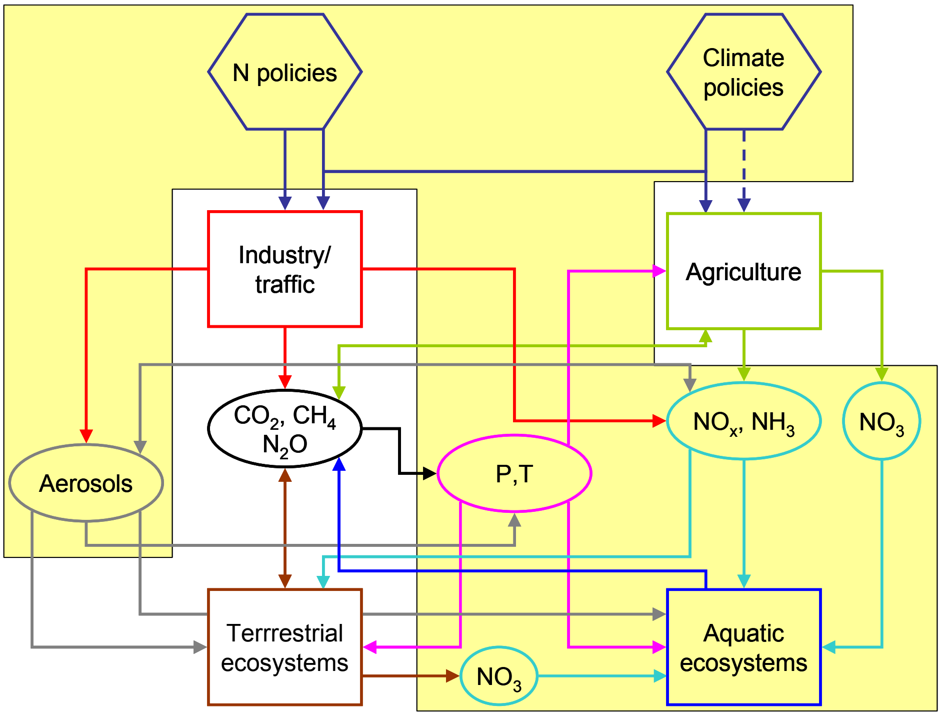


Figure 6.1 Overview of drivers and interactions considered for assessment of the impact of nitrogen inputs on GHG emissions in aquatic ecosystems.

Whether an aquatic system is a source or sink depends on both the type of greenhouse gas, CO_2, CH_4 or N_2O, and the type of system a the fresh water lake (regional system), reservoir, river, estuary, sea/ocean (Figure 6.2). Many undisturbed natural fresh water systems are a source of CO_2, CH_4 and N_2O (Seitzinger & Kroeze, 1998; Huttunen *et al.*, 2003). However, due to eutrophication (by N and P) lakes may act as a sink for CO_2 due to the accumulation of biomass and dead organic material (comparable to terrestrial systems, as described in Chapter 5). Alternatively, nutrient enriched lakes may increase the N_2O and CH_4 emission. Reservoirs are a net source of CO_2 and CH_4 (St. Louis *et al.*, 2000). Estuaries emit large amounts of CH_4 and CO_2 (Middelburg *et al.*, 2002), whereas oceans sequester large amounts of CO_2 (Takahashi *et al.*, 2002).

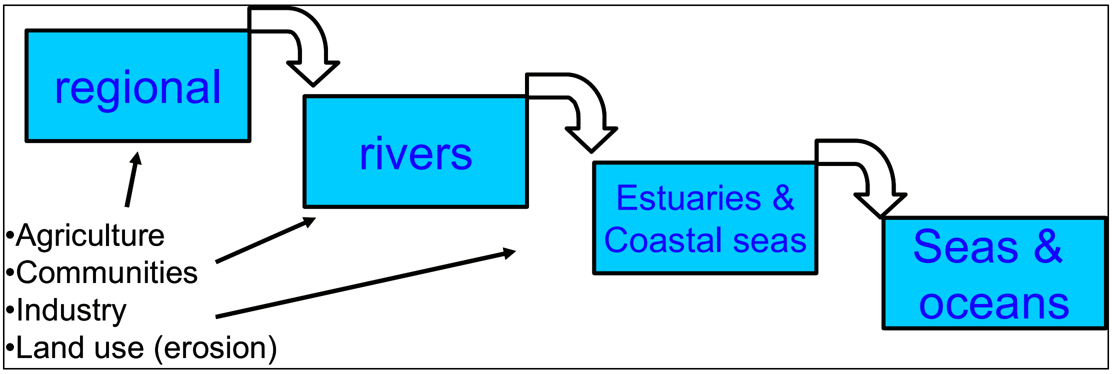


Figure 6.2 Hierarchy of aquatic systems and their relation between N (and C load) and GHG emissions.

The purpose of this chapter is to assess the relation between climate policies and N-related policies and to determine the effect of reactive nitrogen abatement on GHG emissions and sinks in aquatic systems in the Netherlands and Europe.

More particular it will address the opportunities for a positive linkage and the risks for a negative linkage between climate policy and water policy. Furthermore, the natural contribution and trends in GHG emissions from aquatic systems will be discussed.

Water policy in the Netherlands

The aim of the Dutch Water policy is *"keeping the country habitable by maintaining resistant and resilient (healthy) water systems"* (V&W, 1998; RIVM, 2004). Water policy in the Netherlands is primarily focused on safety by measures preventing through e.g. improving the of quality and increasing height of dykes. Ecological *health* of water systems is regulated by policies aimed both at reducing emissions and at improving water quality, however management of water quantity and physical aspects also is very relevant. Eutrophication is the major ecological stress factor. Presently, levels of both phosphate and nitrogen exceed critical values for maintaining target ecosystems. It is generally accepted that phosphate is most limiting for biological production (phytoplankton) in freshwater ecosystems, while nitrogen is the limiting nutrient in marine and estuarine ecosystems. Although N- and P-emissions, both from point and diffuse sources, have decreased 30-90% in the past 15 years, water quality does not meat critical values. Maintaining aquatic ecosystems and target species, therefore is not possible or requires ecosystem management and manipulation.

With respect to freshwater, nutrient policies were based on MTRs (comparable with: Maximum Admissable Concentration and which is defined as 'a target value for a substance indicating the concentration where both ecosystems and humans will not show negative effects') as laid down in NW4 (1998; 2,2 mg.l^{-1} N) and for marine ecosystems on emission reduction targets (OSPAR, 2001; 50% reduction as compared to 1985). At present national water quality policies are under revision in order to implement the EU Water Framework Directive (WFD). The WFD requires EU member states to define and implement measures to assure a Good Ecological Status in all waters in 2015. In contrast to NW4, the WFD is more mandatory with respect to achieving the ecological results, but less stringent with respect to achievement of water quality or emission reduction targets. The implementation of the WFD is presently undertaken; and it is expected that it will lead to external integration of various water policies and to regional differentiation of nitrogen targets and measures. Eutrophication measures may have trade-offs to GHG-emission: Some examples are:

- Increasing nitrogen removal efficiencies of waste water treatment plants, will increase the nitrous oxide emission at the plant level. Net long-term national emission levels are not necessarily higher
- Installing wetlands, riparian buffer zones, flow fields (helophytic filters) in catchments to reduce diffuse N sources will also likely increase nitrous oxide emission, but again the net effects on nitrous oxide emission when taking into account the full nitrogen cascade are less certain. This will depend on both the overall ratio of N-sequestration to denitrification and the ratio of N_2O/N_2 in the gaseous N-loss.

- It is still not clear if the WFD will lead to additional need for reduction of diffuse nitrogen losses from agriculture, as compared to demands by the Nitrate Directive (ND). If it will, then the likely effect will be reduction of total N-use in Dutch agriculture. Such a reduction could be achieved either by reducing the volume of the agricultural sector or increasing N-use efficiencies. Both would almost certainly reduce nitrous oxide emission from agriculture. However, if the disposed production volume would be transferred to a location outside the Netherlands, the net GHG-effect becomes uncertain again. For international Climate policies it is irrelevant where GHG emission reduction takes place.

Also policies with respect to water quantity management may have climate trade-offs.
Policy targets with respect to lower risks of flooding are laid down in the national 'Bestuursakkoord Water' (V&W, 2003). They include provision for inundation of land with low economical value. However, flooding may lead to additional emission of methane and nitrous oxide. In view of expected increased precipitation and precipitation peaks (Table 3.3) due to climate change, additional measures are needed to meet the present protection standards against light flooding. These standards vary: see Table 6.1.

Table 6.1 NBW-standards for protection against light (taken from Kragt et al., 2007: and based on V&W 2003, STOWA 2001).

Land use	standard (1/jr)	Flooded area[1)] (%)
Grassland	1/10	5
Arable	1/25	1
Horticulture	1/50	1
Greenhouse farming	1/50	1
Urban	1/100	0

1) indicates the area criterion: above this areal fraction there is light flooding

Kragt *et al*. (2006) estimate the total area that needs to be reserved for water storage to prevent light flooding at approximately 35,000 ha. This area will be mainly in use for pasture. If this area will be regularly flooded, denitrification of the amply available nitrogen in the soil may be enhanced. It is likely that during flooding emission of nitrous oxide will be enhanced.

In the Netherlands there is no policy aimed at reducing GHG emission from aquatic systems or sinking GHG in aquatic systems.

Impacts of climate change on Dutch aquatic systems

Some expected effects of climate change on aquatic ecosystems are (MNP, 2005b)

- North sea temperature increased >1°C
- Less phytoplankton
- Uncoupling with peak of zooplankton
- Fewer shellfish in Wadden Sea
- Increase of biodiversity in lakes and streams, but countervailing powers
- Increased risk of algal blooms and botulism
- Increased risk of anoxia
- Increased risk of drying up

These changes will lead to changes in both the volume and timing of flows of carbon and nitrogen, which in combination with changes of temperature and residence time affect GHG-fluxes. However, the combined effect of these changes on GHG-emission can not be determined and is beyond the scope of this assessment. An extensive review on the sensitivity of marine systems on climate change is given by Smith *et al*. (1999). Effects are very divers and ranging from effects on biological processes and biodiversity which may negatively affect the CO_2 sequestration to increased flood risk which may cause loss of life.

6.2 Specific overview of linkages

The aquatic system is closely linked with the terrestrial system (see Figure 6.3).

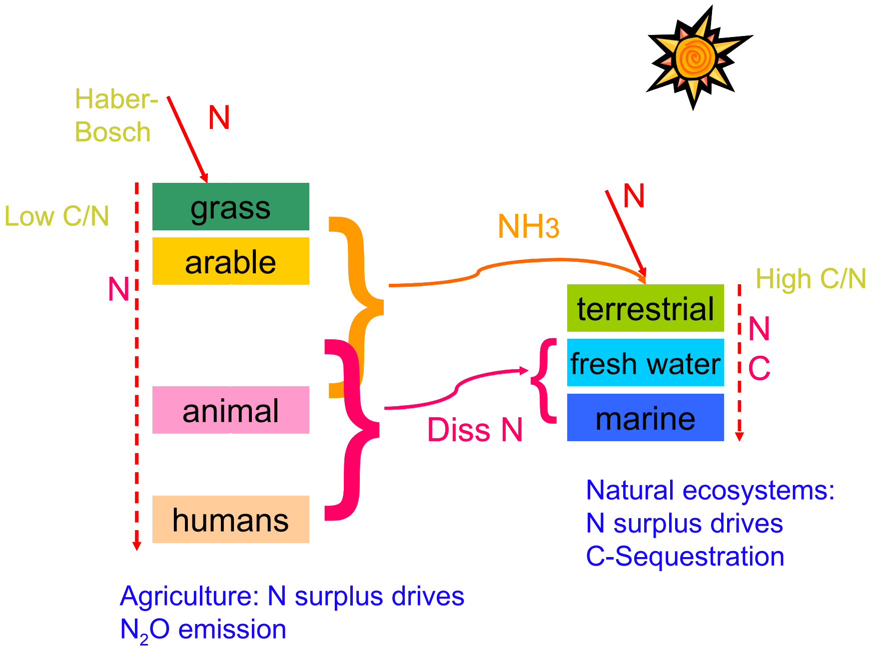


Figure 6.3 Overview linkage C and N in terrestrial and aquatic systems.

Important drivers for the exchange of GHG from aquatic systems are the residence time and the load of both nitrogen and carbon.

At a global level N_2O emissions from riverine and marine systems is highly related to N loading (Seitzinger & Kroeze, 1998), whereas at the level of catchments or individual lakes environmental factors such as oxygen contents and pH are important factor (e.g. De Bie *et al.*, 2002).

More than 50% (290 + 100 Tg.yr^{-1}) of the annual global biogenic CH_4 flux is released from aquatic ecosystems (Tyler, 1991). Natural wetlands, e.g., bogs, fens, flood plains, littoral zone of lakes, marshes and swamps are likely the largest natural sources of CH_4, accounting for about 20% of the current global annual emission 160 + 40 Tg.yr^{-1} (Wuethrich, 1994). Generally, factors governing CH_4 formation and emission from wetlands include substrate supply, absence of oxygen, temperature, pH, salinity and sulphate, vegetation presence and productivity, CH_4 oxidation rate, hydrological conditions and physico-chemical properties. Methane emissions are correlated with C inputs and cycling in cypress swamps (Harriss & Sebacher, 1981). Elevated CO_2 levels enhance CH_4 emissions from wetland ecosystems, including freshwater marshes (Megonigal & Schlesinger, 1997).

There are indications that nitrogen inputs to wetlands and aquatic ecosystems may decrease the emissions of CO_2 and CH_4 (Aerts & Toet, 1997). They conclude that increased amounts of NH^+_4-N supply lead to reduction of decay of organic matter in peat soils and thereby to a reduction of gaseous carbon loss from these soils. Nutrient or glucose additions lead only to a short-term increase in methane emissions from peat soils. N supply may decrease the oxidation capacity of soils for atmospheric CH_4, thereby decreasing the net influx of CH_4 from atmosphere to biosphere (Steudler *et al.*, 1989; Sitaula *et al.*, 1995; Van den Pol-van Dasselaar *et al.*, 1999), but inverse effects have also been found (Bodelier & Laanbroek, 2004). Evidently, the net effect of N load on the net greenhouse gas budget of natural ecosystems is the result of complex interactions and ecosystem feed backs, and highly dependent on local environmental conditions.

Residence time

Residence times strongly determine the N and C processes in aquatic systems. Longer residence times generally yields an increase in nitrogen removal through nitrification and denitrification and immobilization and/or retention. As within terrestrial systems nitrification and denitrification, occurring in sediments and water columns, are an import source of N_2O (e.g. De Bie *et al*., 2002).

Estuaries with a long residence time show higher methane emission compared to those with shorter residence times (Middelburg *et al*., 2002). The net loss of methane from the aquatic ecosystem is influenced by the solubility of methane and subsequent consumption of methane by oxidation in the water column. Van der Nat *et al.* (1997) reported methane oxidation rates for the surface layer of intertidal sediments from the upper Scheldt estuary corresponding to a turnover rate of about 2 hour. Methane oxidation is an important sink of methane in estuaries, that strongly depends on the temperature (increase) and salinity (decrease) (see Middelburg *et al.*, 2002). Fresh water lakes, however, not always show an increase in CH_4 emission at longer residence time. Striegl and Michmerhuizen (1998) showed that in a lake with a longer residence the slightly higher CH_4 production through methanogenesis is largely compensated by the increase in methane oxidation.

C sequestration in fresh water lakes is also strongly influenced by the residence time. Generally an increase in residence time yields an increase in biomass accumulation and by that a decrease in CO_2 release (Striegl & Michmerhuizen, 1998). Consequently, generally an increase in residence time will enhance N_2O and CH_4 and decrease CO_2 emission from aquatic systems. However, when focusing on a (large) catchment as a whole the outcome might be different. E.g. an increase in up stream residence time might result in opposite effects downstream because of higher N (and perhaps C) losses upstreams.

Table 6.2 gives an overview of the different effects of residence time on the release of GHG from various aquatic systems.

Table 6.2 Effect of residence time on the release of N_2O, CH_4 and CO_2 from various aquatic systems.

System	Effect	N_2O	CH_4	CO2
Regional/rivers	Denitrification > Methonagesis > CH_4 oxidation > Biomass >	++[1]	+/-	++
Estuaries+coastal seas	Methonagesis > Biomass > Degassing >	+	++	++
Oceans	Not relevant	Not relevant	Not relevant	Not relevant

[1]. ++ = a strong reducing effect; + = a reducing effect; +/- = an uncertain effect; - = an increasing effect; -- = a strong increasing effect

N and C load

Levels of river borne nitrogen have increased dramatically in the seventies and eighties, as has transport from rivers to marine systems in Europe (NMP, 2001) as well as in Northern America (Turner & Rabalais, 1994) but European loads have stabilised or decreased in the past decade (OSPAR 2006). Nitrogen loading plays an important role in the global climate system by regulating the balance of greenhouse gases produced and consumed in freshwater and estuarine sediments (Seitzinger & Kroeze, 1998). Of particular concern is the N_2O emission, which is currently increasing globally at the rate of 0.2-0.3% per year (IPCC, 1996). In addition nitrogen loading may also be relevant for local and regional effects such as oxygen depletion, toxic and nuisance algae blooms, sedimentation, and biodiversity loss (OSPAR 2006; NMP, 2001).

For Europe most N inputs to aquatic systems originate from agricultural sources (Van Drecht *et al*., 2003): 50% from agriculture, 27% from natural ecosystems and 23% form point sources. Eventually 40% of the antrogenically fixed N will enter the oceans (Seitzinger *et al*., 2005). At a global scale, most N export to coastal zones is discharged in its inorganic form (DIN, 80% of the total N). 97% originates from diffuse sources and 3% from point sources. Diffuse sources originate from N_2 fixation (23%), N deposition (13%), animal manure (28%) and fertilizers (33%) (Seitzinger *et al.*, 2005).

In general more N input yields more C sequestration. However, at higher N inputs P and Si may become the limiting factor. Under such conditions more N does not results in additional C sequestration.

Striegl & Michmerhuizen (1998) showed that the CO_2 and CH_4 exchange from two North-Central Minnesota lakes that differ in C load highly depends on the C inputs, both in the form of dissolved inorganic and organic C. From an inventory of Finnish lakes it was found that CO_2 emission increases with higher trophic state and higher proportion of agricultural land in the catchment (Kortelainen *et al*., 2006).

Lateral carbon fluxes induced by sequestration in the terrestrial system and river transport are significant contributors to the regional carbon budget of the European continent (Ciais *et al.*, 2006). Rivers transport carbon in dissolved and particulate organic forms (about 50%) and inorganic forms (about 50%). Organic C will be buried in the sediment and may be a source for denitrification and methanogenesis; whereas inorganic C will result in CO_2 degassing to the atmosphere as transported water enters coastal seas (see Figure 6.4).

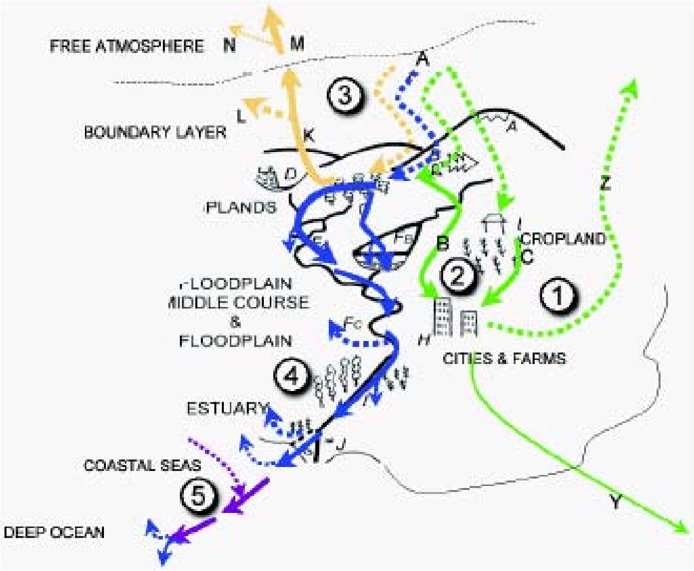


Figure 6.4 Interactions between carbon cycling in terrestrial and aquatic system (reproduced from Ciais et al, 2006).

Figure 6.4 shows the carbon cycle loop involving lateral transport. The associated sources/sinks of atmospheric CO_2 are represented by dotted lines, and horizontal fluxes of carbon in solid lines. In Green: Loop 1 shows the cycle associated with photosynthesis (A), harvest of wood and crop products, transport by domestic (B, C) and international (Y) trade circuits, and human consumption (Z) related release of CO_2. Loop 2 shows the same, but now for forest products (2). Light brown; Loop 3 shows the cycle associated with photosynthesis (A), RCC emissions and atmospheric transport in the boundary layer (K) and in the free atmosphere (M), with oxidation to CO_2 in the boundary layer (L) or in the free atmosphere (N). Blue: Loop 4 shows the cycle of carbon of atmospheric origin (A) transported in the form of DIC, DOC, POC by river systems from upland rivers to inner estuaries, includes outgassing of CO_2 by freshwaters (F_c) and carbon storage in reservoirs (F_b). Purple: Loop 5 shows the fluxes of carbon and CO_2 source/sink from coastal seas.

According to Ciais *et al.*, (2006) the amount of CO_2 released from the aquatic system amounts to 10% of the annual fossil fuel CO_2 emission. Therefore, these component fluxes must be accounted for to accurately assess the European carbon balance, in particular to translate inversion based CO_2 fluxes into carbon budgets.

The effect of N on CH_4 production or oxidation is, however, very uncertain:
- CH_4 oxidation inhibited by input of NH_4, resulting in an increase of CH_4 emission (see e.g. Liikanen and Martikainen (2003))
- In case of N limitation of metamorphic bacteria, more NO_3 results in enhanced CH_4 oxidation, which in turn decreased the CH_4 emission
- On the other hand NO_3 inhibits methanogenesis, resulting in a decrease in CH_4 emission

In addition more C input and increase in anoxic conditions result in an increase in methanogenesis, resulting in an increase in CH_4 emission

These effects occur at a microbiological level. For the overall effect it is important to assess the these effects at ecosystem level. However, knowledge on these overall effects at an ecosystem level too limited for the derivation of fluxes. Qualitative Estimates of effects of N inputs and C inputs on the GHG emission from aquatic systems are summarized in Table 6.3 and Table 6.4, respectively.

Table 6.3 Effect of N on the release of N_2O, CH_4 and CO_2 from various aquatic systems.

System	Effect	N_2O	CH_4	CO_2
Regional/rivers	Denitrification > CH_4 oxidation ? Biomass >	++ 1)	+/-	--
Estuaries+coastal seas	Methanogenesis > Biomass > Degassing >	++	++	+ (?)
Oceans	Denitrification >	+	?	?

1). ++ = a strong reducing effect; + = a reducing effect; +/- = either increasing or decreasing effect; - = an increasing effect; -- = a strong increasing effect; ? = uncertain effect

Table 6.4 Effect of C on the release of N_2O, CH_4 and CO_2 from various aquatic systems.

System	Effect	N_2O	CH_4	CO_2
Regional/rivers	Denitrification > CH_4 oxidation ? Biomass >	++ 1)	+	+/?
Estuaries+coastal seas	Methanogenesis > Biomass > Degassing >	+	++	++
Oceans	Denitrification >	+	+	+

1). ++ = a strong reducing effect; + = a reducing effect; +/- = an uncertain effect; - = an increasing effect; -- = a strong increasing effect; ? = uncertain effect

6.3 Quantification of effects on GHG emissions

Each type of aquatic system behaves differently with respect to GHG.

N_2O emission from aquatic systems

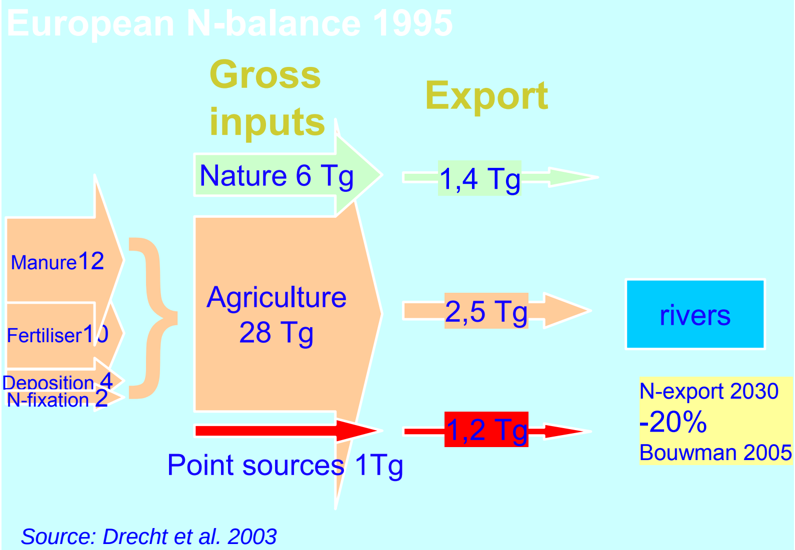
The IPCC method for the quantification of N_2O emission from aquatic systems (indirect emission) is as follows:

N_2O indirect = N_2Oem (Ndep) + N_2O (NO_3 input)
N_2Oem (Ndep) = $EF_4 \times$ Ndep
N_2O (NO_3 input) = $EF_5 \times$ FracL $\times$ NO_3input

IPCC factor (1999):
$EF_4 = 0.01$
$Frac_L = 0.30$
$EF_5 = 0.025$ (=EF_5-groundwater (0.015) + EF_5-rivers (0.0075) + EF_5-estuaries (0.0025)

An EF_5 of 0.025 is considered as an overestimation (Well *et al*., 2005). Therefore, in the IPCC 2006 guideline the EF_5 is set to 0.015.

For Europe, Kroeze *et al*. (1999) calculated the N_2O emissions from aquatic systems due to N inputs for the year 1995. They estimated a total inorganic N (DIN) input of 2,5 Tg N (see also Figure 6.5) and an N_2O-N emission of 140 Gg, which is 6% of the DIN-export, divided in 5% from rivers and 1% from marine systems. These results were checked against atmospheric N_2O records from 1700 to 1995. The anthropogenic part amounts 20% (28 Gg N_2O-N).



Source: Drecht et al. 2003

Figure 6.5 The European N-balance for 1995 (Source: Drecht et al., 2003).

CO_2 emission from aquatic systems

Fresh water systems play a role in C sequestration and CO_2 and CH_4 release (see IPCC, 2001). As a result of primary production in the regional system (lakes and adjacent riparian areas) 319 Mt $CO_2.yr^{-1}$ is accumulated in the sediments of fresh water bodies (Stallard, 1998).

Lakes and rivers act as a source of CO_2: as the water column becomes saturated, respirated CO_2 will be emitted to the atmosphere. Especially during spring and autumn CO_2 is released mainly from small boreal lakes. For all lakes of the boreal region Kortelainen *et al*. (2006) estimated a total annual CO_2 evasion of about 50 Tg C, a value up to 40% of the current global estimate for all lakes.

The role of *estuaries* within the global carbon system has been considered unimportant. However, for European estuaries Ciais *et al*. (2006) showed that they are a notable source of CO_2. European estuaries emit a large part of the CO_2 fixed in terrestrial systems, which ranges from 5 to 10% of the total anthropogenic CO_2 emission. Comparable results were found for South America (Mayorga *et al*., 2005). Mayorga *et al*. (2005) showed that CO_2 emission in the Amazon estuary mainly originated from organic material < 5 years old.

For *oceans* the annual CO_2 accumulation by oceans amounts to 101 $Gt.yr^{-1}$ (IPPC, 2007, 4AR)

Knowledge about regional C cycling is rather fragmentized. However, the overall effect of enhanced N may lead to more N and C sequestration and to higher emissions of both N_2O and CH_4.

CH_4 emission from aquatic systems

Methane release from fens is also significant and is positively correlated with the CO_2 release (see e.g. Alm, 1997). Houweling *et al*. (1999) estimated a flux of 15 $Mt.yr^{-1}$ which is 6% of the total natural sources and 3% of the total CH_4 global emissions (anthropogenic + natural).

Dissolved methane concentrations in rivers are one to two orders of magnitude higher than those in open ocean waters and riverine methane can be traced over long distances (Scranton & McShane, 1991; Jones & Amador, 1993). The distribution of methane in some estuaries is largely governed by riverine inputs and conservative mixing (e.g. De Angelis & Lilley, 1987), while non conservative behaviour due to methane outgassing, oxidation and estuarine sources has been identified in some other estuaries (e.g. Sansone *et al.*, 1999).

Middelburg *et al*. (2002) estimate that estuaries emit between 1.8 and 3.0 Tg $CH_4.yr^{-1}$, which in turn is a very small component of the global methane emission, i.e. 5 to 50 Tg or between 1 and 10% of total methane emission (see e.g. Cicerone & Oremland, 1988).

The role of aquatic systems in the emission and oxidation of anthropogenic CH_4 is rather small. Most anthropogenic CH_4 is produced by waste water production and treatment and enteric fermentation in ruminants.

6.3.1 Interaction N and climate change

The interaction between climate change and eutrophication is complex and predictions of effects are not always pointing into the same direction (see eg. IPCC TAR and 4AR).

However, there are some opportunities for synergism between nitrogen and climate policies:

- Generally less anoxia and by that less GHG emission
- Shorter upstream residence times (during spring and autumn) yield higher discharge of the upstream active microbiological population, resulting in less anoxia and less GHG emission. However, the upstream gain might be compensated downstream.

There is also a risk for antagonism:

- Increase in water storage (wetlands, inundation zones), which in turn may cause an increase in N_2O and CH_4 emission

6.3.2 Effect of climate change on N and GHG emissions from aquatic systems

When considering the effects of climate change on the GHG emissions, prognosed temperature increase and shorter up-stream residence times (due to higher discharges) are the effects that can be mentioned. The extend of the effects on the GHG emissions is not always clear, but in Table 6.5 we make an effort to list these effects in a qualitative way.

Table 6.5 Effects climate change on the emission of N_2O, CH_4 and CO_2 in various aquatic systems.

Effect	Effect	N_2O	CH_4	CO_2
Temperature increase	Decomposition > Degassing >	++	+/-	++
Higher discharge	Not relevant	Not relevant	Not relevant	Not relevant
Shorter up stream residence time	Methonagesis > Biomass > Degassing >	+	++	++

6.4 Effect of existing and possible future policies/measures

Measures related to N emissions

Measures focusing on the reduction of the nitrogen input to aquatic systems, will start with changes in fertilizer use, nitrogen deposition, animal numbers (and thus manure) and waste water treatment efficiencies. These measures will have an effect on the inorganic and/or organic nitrogen input to the aquatic systems. Table 6.6 gives a qualitative overview of the effects of the different N mitigation measures on GHG emissions.

Table 6.6 Quantification of reduction in N load on N_2O, CH_4 and CO_2 from various aquatic systems.

Driver	Effect	N_2O	CH_4	CO_2
Reduction fertilizer use	Less Nin	++ [1)]	?	-
Reduction N deposition	Less Nin	++	?	-
Reduction animal numbers (manure)	Less Nin and Norg	++	?	?
Improvement wastewater treatment	Less Nin and Norg	++	+	?

[1)]. ++ = a strong reducing effect; + = a reducing effect; +/- = an uncertain effect; - = an increasing effect; -- = a strong increasing effect; ? = uncertain effect

Measures related to NH_3 and NO_x emissions

Dimethyl sulfide particles emitted from marine algal blooms are believed to play a relevant role in controlling climate change (Gabric, *et al*, 2003). The DPSIR mechanism is: less emission of N to the marine environment will decrease algal growth. Specific species of marine algae produce Dimethyl Sulphinoniopropionate (DMSP) which is converted to Dimethyl sulphide. Dimethyl sulphide is a precursor of marine aerosols, which are believed to play an important role in cloud condensation nuclei. Impacts on atmospheric chemistry and climate are believed to be substantial (Kiene: http://faculty.disl.org/rkiene.html). GMC simulations predicted an decrease of artic ice cover by 20% due to reduction of DMS emission and subsequent cloud formation (Gabric *et al*., 2003). This case is an example of a potential antagonism between N-reduction and GHG-reduction policies, as N-availability is one key factor stimulating DMS-producing algae.

6.5 Conclusions

Both fresh waters and coastal zones are a net source of GHG. The total anthropogenic contribution to GHG emission through surface waters is relatively small. Most significant is the N_2O emission resulting from discharge of waste water and from agricultural N losses to surface water.

Effects of climate change on GHG emission from surface waters might be large. The net effect is, however, uncertain. Higher discharges in spring and autumn yield a decrease in GHG, whereas anthropogenic water management measures to regulate discharge and to prevent (summer) drought may increase GHG emissions.

Effects of N measures to reduce N loads yield a decrease in emissions of N_2O and possibly also CH_4, but possibly a decrease in CO_2 sequestration.

The overall total effect of N on GWP induced by aquatic systems is highly uncertain. Contrary to terrestrial systems no quantitative information is available on the net C sequestration of the aquatic system, not for the actual situation as well as for the past and the future.

7 Conclusions of the assessment

In the different sections of this assessment several conclusions were drawn. These conclusions are listed again, according to the five separate subsystems.

Emission of NH_3, N_2O, CO_2 and fine particles in energy production

- When looking at the energy sector as a whole, the way in which energy is produced now and in the future is largely determined by fuel price and availability of fossil fuels. The amount of energy used depends on developments in consumer behaviour, which can also be influenced by changes in climate (higher temperature, less heating - more air conditioning);
- Synergies between GHG and N emission reduction are possible in the energy production sector where NO_x and CO_2 can be reduced simultaneously e.g. by sustainable sources and energy saving. In some cases also reduction of SO_2 and PM is possible at the same time;
- It should be noted however that antagonisms between measures focussing at specific compounds (i.e. GHG and N) are also possible. This is especially true for the use of biofuels and bio-energy. The net result is not always clear and should be investigated in more detail.
- More in general, GHG emission reductions are possible when introducing N (NO_x, NH_3) reduction measures. However, this is not exclusively valid for the energy sector, but holds for different other economic sectors, such as transport and agriculture; Pollutant swapping, however, needs to be investigated, e.g. when ammonia is reduced by manure injection without reducing the N overload, leading to increased N_2O emission.
- When only looking at the energy sector:
    - measures that are defined mainly focus on CO_2 reduction (with possible simultaneous reduction of NO_x and SO_2);
    - there are only a few options defined for reduction of NO_x and SO_2 in this sector
    - quantification of the coupled CO_2/NO_x/SO_2/PM emission reduction is possible, but requires further investigation
- Aerosols can partly compensate global warming, due to their cooling effect. When reducing the emission of NOx and/or NH_3 it will have an effect on the overall cooling effect. However, the regional impact of such emission reductions on the cooling effect is still poorly understood. It is advisable to quantify the side-effects of emission reductions and adopt the degree of implementation of policies/measures to the implementation to reduce GHG emissions.

Impact of N-fertilization on GHG emissions in agricultural systems

- Agriculture has substantial but partly evitable net GHG effects both through the arable sector (fertilizer), and the animal sector (enteric fermentation and manure storage).
- NEC ammonia measures enhance N_2O emission through mandatory manure incorporation, but we lack the knowledge to weigh the ammonia-ecology benefits against the GHG-effect of additional nitrous oxide emission.
- The nitrate directive has lead to 30% reduction of total N inputs to Dutch agricultural soils and has direct co-benefits for climate via reduction of N_2O emission. Further reduction to increase nitrogen efficiencies is possible.
- Including GHG-effects in damage effects and using the approach of marginal financial gain including externalities like damage by eutrophication and climate change should lead to substantially lower N-fertilizer recommendation for both food and energy crops.
- (Co-) digestion of manure can be a worthwhile measure to reduce methane and nitrous oxide emissions from manure storage, and to increase the fertilizing value of manure as compared to chemical fertiliser. Focusing too much on biogas and energy generation may lead to unsustainability if this would reduce replacement of chemical fertilizer by manure and reduce the organic matter supply to the soil.
- N_2O emission from agricultural soils is very complex which is major reason for frequent revision of emission factors.

- Effect of climate change (temperature and precipitation) on N and C-cycle in relation to GHG emission and N-losses for agro-ecosystems can be substantial but are hardly quantified for the Dutch situation and therefore need further attention.

Impact of biofuel use and land use changes on GHG emissions in the energy sector and agriculture

There is a broad range of LCA results for GHG mitigation for any given biofuel due to different input assumptions (corresponding to different actual practices) and methods, but some broad conclusions are possible:

- N_2O is a large contributor to the overall GHG balance for (food and) biofuels. N_2O is emitted at fertilizer production, application and further in the cascade (producing animal food, manure, losses to groundwater and atmosphere, etc.)
- The GHG emissions resulting from fertilizer production is dominated by N_2O. Furthermore, N_2O from fertilizer production is the largest N_2O source. Currently catalysts are available to reduce these emissions with more than 90% and are being implemented
- Nitrogen losses increase with increased fertilizer use, similar to the nitrogen-food cascade. Energy crops might use more fertilizer than food because they are produced on marginal grounds and production will be driven by higher ton biomass per ha with a low fertilizer price compared to a high (biofuel) yield and price
- Grain-based biofuels offer less GHG mitigation than lignocelluloses-based fuels due primarily to lower effective yields
- Among commercial biofuels today, sugarcane ethanol gives highest land use efficiency for GHG mitigation
- In longer term, land use efficiency for GHG mitigation is likely to be highest for lignocelluloses plantation biomass (FT or DME in 2010/2015 timeframe, ethanol in 2020/2030 time frame)

Biomass for biofuels vs. biomass for electricity

- Less GHG mitigation per hectare if biomass is used to make biofuels than if it is used to make electricity displacing coal power. (This is true with existing steam cycle biopower technology and more true with future bio-IGCC.)
- If bio-electricity is displacing NGCC electricity or electricity from any fossil-fuel combined heat and power, then biofuels (from sugarcane or from lignocelluloses crops) may give greater GHG mitigation per hectare.
- Cost of GHG mitigation (€/tC avoided) for stationary versus transport applications has not been examined, but likely would be lower for higher GHG mitigation options.

Risk and opportunities of climate policies and nitrogen policies

- Decentralised biofuel production leads to higher NO_x emissions; large scale production (de-NO_x SCR) and fuel use (catalytic converters) do not yield higher NO_x
- (Co-) digestion is a widely promoted option to produce biogas and/or heat and power. However, there is a competition between animal feed and energy/biofuels. Furthermore, it might be sustainable with respect to energy production, but when focussed on GHG reduction only, it is more effective to only use manure digestion without addition of mais.
- Increased biomass production yield higher N emissions in the existing cascade (similar to food production nitrogen cascade)
- 2nd generation biofuels is favourable in relation to energy/GHG balances. However, the competition with animal feed needs attention.
- NL depends on import to reach targets: opportunities to regulated imported ‘commodity’ fuels with N-certificates?

Overall: an increase of NUE and further development of 2nd generation biofuels are necessary

Impact of nitrogen deposition on greenhouse gas emissions in terrestrial ecosystems

The following conclusions (main messages) can be drawn from this study:

- Nitrogen deposition has overall a positive effect on the global warming potential. The net effect of carbon sequestration by terrestrial ecosystems, induced by an increased growth and a reduced decomposition largely offsets the increased N_2O emission induced by elevated N deposition. The impact of N deposition on the soil CH_4 sink is generally negligible.
- On average the above ground carbon sequestration in forests is 15-30 kg C/kg N deposition, of which 33% is sequestered on-site and half of that amount off-site, leading to an assumed sequestration of 7.5-15 kg C/kg N, being equal to 27- 55 kg CO_2-eq/kg N . For other terrestrial ecosystems, the effect of N deposition on C sequestration can be neglected since the sequestered C is released again within a short time period.
- For both forested and unforested ecosystems, a reasonable range in below ground soil C sequestration in response to N deposition seems 20-30 kg C/kg N deposition N deposition, being equal to 73- 110 kg CO_2-eq/kg N.
- Assessing the efficiency of existing or future measures related to reduced N use requires a full quantification of its effects on the exchange $CO_{2,}$ N_2O and CH_4 by complete life cycle analyses, including: (i) release of CO_2 and N_2O related to the production of N fertilizer, (ii) C sequestration in agricultural soils in response to a change in N input, (iii) C sequestration in terrestrial ecosystems (forests) in response to a change in N (NH_3) input induced by the change in N use, including on-site and off-site sequestration and (iv) N_2O exchange from agricultural soils, non agricultural (forest) soils and re-emission from aquatic systems.
- Quantification of all above pathways shows that the overall impact of 1 kg N used in agriculture on the global warming potential (in kg CO_2-eq) may be negligible when the N is applied in the form of animal manure. When it is applied in the form of N fertilizer, it is likely to have a negative impact on the GWP due to the CO_2 and N_2O emissions during the production of fertilizer, which are, however, highly uncertain.

Impact of N-inputs on GHG emissions in aquatic ecosystems

- Both fresh waters and coastal zones are a net source of GHG. The total anthropogenic contribution to GHG emission through surface waters is relatively small. Most significant is the N_2O emission resulting from discharge of waste water and from agricultural N losses to surface water.
- Effects of climate change on GHG emission from surface waters might be large. The net effect is, however, uncertain. Higher discharges in spring and autumn yield a decrease in GHG, whereas anthropogenic water management measures to regulate discharge and to prevent (summer) drought may increase GHG emissions.
- Effects of N measures to reduce N loads yield a decrease in emissions of N_2O and possibly also CH_4, but possibly a decrease in CO_2 sequestration.
- The overall net effect of aquatic processes on GWP is uncertain. Contrary to terrestrial systems no quantitative information is available on the net C sequestration of the aquatic system, not for the actual situation as well as for the past and the future.

General observations from this assessment

Besides the different conclusions for the different subsystems, some general policy relevant observations are presented here. They mainly focus on the most important interlinkages (both positive and negative) between nitrogen and climate policies, not yet firmly embedded in present policies. These observations are inspired by insights from the underlying studies and are important areas of interest for the future

Observation 1: Large scale production of biofuels

Use of biofuels or bio-energy requires more knowledge before it is applied on a large-scale because of (i) the uncertainty about their net CO_2 effect, (ii) the negative side-effects on the nitrogen cycle and (iii) the competition with the need of land for nature and/or food production. This is especially true in relation to the so-called first generation fuels. Negative side effects on the nitrogen cycle are additional emission of N_2O and NH_3, and increased leaching of nitrogen to aquatic systems due to an increased used of artificial fertilizer. There are also additional NO_x

emissions during the production and use of bio-fuels, like with decentralised combined heat & power installations. There is a clear need for integrated assessments addressing the all impacts related to the use of biofuels.

Observation 2: Enhanced carbon storage in natural systems through nitrogen fertilisation

Stimulating carbon storage in organic matter of natural and semi-natural ecosystems by increased input of nitrogen is potentially a relevant but temporary option to compensate for the emission of GHG. The use of this option as part of climate policy can conflict with nitrogen policy, because the increased generation (or lower ambitions for reduction) of reactive nitrogen by production of fertilizer and fossil fuel combustion leads to effects such as eutrophication, acidification, air pollution, etc. A complete quantification of the effects of nitrogen use on the exchange of CO_2, N_2O and CH_4 by means of a life cycle analysis (LCA) will give more insight in the most optimal use of nitrogen in agriculture with a focus on the net GHG effect.

Policies aimed at enhancing carbon sequestration in forests, (e.g.) by allowing a continuation of the present high nitrogen deposition levels, implies a larger value being put to the societal damage due to climate change compared to eutrophication. At the moment science cannot provide a suitable method for supporting this decision. Societal damage by climate change and eutrophication are both originating from changes in biodiversity, but proper weighting requires that this damage (and other negative effects) are expressed in comparable units.

Observation 3: Need and opportunity for the reduction of nitrogen fertilizer

Given the negative effects of an excess use of fertilizers, in terms of eutrophication, acidification, large scale air pollution, water pollution and climate change (GHG emissions), it is very relevant to optimise the production and use of chemical nitrogen fertilizer. Emission of N_2O during production of chemical fertiliser can be reduced by implementing existing process technology. There are also various options to increase the efficiency of the use of chemical fertilizer. The current opinion in agriculture is that when cheap fertilizer is available, using large quantities of nitrogen is a good option to lower the risks for reduced profits by lower crop yields. However, when also considering societal damage due to climate change and other nitrogen related effects, the economical optimal fertilisation level will be lower. Particularly in the industrialized countries measures aimed at increasing the fertiliser efficiency are effective to mitigate climate change and other nitrogen related effects without additional risks to food production.

Observation 4: Integrated nitrogen policy requires more insight in costs and benefits

It is currently almost impossible to weigh the costs and benefits of different options for nitrogen and climate policy. Whereas some effects can be monetised, the contribution of nitrogen can not always be quantified. This holds e.g. for the additional CO_2 sequestration by nitrogen enrichment, the contribution of nitrogen to global cooling of aerosols, the pollutant swapping between ammonia and N_2O, contribution to biodiversity loss, the chain of bio-fuels, etc. More research needs to be done to find a common ground for comparison of different measures and to quantify damage effects per unit of nitrogen for the various nitrogen compounds.

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List of abbreviations

BAT	Best Available Techniques
C	Carbon
CH_4	Methane
CO_2	Carbon Dioxide
DIC	Dissolved Inorganic Carbon
DME	Dimethyl Ether
DMS	Dimethylsulfide
DMSP	Dimethylsulfoniopropionate
DNDC	Denitrification-Decomposition
DOC	Dissolved Organic Carbon
EDGAR	Emission Database for Global Atmospheric Research
EFM	Edinburgh Forest Model
EIA	Energy Investment Deduction
EJ	Exa Joules (10^{18} Joules)
EMEP	European Monitoring and Evaluation Program
ETBE	Ethyl Tert-Buthyl Ether
EU	European Union
FAEE	Fatty Acid Ethyl Ester
FAME	Fatty Acid Methyl Ester
FAO	Food and Agriculture Organization
GAP	Good Agricultural Practices
GE	Global Economy (emission scenario)
GHG	Greenhouse gases
GWP	Global Warming Potential
IGCC	Integrated Gasification Combined Cycle
IPPC	Integrated Prevention and Pollution Control
IPCC	Intergovernmental Panel on Climate Change
LCA	Life Cycle Analyses
MEP	Milieukwaliteit Energieproductie
MTR	Maximaal Toelaatbaar Risico (Maximum Permissible Risk)
N	Nitrogen
ND	Nitrate Directive
NEC	National Emission Ceilings
NCD	National Ceilings Directive of the EU
NGCC	Natural Gas Combined Cycle
NGE	Net Greenhouse-gas Exchange
NH_3	Ammonia
NMP	Nutrient Monitoring Program (OSPAR)
N_2O	Nitrous Oxide
NO	Nitrogen Mono-oxide
NO_2	Nitrogen Dioxide
NO_x	Nitrogen Oxides (NO + NO_2)
NO_3	Nitrate
NPP	Net Primary Production
NUE	Nitrogen Use Efficiency
OSPAR	Oslo & Paris Convention
P	Phosphorus
PJ	Peta Joules (10^{15} Joules)
PM	Particulate Matter
POC	Particulate Organic Carbon
POR	Regulation on derogation production rights manure legislation (in Dutch: Regeling Ontheffing Productierechten Meststoffenwet)
RME	Rapeseed Methyl Ester

ROB	Reductie Overige Broeikasgassen (Reduction Other GHG)
SAR	Second Assessment Report of the IPCC
SE	Strong Europe (emission scenario)
SCR	Selective Catalytic Reduction
SME	Soybean Methyl Ester
SNG	Synthetic Natural Gas
SO_2	Sulphur Dioxide
SOC	Soil Organic Carbon
TAR	Third Assessment Report of the IPCC
UN	United Nations
UNFCCC	United Nations Framework Convention on Climate Change
VAMIL	Willekeurige Afschrijving Milieu-Investering
VOC	Volatile Organic Compounds
WAB	Wetenschappelijke Assessment en Beleidsstudie
WFD	Water Framework Directive
WTT	Well-To-Tank
WTW	Well-To-Wheels