The Impact of Industrial Nitrogen Fertilizer Taxation on Agricultural N_2O Emissions, Agricultural Production and Food Prices

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Acronyms

AFR Sub-Saharan Africa, MAgPIE region, see Annex A.4
AWMS animal waste management system
CPA Centrally planned Asia (incl. China), MAgPIE region, see Annex A.4
EUR Europe (incl. Turkey), MAgPIE region, see Annex refannex-countries
FSU Former Soviet Union, MAgPIE region, see Annex A.4
GHG greenhouse gas
IPCC International Panel on Climate Change
LAM Latin America, MAgPIE region, see Annex A.4
MAgPIE Model of Agricultural Production and its Impact on the Environment
MEA Middle East/North Africa, MAgPIE region, see Annex A.4
Mt million tons
NAM North America, MAgPIE region, see Annex A.4
Nr reactive nitrogen
N_2O nitrous oxide
ODS ocone depleting substance
PAO Pacific OECD (Japan, AUS, NZL), MAgPIE region, see Annex A.4
PAS Pacific Asia, MAgPIE region, see Annex A.4
SAS Southern Asia (incl. India), MAgPIE region, see Annex A.4
Tg Teragram

Abstract

Since the beginning of the industrial revolution, reactive nitrogen (Nr) in terrestrial systems has more than doubled. Reactive nitrogen is not only an important nutrient for plant growth, thus indirectly safeguarding human alimentation; it also disturbs natural systems when it is abundantly available. In the form of nitrous oxide (N_2O), it is a key driver of ozone depletion and global warming.
There is thus a trade-off between the aim of boosting agricultural production with reactive nitrogen and the wish to preserve natural services which are endangered by nitrogen pollution. The current markets are unable to deliver a social optimal outcome to this trade-off.
This study presents a model to evaluate the impact of an industrial nitrogen fertilizer tax on agricultural N_2O-emissions, agricultural production and food prices. The model is based on the Model of Agricultural Production and its Impact on the Environment (MAgPIE), which determines optimal agricultural cultivation patterns on a global scale. The existing model is extended with nitrogen balance constraints to estimate emissions accurately and to simulate the impact of input-taxes.
The results of the model indicate that a tax has the potential to mitigate emissions at low costs; yet food prices react sensitively to taxation and create a clear trade-off between emission mitigation and nutrition safety.

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Chapter 1

Introduction

1.1 Problem and Relevance

Reactive nitrogen (Nr) is a key nutrient, which is required by all living beings on this planet. Yet, its fixation is very energy-intensive and hardly occurs through natural processes. The scarcity of reactive nitrogen thus severely limited pre-industrial agriculture and restricted the carrying capacity of the planet (Smil, 2002). The industrial revolution changed the nitrogen-cycle substantially. Fossil fuel combustion and land-use change released reactive nitrogen, which was stored in long-term reservoirs, and the discovery of ammoniac synthesis by Fritz Haber and Carl Bosch allowed the production of cheap industrial nitrogen fertilizer out of natural gas. The terrestrial production of reactive nitrogen more than doubled since pre-industrial times (Galloway *et al.*, 2005).

The new affluent availability of the once scarce nutrient reactive nitrogen unleashed a green revolution in agriculture. However, while an adequate nutrition of the world population would not be possible without industrial nitrogen fertilizers, the abundance of reactive nitrogen also endangers the fragile stability of the planet. The earth system is *'tuned to operate with reactive nitrogen in short supply'* (Sutton *et al.*, 2009). The oversupply of reactive nitrogen has deleterious effects on air quality, atmospheric chemistry, soil conditions, biodiversity and water quality. Nitrous oxide(N_2O) plays a central role in nitrogen pollution.

N_2O is an ocone depleting substance (ODS). It can destroy the protective stratospheric ozone layer of our planet and thereby expose living beings to harmful radiation. Ravishankara *et al.* (2009) diagnosed recently that N_2O is the single most important ODS and will probably remain so for the next century.

Furthermore, N_2O is a very powerful greenhouse gas (GHG) which absorbs the infrared radiation from the sun and traps heat within the atmosphere, causing global warming. Nitrous oxide is a GHG 298 times more powerful than CO_2 and has an average atmospheric lifetime of 114 years. Since the industrial revolution, its atmospheric concentration rose from 270 to 319 parts per billion in 2005, and it continues to rise by 0.26% per year (Forster *et al.*, 2007). N2O emissions make up $\approx 8\%$ of total anthropogenic emissions and are mainly emitted in agriculture (Bernstein *et al.*, 2007). If current trends persist, global warming may likely transcend the 2° Celsius benchmark and cause irreparable harm to natural systems and human civilisation (Bernstein *et al.*, 2007; Richardson *et al.*, 2009).

Climate change and the depletion of the stratospheric ozone layer are *'global subjects'* (Schellnhuber, 1999), a challenge to present and future generations. It will show whether humanity can develop an identity of shared fate and counter these threats with adequate action.
While the Montreal protocol, an international binding treaty aiming to protect the ozone layer, showed that collective action can be successfully reached also on a global scale, N2O is not included into the treaty up to now.
Combating climate change is an even more ambitious plan. The Kyoto-Protocol, an international treaty that institutes measures for the stabilization of greenhouse gas concentrations in the atmosphere at sustainable levels, was enacted in 1997 and entered into force in 2005. The treaty sets a limit to GHG emissions of industrialized countries and introduced flexible mechanisms like emission trading to reach this target. Yet, up to the present, the Kyoto Protocol excludes most GHG emissions that occur in land-use, land-use change and forestry(LULUCF). Agricultural emissions are fully neglected, even though they account for $\approx$ 10-12% of total anthropogenic emissions (Smith *et al.*, 2007b), and might account for 10-40% of the global cross-sectoral mitigation potential in the next century (Rose *et al.*, 2007). It remains open which changes the next United Nations Climate Change Conference in Copenhagen achieve in December.
While it is a declared aim to include all type of relevant sources and sinks of GHG into the climate change policies (UNFCCC, 1992), how to include LULUCF is an ongoing question. Discussions remain controversial because the sector is subject to large scientific uncertainties and touches a number of vital issues: Ecosystems are not only sinks and sources of greenhouse gas emissions, but also supply food, fuel and shelter to humans, provide environmental services, create livelihoods and protect biodiversity (Schlamadinger *et al.*, 2007b). Even though these aims may be compatible in some cases, there are certainly also trade-offs which have to be considered. Most importantly, GHG mitigation may increase the costs of agricultural production and thus food prices. This may lead to malnutrition, subalimentation or hunger.
A better understanding of these interdependencies is required to facilitate decision processes and ensure the effectiveness of political action. Like a look into the mirror, the scientific community has to show the world where it stands. Yet a normal mirror would not suffice. A magic mirror is required that can not only show the world as it is, but also how it will be and how it could be. These rather pretentious features are necessary to help humanity find solutions, analyzing the impact that different solutions might have and thus informing rational decisions on which actions are most adequate.

This thesis shall present a sketch of such a magic mirror, still one that rather belongs into a cabinet of curiosities than into the empresses castle. The mirror is an integrated economic model which describes the present agricultural sector and the N2O emissions attributed to agricultural production, forecasts their future developments under business-as-usual conditions and simulates future developments under the condition that an industrial nitrogen fertilizer tax is introduced. The model shall give an impression of which impacts an industrial nitrogen fertilizer tax may have on the agricultural sector and on food prices, and which role it could play for climate change mitigation. The latter task requires a model on a global scale, merging the mitigation potential of all world

regions.
This thesis will use the Model of Agricultural Production and its Impact on the Environment (MAgPIE) (Lotze-Campen *et al.*, 2008, 2009), which is well suited for the above task: It is a global economic land-use optimization model with high spacial resolution and an extensive representation of the agricultural supply-side. Yet, as the current version neither simulates nitrogen-flows, nor N_2O-emissions or the simulation of policy-impacts, it has to be adapted for this task.

1.2 Distinctiveness of the Approach and Literature Overview

While environmental research started off from a purely natural science perspective, it now increasingly includes the anthropogenic sphere as a key factor in natural processes. The impact of humankind on the earth system is of such severity that it can no longer be considered as a black box with undefined properties. Actually, reactive nitrogen accumulation, climate change, sea-level rise or deforestation are in principal *'social phenomena'*, driven by human industry, fossil fuel combustion and land-use change. Yet, the capability of understanding these social processes still lacks far behind the scientific skills for understanding natural processes. (Lucht & Jaeger, 2001; Lotze-Campen *et al.*, 2002)
At the same time, the economic theory of the last century was a science almost detached from the natural world. Interrelations were estimated econometricly using time-series data, highly abstracted (for example in the form of production functions with few parameters like capital and labor) and calibrated with measured economic data of a base year; underlying material flows and natural limitations remained largely unconsidered. Thus, such frameworks where incapable of coping adequately with natural limitations and issues of sustainability (Lucht & Jaeger, 2001; Lotze-Campen *et al.*, 2002). Hence, the accuracy of natural science models may be a chance to endow economic models with better predictive capabilities.
It is thus time for both sciences to invest into the holistic approach of earth system analysis. This study shall take this approach, taking into account the constraints of natural processes on the economic sphere and the feedback of the economic sphere on the environment.
On the one hand, MAgPIE has certain features of a bio-physical natural science model: it is based on the simulation of potential agricultural yield levels for numerous crop types, livestock types and cultivation areas. Also the model-extension realized for this thesis is grounded on a phsyical material-flow reasoning, which attachs a nitrogen input or withdrawal to every agricultural activity. This physical foundation allows for a finer depiction of the agricultural production function, the *'action space'* in which social interactions may take place. This detailed production function of the model is a precondition for simulating (instead of presupposing) structural shifts within the agricultural sector; it could not be realized with a standard production function like the Cobb-Douglas function (Mitra-Kahn, 2008). The disaggregated production function also helps to estimate the amount of pollution. The underlying material flows allow for a

quantification of emissions that are a prerequisite for estimating welfare losses of aggrieved parties.
Yet, a purely natural science model would also be unable to simulate the dynamics within the agricultural sector. It could not explain why farmers choose to cultivate certain crop types in the current quantity and at the current location, and it could not account for changes in these cultivation patterns. It could also not explain why additional constraints (say a policy intervention) on one good may have an effect on the produced quantity of a substitute. Yet, these dynamics can be partly explained by the optimization process of resource allocation realized through relative market prices (Perman *et al.*, 2003).
Due to this holistic approach of the MAgPIE model, it can explain certain dynamics and interactions that are insufficiently depicted by other studies.

1.2.1 Simulation of agricultural nitrogen flows

Current studies which estimate industrial fertilizer consumption use mainly two approaches:
The first, 'statistical' approach uses regressions based on past observed datasets on food consumption, population, GDP growth or other parameters and the observed industrial fertilizer consumption. Then it employs future projections of the independent variables to prognose future industrial fertilizer consumption. Examples for this type of approach are Bumb (1995); Daberkow *et al.* (2000); Dyson (1998); Gilland (1993) and probably Prud'homme (2005), although the methodology is not made explicit in the last case.
The second, 'agronomic' approach assumes that the reactive nitrogen withdrawn by the harvested crops has to be replaced by industrial fertilizer. A fixed withdrawal/input ratio replaces the regression. Examples for such studies are Bumb & Baanante (1996) and Frink *et al.* (1999).
Even though these approaches are often enriched with expert judgments or further statistical finesses, this can not conceal the fact that they have a poor theoretical back-up. While the first type of approach is purely statistical and holds no information about underlying processes, the second approach at least uses a simple material-flow reasoning. Yet, none of the latter studies accounted for the fact, that there are numerous reactive nitrogen inputs besides industrial nitrogen fertilizer, and that their proportion of total reactive nitrogen inputs is not necessarily fixed. The dynamics determining future changes in these proportions are very complex and certainly difficult to estimate by expert guesses.

1.2.2 Simulation of N_2O emissions

There are several studies, which estimate the current and future N_2O emissions in agriculture.
Crutzen *et al.* (2007) take a very simple top-down approach. They start from the annual amount of total nitrogen emissions, which can be measured in the atmosphere. Thereof, they subtract the pre-industrial level of emissions to receive anthropogenic emissions. Subtracting the N_2O emissions of all other major polluting sectors (mainly industry and land-use change) leaves the agricultural N_2O emissions as residuum. The estimate, as simple as it is, may be a good cross-check for bottom-up estimates. Yet it can hardly be useful to estimate the

precise future developments of the agricultural sector.
Bouwman & Boumans (2002) estimate the N_2O-emissions from a bottom-up approach from 846 field measurements. Yet, the study only includes direct agricultural soil emissions, and does not account for emissions occuring in animal waste management systems or for indirect emissions which occur after leaching or volatilization of N2O.
In 1996, the International Panel on Climate Change (IPCC) published Guidelines for National GHG Inventories (IPCC, 1996), which allow for a comprehensive estimation of agricultural emissions. This methodology has been taken up or slightly changed by a large number of studies. Oenema *et al.* (2005); Strengers *et al.* (2004); US-EPA (2006a), calculated N_2O-emissions on a global scale, while Food and Agriculture Organization (2006) focuses only on the direct and indirect N_2O-emissions from livestock production.
In 2006, the IPCC published new Guidelines for National GHG Inventories (IPCC, 2006). Yet, up to the moment no major study was found that adopted this methodology into its estimates. The new methodology has substantial differences in emission factors to the 1996 methodology.

1.2.3 Simulation of mitigation costs and policy impacts

A large number of studies, which examine the costs of mitigating agricultural non-CO2 emissions, are based on a bottom-up approach. They estimate costs and emission-savings of specific abatement measures, e.g. costs and emission savings of conservation-tillage. Then, costs and savings of the abatement option are multiplied with the agricultural area to which they could be applied. Examples for these studies are DeAngelo *et al.* (2006); Graus *et al.* (2004); Rose *et al.* (2007); Smith *et al.* (2007a) and US-EPA (2006b).[1]
The bottom-up approach takes market prices for agricultural products as exogenously given. Yet, many of the abatement measures, as for example reduced fertilization, have also an impact on total agricultural output. Reduced supply will most probably lead to higher market prices. This food price increase might in turn give an incentive to expand production and thus fertilizer inputs. Parts of the tax impact might thus be offset by these macro-effects and cannot be covered by a bottom-up approach.
While the studies just mentioned only calculated economic potentials of emission abatement, there are also studies which explicitely estimate the impact of a tax:
Xiang *et al.* (2007) estimate the price elasticity of fertilizer consumption in order to simulate the effect of a fertilizer tax on nitrogen application. This is certainly a suitable approach for estimating the policy impact of rather low taxes. Yet, this approach might be unable to estimate the impact of a tax with a powerful steering effect. For considerable steering effects, it can not be assumed that farmers react to large scale price shifts (which can hardly be observed in reality) as they would do to marginal shifts.

[1] Despite the large number of publications, the literature is heavily interlinked and self-referential. For instance, all studies of non-CO2 mitigation mentioned above are directly or indirectly based on Bates (2001) and Gerbens (1998) or both. These two studies investigate a rather small number of case studies in Europe. Even though subsequent studies corrected or enlarged the original investigations, there is still an enormous lack of cost-benefit analysis on the ground.

Berntsen *et al.* (2003) use the farm-level model FASSET to assess the impact of a fertilizer tax on Danish farms. The study accounts for a very large number of nitrogen flows on farm-level and has an economic optimization approach to choose the farming activities according to profit maximization. The model is very suited to assess the impact of a tax on a single farm. Yet, the model can only be used for estimating the imapct of certain sample farms, and cannot be used for an economy-wide or even world-wide scale, because bottom-up data is missing and macro-effects are not considered.

Summing up, extensive research has been carried out on nitrogen flows, N_2O-emissions and simulation of mitigation costs and policy impacts. Yet, the model presented in this study can make a number of contributions, which have not been covered by recent studies.
Concerning the estimation of nitrogen flows, it presents an approach with theoretical back-up, considers a large number of nitrogen flows and determines the strength of each flow according to endogenous price dynamics. A high spatial disaggregation furthermore guarantees that not only the global nitrogen budget is balanced, but also that local systems fulfill this condition. In this way, industrial nitrogen fertilizer demand can be isolated more clearly.
Emission forecasts can be improved, because the study is based on the new 2006 IPCC Guidelines of national GHG inventories, and because the emission sources are calculated endogenously in the model. Also, the model allows for a spacial allocation of nitrogen emission sources.
Finally, the model creates a strong framework to estimate mitigation potentials and policy impacts on N_2O-emissions. Compared to other studies, it can simulate the policy impact on the agricultural price system and estimate substitution elasticities and mitigation costs endogenously.
The strengths in these three issues should not be seen each for their own, as their bundling in one model may lead to synergistic effects: Better depiction of nitrogen flows lead to better estimation of N_2O emissions; when nitrogen limitations are taken into account, the economic dynamics can be simulated more adequately; in turn, when nitrogen flows and emissions can be simulated well and the economic dynamics are understood better, the impact of policy intervention can be estimated more precisely.

1.3 Overview

Following this introduction, the second chapter of this thesis will develop the theoretical background. It shall be described how agricultural production is linked to nitrogen application, how nitrogen application is linked to pollution and how pollution is linked to welfare losses. To give a holistic picture of natural interdependencies, all types of nitrogen pollution (not only N_2O) shall be discussed. Then it shall be explained, why the current markets are unable to react appropriately to pollution and which policies are available to the state to combat the resulting market failures. Every policy has certain advantages or disadvantages; the study will highlight them and suggest on this basis that an industrial fertilizer tax may be an adequate solution. To test this hypothesis, five indicators of policy efficacy will be developed, which may help to judge the effectiveness of an industrial fertilizer tax.

The third chapter describes the framework of the agro-economic optimization model MAgPIE[2], which shall be used to quantify the indicators. The first section of the chapter will explain the standard version of the MAgPIE. Yet, this standard version is unable to estimate emissions or the impact of certain policies. Therefore, the second section of the chapter will describe the model-expansion, which was developed for this thesis. While the standard version of the model does not account for the nitrogen cycle, the extended model implements a nitrogen market with various activities influencing Nr inputs and withdrawals. A new constraint is introduced that requires that the nitrogen budget is balanced out in all production areas. Based on the simulated nitrogen budgets, the model calculates N_2O-emissions using the newest IPCC Guidelines for National Greenhouse Gas Inventories from 2006 (IPCC, 2006).
In the fourth chapter, the model-outputs shall be presented, in particular the indicators developed in the theory-chapter. The results shall be briefly compared to other studies and measured data.
The fifth chapter will explain the limitations of model framework, and reveal underlying assumptions and data shortcomings. The model output from the fourth chapter will then be discussed under these considerations.
Finally, the conclusion will name some implications, which may be drawn from the study, and give an outlook on required future research.

[2]MAgPIE stands for *'model of agricultural production and its impact on the environment'*

Chapter 2

Theory

The aim of the Theory Chapter is to find adequate policy options to combat nitrogen pollution and to develop indicators to rate whether these policies are effective and efficient instruments under a broader set of aims. The argument will be developed in four parts.
In the first section of the chapter, the reader will take the perspectives of three agent. The first perspective is the position of the farmer: why do farmers apply nitrogen to their soils and what determines which actions they undertake? Secondly, we will follow the voyage of an arbitrary small reactive nitrogen compound from its fixation to its float through nature until it finally becomes uncreative again. The multiple transformations and the effects it had on its environment shall be described. Thirdly, the reader examines how people are affected by the alterations of natural systems, and it shall be explained why these alterations have a negative impact on most people below the line. The impact may be dampened by adaptation, yet a negative residual damage remains.
The second section of the chapter will ask the question, how a balance can be found between the opposing interests of farmers and affected people. In the second section, it will become clear, that current defective markets cannot harmonize the interests of farmers and affected parties. It shall be explained why the presence of external effects leads to a situation which is not optimal from a social welfare perspective.
In the third part, the study will evaluate different policy options which tackle market failure, and explain their method of internalisation. Because they are of special importance as regards multi-source nitrogen pollution, policy implementation costs will be evaluated separately. These costs circumvent that the impact of market failure can be undone completely and require second-best solutions. Policy implementation costs also determine the superior eligibility of incentive-based policies like taxes, subsidies or certificate markets over hard directives. Finally, it shall be evaluated, what effects different policies have on the distribution between individuals. Of special importance are the effects on low-income consumers and on the state budget.
The fourth part will then present one possible second-best policy option concerning nitrogen pollution, the taxation of industrial fertilizer. It shall be shortly discussed what advantages this option holds before we ultimately develop some quantifiable indicators that will allow to make policy options comparable.
In chapter 3, these indicators shall be computed for an industrial fertilizer tax

and for an emission tax, using an empirical model.

2.1 Agent Perspectives

2.1.1 The farmer's perspective: nitrogen input and nitrous oxide emissions

This section will shortly illustrate, why farmers need nitrogen inputs, why nitrogen is a scarce good, which natural and anthropogenic supplies of nitrogen exist, and how the farmer makes up his mind about the type of nitrogen fertilization and the right quantity to use.
In a very broad definition, farmers can be seen as producers of organic materials of plants for food and feed or for textiles and other industrial purposes; livestocks for meat, milk, eggs, leather and other purposes. All organic materials feature DNA, RNA and proteins that are built out of nitrogen compounds. The chlorophyll molecules that are essential for photosynthesis are also composited out of nitrogen(Kramer, 2005). The nitrogen content of dry biomass is substantial: Plants contain approximately 2-6% (Taiz & Zeiger, 2000), meat approximately 14 to 20 percent nitrogen(Smil, 2002). For their nutrition, every human requires at least 3 kg Nr per year(Loomis & Connor, 1992; Frink *et al.*, 1999). Thus, nitrogen is an essential prerequisite for any kind of life on this planet and a non-substitutable input for agricultural production. Farmers have to ensure sufficient nitrogen availability to be able to produce a given quantity of plants and livestock.
While there is a need for nitrogen, the supply of nitrogen is unfortunately rather limited. This seems paradoxical, as it is available in almost infinite amounts in the environment: 78% of the atmosphere consists of nitrogen. However, in most cases nitrogen is bound as N=N in a paired-atom structure. Plants and animals are unable to make use of this molecule. It needs to be split and rearranged (a process often paraphrased as *'fixation'*) to reactive nitrogen (Nr), for example as an ammoniac compound (NH3). Yet, separating this strong bond is very energy-intensive and hardly occurs in nature. Additionally, Reactive nitrogen frequently decomposes back to N2 in the process of denitrification. [1] Thus, reactive nitrogen remains a scarce good (Smil, 1997).
Still, two ways of natural fixation of nitrogen exist: Firstly, the event of lightning sets free large amounts of energy and fixes nitrogen from the atmosphere.[2] Secondly, certain bacterias and microorganisms are able to fix nitrogen(Smil, 2002). This form of nitrogen-fixation is also very energy-consuming, and therefore many nitrogen-fixing bacterias live in symbiosis with certain plants: mostly Leguminosae (legumes, peas, beans and pulses), but also sugar cane, rice or alder trees. While the plants are profiting of the nitrogen fixed by the bacteria, the bacteria is profiting of the photosynthetic energy of the plant. It is estimated that the rhyzobia-bacterias, living in symbiosis with legumes, consume as much as 20% of the photosynthetic energy of the plant (Deacon, 2009). Up to the middle of the 20th century, agriculture was almost entirely based on these two

[1] denitrification is the anaerobic microbial reduction of nitrate to nitrogen gas (N2)(IPCC, 2006).
[2] Fields who experienced a recent lightning strike often possess outstanding green vegetation.

natural types of nitrogen fixation. By the cultivation of legumes, observance of fallow periods, and by growing and ploughing in cover crops, this nitrogen-fixation was consciously accelerated by agricultural practices. Furthermore, Nr containing wastes like manure, slurry, plant residues or compost were reintegrated into the the nitrogen-cycle. Nevertheless, natural potential set certain boundaries to agricultural production: even under optimal circumstances, Nr input per hectare was limited to about 120-150 kilograms N. Furthermore, the required cultivation of legumes has lower yields and a more restricted field of utilization than conventional crops have. Hence, the lack of reactive nitrogen was for a long time one of the hardest constraints on agricultural production (Smil, 1997, 2002).

It was not until Carl Bosch and Fritz Haber developed the chemical process of ammoniac-synthesis in 1910, that agricultural production was able to disengage from its natural limitations. Their invention allowed the conversion of N=N to ammoniac by letting nitrogen from the atmosphere react with hydrogen, mainly from natural gas. Starting from the 1940s, the use of chemical fixed nitrogen grew exponentially and lead to a 'Green Revolution' in agriculture. Today, chemical fixation delivers already 40% of the Nr demand of agriculture (Smil, 2002). Still, the Haber-Bosch Synthesis is also very energy-intensive. Roughly 1% of the global energy production is used for the fabrication of nitrogen fertilizer(Smith, 2002).

In the last century, two more 'sources' of reactive nitrogen emerged: the combustion of fossil fuels, which volatilizes reactive nitrogen from below-ground resources, and land-use change (mainly deforestation and drainage of wetlands) which sets free the reactive nitrogen that was stored in living biomass. Also parts of the nitrogen from these sources reach agricultural soils and serve as inputs to agricultural production (Vitousek *et al.*, 1997)

Summing up, the different nitrogen inputs have approximately the following magnitudes: Lightning accounts for $\approx$ 5 Tg[3] Nr(Galloway *et al.*, 2005), biological nitrogen fixation on agricultural land now produces $\approx$ 40 Tg Nr per year; the combustion of fossil fuels sets off $\approx$ 20 Tg Nr per year; deforestation, biomass burning and wetland drainage release $\approx$ 70 Tg Nr per year; and the chemical fixation of nitrogen for industrial fertilizer creates $\approx$ 80 Tg Nr per year. Compared to these anthropogenic inputs of more than 200 Tg Nr, natural terrestrial fixation of nitrogen is $\approx$ 90-140 Tg Nr per year (Vitousek *et al.*, 1997, estimates for the early 1990s).

After explaining which sources of reactive nitrogen are available to the farmer, the following part will investigate how the farmer makes up his mind to deliver the nutrients required for his production. Farmers have multiple options to provide the nitrogen for their crops. They have a large action space, in multiple dimensions: First, they can select the crop type they want to cultivate, with individual need for nitrogen. They can choose the intended output level, with higher yields requiring more nitrogen inputs. Second, they can choose among different nitrogen sources, including industrial fertilizer, biofixation, crop residues and compost and other nitrogen-enriching practices like fallow periods or the ploughing in of cover crops. Thirdly, they can increase the efficiency of fertilization. Losses usually occur when reactive nitrogen is not taken up immediately by plants. They are especially high in humid and warm areas. Losses

[3] One Teragram (Tg) equals 10^{12}g or one million metric tons

of reactive nitrogen can be minimized when farmers choose the right source, rate, time and place of application(Roberts, 2007). To minimize losses, reactive nitrogen sources should be conform with soil, climate and crop conditions. Additional to fertilizer, nitrification or urea inhibitors can reduce run-off and volatilization of reactive nitrogen.

The optimal rate of application can be estimated on the basis of soil nitrogen balance calculations, including all inputs and withdrawals of an agricultural field. This practice can be improved by representative soil, drainage water and plant tissue analysis. Nr input should be balanced with other nutrients, as reactive nitrogen might not be taken up because of other nutrient limitations. Fertilization timing should coincide with plant uptake of nitrogen. Farmers should account for the main growing times of their plants and bring out fertilizer multiple times during the year to minimize losses. Fertilizer placement should deliver the nitrogen to the parts of a field that actually require nutrients. Fertilizer application equipment should be maintained and well-calibrated to prevent application overlaps. Sensors and satellite data can be used to apply the fertilizer to the appropriate places. Subsurface application of fertilizer like anhydrous ammonia should be in adequate depth. Finally, all management practices should be coordinated and planned on a system-level, accounting for several growing-periods and crop rotations(Snyder *et al.*, 2009).

Given the large action space of the farmers, which options will they choose? To predict their behavior, one can make certain assumptions: First, that their welfare is positively correlated with the income gained from their agricultural enterprise. Second, that they seek to maximize their personal welfare. And third, that they act rationally. On the basis of these assumptions, it becomes clear that farmers will select the cheapest nitrogen sources available. The costs of each source are composed of different elements: purchase price for inputs bought on the market (e.g. artificial fertilizer), labor time (e.g. for deploying fertilizer), capital costs (e.g. spreader maintenance), opportunity costs (e.g. lower market price for legumes), education costs (e.g. knowledge about adequate fertilizer timing) and so on. These costs do not remain static, but change with market prices and available technology. Price changes induce substitution between different agricultural activities. For example, when industrial fertilizer prices rise due to higher natural gas prices, farmers will increasingly plant legumes or put more effort on improving fertilizer efficiency. Thus, the mix of Nr sources remains flexible and difficult to foresee.

If one takes into account, that farmers often possess incomplete information on their own production function and on upcoming external events, some peculiarities of their fertilizing behavior can be explained. First, farmers will never exactly hit upon the right amount of fertilizer that is just sufficient for the intended output level. They will either under-fertilize the soil, which leads to restricted plant growth or minor quality, or over-fertilize, which leads to waste of nitrogen or quality losses(). Second, as lack of nitrogen is a hard constraint for plant growth and leads to higher income losses than wasted fertilizer, farmers tend to use excessive amounts of fertilizer to minimize the risk of under-fertilizing (Pearce & Koundouri, 2003). Thus, the less information they possess, the more fertilizer they will use, and the lower will be the nitrogen efficiency. Similar effects can be induced from the existence of fixed costs: optimum fertilization would deliver the Nr exactly at the points in time, when it is needed for plant growth in order minimize losses. However, bringing out fertilizer has high fix

costs, as it takes time and other costs of operation to spread the fertilizer on the field. Thus, the higher the fix costs are, the less often will the farmer enter the field to fertilize, the higher are Nr losses and the lower will be agricultural efficiency (Sheriff, 2005) (See picture).
In conclusion, Nr is an essential input for all kinds of farm production. Multiple options exist to satisfy this requirement, however, farmers will select one definite set of options in their pursuit of income maximization. This set is defined by the prices that are formed on the market. It was also shown, that incomplete information and fixed costs are good explanations for nitrogen use which exceeds the absorption by agricultural products. The next section will go into the destiny of the reactive nitrogen that was not incorporated into plants and explain the effects which it has on the environment.

2.1.2 The multiple destinies of nitrogen

It makes sense to evaluate the further processes of nitrogen independently from the production function of the farmer, as the farmer is unable to influence much of what was triggered by his behavior. This section will describe the effects reactive nitrogen has on the environment, while the following section will have a look at how these changes of the environment affect human welfare.
When the reader takes the position of a small, reactive nitrogen compound in the following, he has to bear in mind that it is the life cycle of a special nitrogen compound. Not all particles have this type of life cycle. Still, it is not an unusual or unrepresentative one: reactive nitrogen can have multiple destinies, the so-called "‘nitrogen cascade"’ (Galloway *et al.*, 2003), before it finally returns into its nonreactive N2 bound.
The small, reactive nitrogen compound came into being when it was fixed by the Haber-Bosch Process, pulled out of its non-reactive triple-bound to end up in an ammoniac compound. Afterward it was made to react with nitric acid to form ammonium nitrate(NH4NO3), an ordinary industrial fertilizer. When the fertilizer was applied to the soil, it belonged to the unfortunate part which was not attracted by any of the plant roots, but remained in the soil. Frustrated by its destiny, it turned into a free radical and left the soil as the gas NO. By nature unstable, it had not resided long in the troposphere when it collided with a volatile organic compound. In their reaction, they caused the production of ozone (O_3) (Green, 2008). Ozone in the troposphere has a deleterious effect on living beings. Plants can suffer visible injury and reductions in growth, animals and humans suffer from respiratory illness, cancer and cardiac disease.
Since the reactive nitrogen compound resided in the atmosphere, it was able to travel several hundred miles within few days(Dentener *et al.*, 2006). Therefore, when the small reactive nitrogen was washed down with the rain, it was far away from the agricultural area where it had originally volatilized. Taken up by the water, it increased the acidity of the rain. In the form of nitrous acid, it set free toxic inorganic aluminum which disturbs the root or foliar element ratio of trees. Unbalanced ratios of Ca:Al or Mg:N can lead to reductions in net photosynthesis or even increase tree mortality(Bricker & Rice, 1993).
Subsequently the reactive nitrogen compound was swept into a nitrogen poor river. Life here was specialized to take up the few nutrient inputs that are available, and the nitrogen got soon incorporated into an algae. Yet, the further down the algae was transported in the river, the more nitrogen was leaching

from the nearby agricultural acres, and soon nitrogen was abundantly available. Now, the plants that were specialized in surviving on low nutrient levels were displaced by other, faster-growing, nutrient rich species, which were better adapted to these circumstances. When the river came close to the coastal zone, eutrophication[4] increased and the biodiversity was largely reduced: small, unicellular algae dominated over more complex organisms, and light was absorbed in the upper centimeters of the water, making photosynthesis near the ground impossible. The ecosystem now became increasingly unstable: during the day, large amounts of oxygen were produced; however, in the night, bloom of respiring microorganisms led to anoxia(no oxygen) or hypoxia(low oxygen), which triggered the extinction of fish and shellfish resources. When the small nitrogen particle finally arrived in the ocean, the algae into which it was incorporated was long dead. It decomposed in the ocean, and the nitrogen was set free in form of the gas N_2O.

In this form, the reactive nitrogen was able to reflect solar radiation in spectral windows which are not absorbed by other gases and contributed to the greenhouse effect. It trapped heat radiation from the soil and increased global warming. As greenhouse gas, N_2O is 298 times more effective than CO2(Forster *et al.*, 2007).

The small reactive nitrogen compound remained for many decades (the average atmospheric lifetime is 114 years(Forster *et al.*, 2007)) and rose up to the stratosphere, where it was hit by solar rays in such way, that it became radical again and broke down into NO. In this form, it catalyzed the destruction of the ozone layer, a life-protecting shield of our atmosphere. Finally, declined got back to the troposphere, was washed down with the rain into a peat land, where it was denitrified back into N2.

This was the closing part of the exemplary life-cycle of a reactive nitrogen particle. All processes described above are well-documented in scientific literature: For the production of tropospheric ozone see (Green, 2008). The origin and impacts of acid rain can be found at (Wellburn, 1988) and (Cronan & Grigal, 1995). The problems of eutrophication are for example expounded in (Rabalais, 2002), and the effects on biological diversity in (Smith *et al.*, 1999), (Phoenix *et al.*, 2006), (Sala *et al.*, 2000), (Bobbink *et al.*, 1998). For Nitrous Oxide and impacts on climate change see Forster *et al.* (2007), and for the consequences on the stratospheric ozone layer see (Crutzen & Ehhalt, 1977).

As mentioned above, this is just one possible pathway of the nitrogen life cycle which does not depict all possible destinies of nitrogen. Reactive nitrogen has numerous further important impacts on natural systems: acidification does not only affect forests, but also lakes and streams (Schindler, 1988).

Eutrophication, in turn, also affects nutrient-poor terrestrial ecosystems(Matson *et al.*, 2002) and coral reefs(Rabalais, 2002). Forest and grassland productivity might increase up to a certain threshold (Aber, 1995) and function as carbon sink (Townsend *et al.*, 1996). Comprehensive studies that compile a large number of effects on natural systems are (Galloway *et al.*, 2003) and (Vitousek *et al.*, 1997).

Those were solely the direct impacts on environmental systems. Yet nitrogen pollution may trigger a whole chain of developments which can be shown best in the case of climate change: the greenhouse effect does not only lead to an

[4]Eutrophication is an increase of nutrients in a terrestrial or aquatic ecosystem.

increase of mean surface and ocean temperature, but may also change precipitation patterns, increase water scarcity, melt the arctic sea ice and thereby increase the sea-level, change the direction of ocean currents and promote the occurrence of extreme weather events. Because ecosystems are usually adjusted to the macro-climate, changes of the macro-conditions will lead to disturbance and displacements of ecosystems. Not all biological species are prepared for these adjustments, and a temperature increase will lead to the extinction of a significant number of species within the next 100 years(Richardson *et al.*, 2009). The extinction of species which bear a certain function within an ecosystem will lead to a further destabilisation of ecosystems.

In conclusion, while reactive nitrogen is essential to plant production, excessive quantities can fundamentally disturb natural processes, as the natural systems are balanced on moderate inputs. The deleterious effects occur in the atmosphere, in terrestrial and in aquatic systems.

Importantly, nitrogen pollution has some special characteristics that are outstanding: First, as reactive nitrogen can exist as gas and can be dissolved in water, it is highly volatile and extremely mobile. The consequences of redundant nitrogen are regional and global. Secondly, because of the cascade effect mentioned above, reactive nitrogen laid off at a certain place cannot be linked to just one environmental impact, but has a multitude of effects. Furthermore, it does not have to pass through all processes only once, but can remain within the cycle as some kind of catalyst. This leads to the third point: reactive nitrogen is a stock pollution and can remain within the cycle for indefinite time, or can be stored in long-term reservoirs. Thus, its release has consequences on the environment even centuries after its fixation.

These effects lead to the point that the geographical location and the entry point to the N-cycle, where the Nr is injected, is little relevant. It is more important, how much nitrogen is injected, and how much of it can be denitrified. The complexity of the processes and the comprehensiveness within space and time makes nitrogen pollution a so called 'non-point-source-emission'. This means that the pollution cannot be measured at few places and points in time, but can only be elaborately approximated on the basis of statistically extrapolating sample measurements (Xiang *et al.*, 2007).

2.1.3 Affected parties: impacts of nitrogen enrichment

The last subsection described major impacts of nitrogen pollution on the environment: production of tropospheric ozone, depletion of the stratospheric ozone layer, global warming, changed climatic conditions, increase of extreme weather events, sea-level rise, acidification, nitrogen accumulation in terrestrial and aquatic systems, eutrophication, altered and disturbed ecosystems and biodiversity losses.

The high mobility of nitrogen and its multiple effects on climate and biosphere lead to a situation, in which virtually all parts of the world are affected by nitrogen pollution. It is clear that not only the natural sphere but also the human sphere is concerned. The aim of this section is to analyze which impact nitrogen pollution has an humanity. It shall be explained which services the environment delivers; that both diminishment and change of these services will lead to a damage to humans, which are adapted to current situations; that adaptation may dampen the initial impact of changing environmental conditions, but that a residual damage will remain. Finally we will shortly explain why nitrogen pollution leads to rather heterogeneous impact on different individuals. Figure 2.1.3 gives an overview of major types of pollution, and some of their negative impact on human utility.

Natural services are usually subdivided into resource inputs (e.g. plants used for agricultural production or potable water), sinks for the assimilation of wastes(e.g. recycling capacities of aquatic systems), amenity services to households (e.g. recreational value of lakes or a beautiful landscape) or life-support services for firms and households (e.g. ozone layer or stable sea levels)(Perman *et al.*, 2003). Next to the use value of the environment which is derived from consumption, environmental systems may also have further values to humans: existence value arises independently of the consumption from the fact that the environmental service exists; option value arises from the possibility to use this service on day; and the bequest value arises from preserving a service for the afterworld ().

Humans depend on environmental services, and their delivery increases human welfare. Yet, the value of these services does not only depend on the quantity and quality of services provided, but also how strongly societies are adapted to natural conditions:

While societies settled in virtually all parts of the world and under the grimmest natural and climatic conditions, they managed to do so because they adapted their agriculture, forestry, settlements, industry, transportation, health, water resource management to local conditions(Downing *et al.*, 2001a). This required large amounts of human, cultural and physical capital that is largely untransferable and only functions in the local context; thus typical sunk costs that cannot be retained when another investment would become more convenient.

Knowledge about local climatic conditions, ecosystems and natural climate variability (for example understanding of the El-Niño-Southern-Oscillation) helps for example to determine optimal crop cultivation patterns or building adequate housing facilities or creating famine-early warning systems (Adger *et al.*, 2007). Yet this information is partly bound to understanding local climatic and weather phenomena and cannot be transferred to arbitrary contexts.

As Ostrom *et al.* (2002) points out, also cultural patterns and social institutions emerge around ecological differences. The creation of such institutions is an evolutionary process; a process that is closer to imitation than to learning, even though it follows certain rule. One rule pointed out by Ostrom is ‘When you're in Rome, act like a roman’: imitate the behavior of the people in your context to become adapted to local conditions. This is certainly only an effective strategy, if other social members are already adapted to more or less stable ecological system.

Societies might also adapt to environmental conditions with physical investments: building of dams reduce the impact of floods, water storage balances variations in water availability and irrigation systems distributes water to agricultural sites. Again, these investments are largely immobile, as can be well seen for the Indus irrigation projects in the Punjab province of Pakistan which ran out of water (Pearce, 2007). Also, changes in climate patterns might make these investments of no avail.

Finally, also the human health system adapts to the local environment by de-

veloping resistances against dominant diseases. Because of these resistances, epidemic can spread much slower and die back faster if they occur in regions that were already affected by this disease. Yet, new diseases spread fast and have deleterious effects.
The context-dependency of natural services is crucial to understand their value and also to understand the impact that altered conditions of climate and ecosystems have on social welfare. It is central, that not only diminished natural services may reduce welfare, but also that a changed set of services destroys the value of sunk cost capital. While a Mediterranean climate certainly offers a good basis for recreational value, a climate shift within an alpine ski region will definely cause more harm than do good.
Nitrogen enrichment of natural systems can have both positive and negative effects on human welfare.[5] Yet out of above-mentioned considerations, the immobility of capital makes shifts in macro-conditions which would be judged indifferently in the absence of investments, to an unfavorable change. Nitrogen accumulation is thus far from being a zero-sum game: there is large consensus in scientific literature that the negative impacts outweigh the positive ones.
This initial impact on humans may yet be dampened by adaptation. The adaptive capacity depends on several parameters. First, ecosystems themselves have a certain autonomous natural adaptive capacity and can absorb external shocks, for example through nitrogen incorporation into plant biomass or denitrification. This capacity is most probably larger for undisturbed ecosystems than for ecosystems already altered by human interference. Second, human societies possess an own adaptive capacity that may absorb negative impacts. According to Downing *et al.* (2001a) the adaptive capacity of humans is mainly determined by the availability of economic resources, current level and capacity to produce technology, high level of information and knowledge distribution, a comprehensive infrastructure and working social institutions. The impact which resides despite adaptation is called net-impact. Total damage costs include these net-impact plus the costs of adaptation that would not have been required without an anticipated change in climate and ecosystems(Smit & Others, 1999).
The damages occurring to different individuals, groups, societies and nations are very heterogeneous. Also, within societies they may differ between groups of different age, class, gender, ethnicity, religion, health and social status (Adger *et al.*, 2007). Variety of vulnerability comes from different impacts Nr enrichment has on different regions(islander are affected more by sea-level rise than mountaineers); the degree of dependency on natural services (women are more involved in livelihoods linked to natural services than men (Davison, 1988)); and the adaptation capacity available to the individual (homeless people are very vulnerable to environmental hazards (Wisner, 1998)).
In conclusion, a large proportion of present and future humans will suffer severely under the diverse impacts of nitrogen enrichment on natural services, even though the impact of nitrogen enrichment differs greatly among individuals. Adaptation may delimit the impacts of nitrogen enrichment; yet adaptation is costly and cannot circumvent certain net-impacts.

[5] positive effects are for example, that forest and grassland productivity might increase up to a certain threshold (Aber, 1995) and function as carbon sink (Townsend *et al.*, 1996), or that climate change will lead to 5 - 20% yield increases in North-American rain-fed agriculture (Field *et al.*, 2007)

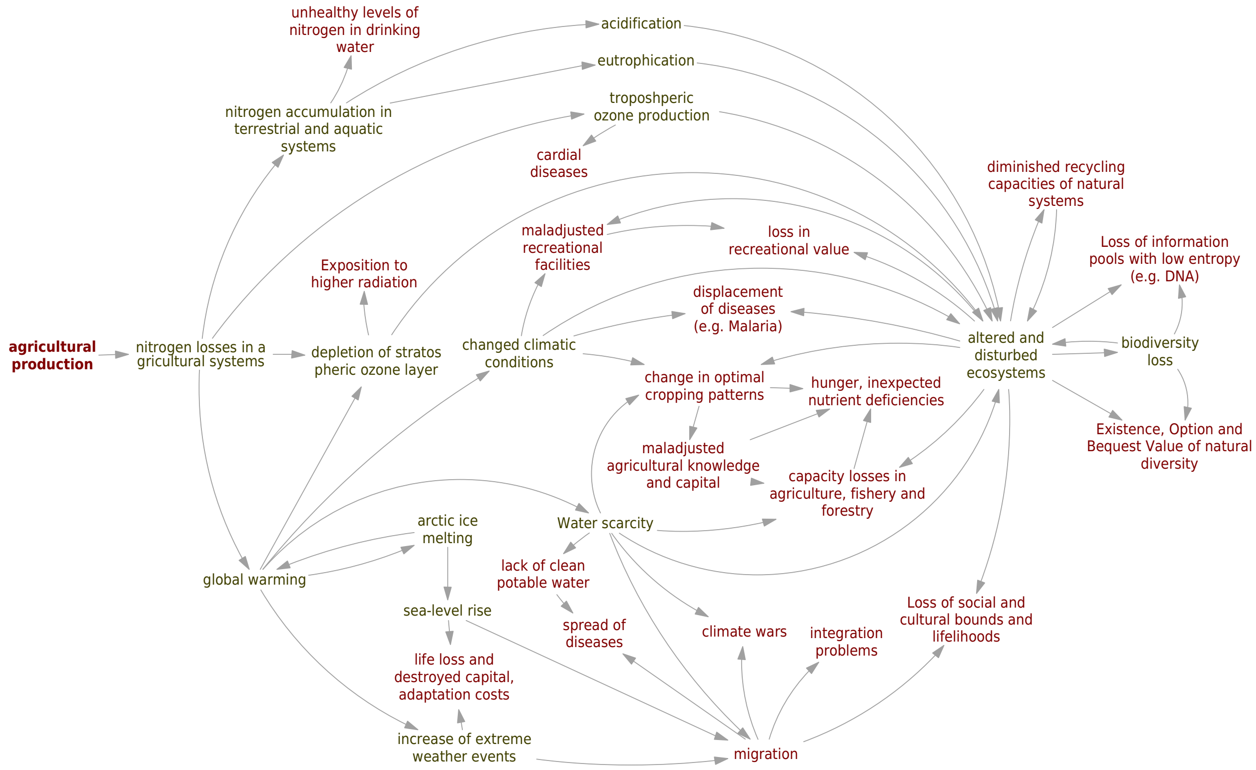

Figure 2.1: Major impacts of nitrogen enrichment(green) and their effects on human behavior and welfare(red). Compiled from Galloway *et al.* (2003), Vitousek *et al.* (1997), and Richardson *et al.* (2009)

2.2 Balancing the Interests

In the last section, two opposing interests were displayed: the interest of the farmer who requires nitrogen inputs for agricultural production is interwoven with the interest of third parties who suffer from the consequences of nitrogen pollution. The two interests stand in a trade-off relation, which means that furthering one interest will constrain the fulfillment of the other. The question now arises what an appropriate balance between these two interests could be. The aim is to find an optimal solution from a social welfare point of view.
To make the two interests comparable, a valuation of both utility to the farmer and damages to third parties is required and included into a social welfare function. The first subsection will explain the problematic of economic valuation and of setting up an 'objective' social welfare function. Subsection 2.2.2 will analyze which pollution level will occur under the current market framework. It will become clear that present markets are unable to deliver an outcome which is optimal under a social welfare perspective, mainly due to the existence of external effects. Subsection 2.2.3 will investigate why markets cannot deliver this optimal outcome, before the next section 2.3 will take a look at the policy options available to improve the situation.

2.2.1 Valuation approaches and social welfare functions

To make decisions in trade-off situations, a valuation of all options is required. This is also the case for the trade-off between the welfare and the damages to third parties. While every individual bases everyday personal decisions on subjective valuations, it remains a hotly debated subject within the economic community what the adequate way of 'objective' valuation is and how social welfare can be derived from individual welfare.
The predominant approach among economists is to valuate goods and services according to their market prices. This method has the advantage that price data is widely available and easily accessible for many goods. Yet this method also entails a number of theoretical and practical shortcomings:
Market prices can only depict exchange values, which reflect(assuming competitive markets) the marginal benefit of the consumer who owns the good. Yet, who the current owner is (and whose marginal benefits of consumption determine the price) is based on an historically specific distribution of resources (Downing *et al.*, 2001b). Marginal benefits of consumption may to a great extent between individuals. A rich person might value an additional cubic meter of water for his golf place with an higher pecuniary value than a poor person values an additional liter of drinking water. When damages are measured according to market prices, the damage costs occurring on a golf place might easily outweigh the damage costs occurring to a poor village in a developing country without proper access to drinking water. It is questionable which insights such a valuation method may allow policy makers if they do not agree with the current distribution of resources (Endres & Holm-Müller, 1998). Comparing damages solely on the basis of market prices is thus highly problematic, especially in an international context. Yet market price evaluation might be an appropriate tool for judging trade-off situations between individuals with a resource distribution which is accepted by all individuals concerned, for example within communities with strong social coherence.

Furthermore, market prices only reflect the marginal benefits of consumption as long as no market failures occur. Yet, market failures are almost omnipresent Stiglitz & Greenwald (1987). Market failures thus bias market price valuations substantially. For example, a monopoly might lead to price-setting above equilibrium, and to ostensible higher welfare losses when the service is delimited. For certain goods, no market prices exist at all, because no markets exist; according to market valuation, they would thus have a price of zero.
Yet, certain methods have been developed to estimate 'unbiased' market prices even where no markets exist. They can be estimated indirectly, via the analysis of managerial parameters, analysis of abatement cost expenditures, the transport cost method or the hedonic pricing method; and they can be estimated directly, e.g. by the contingent valuation method or reference markets or market simulation (for a detailed description of these methods see Endres & Holm-Müller (1998) and Perman *et al.* (2003)). Yet there is still a strong concern regarding the these methods' practicability and usability of results (Downing *et al.*, 2001b).
As nitrogen pollution is a stock pollution, negative effects of current emissions will continue into the future. The damages occurring in the future thus have to be included into today's considerations, which creates certain controversies: First, as market prices are not stable, future price developments have to be considered for market price valuations. These in turn depend on the preferences of future generations, which are difficult to foresee. Second, future damages might be discounted with a social discount rate, as current resource application might lead to capital stocks for future generations. Also here, there are different opinions about the appropriate height of these social discount rates (Toth, 2000).

Alternative valuation methods, like the consideration of the use value instead of the exchange value, fail mainly because utility cannot be measured cardinally and because utility cannot be compared inter-personally (Arrow, 1988; Endres & Holm-Müller, 1998). Still, some approaches were developed to estimate and compare well-being. For example, the physical quality of life index (Morris, 1979) or the human development index of the UN use a large number of indicators like life-expectancy, education or wealth to rate welfare interpersonally and across nations. These approaches cannot be considered objective as they are based on subjective weighting of different parameters; still, they may help to improve or correct valuations derived from market-prices (Endres & Holm-Müller, 1998).
Economic valuation remains thus an subjective discipline which is based on assumptions about human preferences. Yet, even if objective values of individual utility could be determined, it is an open question how individual welfare can be aggregated in *'social welfare'*, which determines the optimal distribution between individuals. Approaches vary widely: The *'utilitarian approach'* wishes to maximize total welfare as a sum of individual utility. This results in equal marginal utilities between individuals, but may lead to heterogeneous levels of utility. An utilitarian society would give a resource to the person who makes the most out of it, even if he was already well-off. In contrast, the *'egalitarian approach'* wishes to maximize the utility of the worst-off member of society. This leads to lower heterogeneity in individual utility, but to diverging marginal utilities. Further social welfare functions can be found in Petersen (1996). Yet none of these social welfare functions can be considered objective in a scientific

sense.
As science cannot deliver 'objectively' an economic valuation approach or a social welfare function, many economists stick to a kind of minimum-consensus of welfare economics, namely 'pareto-efficiency': a situation can be identified as being suboptimal from a social welfare perspective, if one individual can me made better off without decreasing the utility of others. This approach requires ordinal measurement of utility and no inter-personal comparison and can thus be considered scientifically 'objective'. Yet, pareto-efficiency is only a necessary but no sufficient condition for optimality. Stiglitz (2001) remarks that this approach allows economists to concentrate on efficiency issues, while masking distributive consequences and giving the false impression of being unbiased.[6]
Thus, even though it can be shown that pollution leads to a pareto-inefficient outcome (see for example (Buchanan & Stubblebine, 1962)), for the following theoretical considerations an approach shall be used which intentionally allows for social welfare to rise even in the case that an individual is worse-off than in the original situation. A simple utilitarian function Ω shall be assumed, which is composed of the utility of the farmer U_f minus the damage costs D occurring to the aggrieved parties. The damage cost function may be defined politically and shall represent opinion the social benefit losses from other agents through pollution, which may also include distributive considerations.
After defining what an social optimal outcome shall be, the next section shall make visible that markets are unable to provide this outcome.

2.2.2 Market Failure

Currently, the real existing mal-functioning markets often do not reach an profitable outcome for the farmer, not to mention an optimal outcome from a social welfare perspective. For example, Matson et al (Matson *et al.*, 1996) showed in a study analyzing two sugar cane plantations in Hawaii that the farm which used more knowledge-intensive farming techniques spent one third less fertilizer, had 10-fold lower emissions and was at the same time more profitable. There are numerous reasons that may lead to excessive nitrogen application. To name a few:

- Malfunctioning capital markets impede farmers to use efficient but capital-intensive technologies like precision farming.[7] Origins may lie in asymmet-

[6]Pareto-efficiency can be valid for a multitude of situations with totally differing distributional outcomes. In theory, every pareto-efficient distribution could be reached by supplemental lump-sum transfers. Yet its realization would require complete information and the absence of transaction costs. Thus, economists often concentrate only on the aim to create a pareto-efficient situation. Yet, the change from a pareto-inefficient situation to a pareto-efficient situation is by no means necessarily a pareto-efficient transition itself: a tax on pollution might for example fully internalize external effects and create a pareto-efficient situation; but the taxed polluter will be still worse off than in the situation before the tax was introduced. Compensation payments from the tax-income to the polluter are hardly ever realized. The advocacy of pareto-efficiency has thus the claim to reach a situation in which nobody looses, although this pretense is seldom fullfilled

[7]Precision farming is a farming technique which takes into account intra-field and intra-time variations of soils, weather-conditions, nutrients and crops, and adapt farming techniques respectively. Data from satellites, sensors and information management tools is used to estimate optimal fertilizer input and timing. This technology reduces fertilizer losses substantially, but requires high capital investments for data collection and precision-management machinery. Batte & VanBuren (1999)

ric information and adverse selection (Stiglitz, 2001) which lead to credit rationing and interest rates below market clearing.

- The dilemma of public goods applies to fertilization technology: determining the optimal fertilization quantity, timing and technique requires field studies, yet the results can be easily copied by other farmers. Thus the full benefit of producing this information cannot be appropriated and the information is therefore not produced in a sufficient amount by private actors (Perman *et al.*, 2003).

- The environment as a sink for wastes can be seen as a common good for all farmers which is exploited too fast, because no single agent can appropriate the opportunity costs of exploiting it in the following period. Similarly, the private rate of time preference may diverge from the social rate of time preference, resulting in a faster exploitation of a natural resource than would be socially optimal.

- Government failure may also be a reason for excessive pollution: fertilizer subsidies with the aim of fostering agricultural development lead to excess fertilizer application (Bumb & Baanante, 1996). Subsidies for animal products can be judged in a similar way, as they increase the substitution of vegetarian food producing low emissions with meat products, leading to much higher emissions.

Even though these dysfunctions are important, it would exceed the scope of this work to analyze them in more detail, among others because every single market failure requires a specific policy response.
Instead, this work shall focus only on external effects, which are probably the single most important market dysfunction in this context.
A (non-pecuniary) external effect can be defined as an interdependency of an agents utility function and the behavior of another agent that is external to the price system.[8] In our case, the pollution caused by the nitrogen application of the farmer decreases the utility of other individuals. It shall be explained in the following by a simple economic model, why the *'invisible hand'* of the market is unable to fix a level of pollution which is optimal from a social welfare perspective.
To keep the model simple, lets assume a world with only one price-taking[9] farmer and one individual who is affected by the pollution. Let the farmer use a combination of production inputs to produce a set of outputs. Let X be a positive m-vector $(X_1, \dots, X_m)$ that represents the inputs and management practices of the farmer.
The output of the farmer is defined by the positive m-vector Y. The quantity and combination of the applied input factors determine the output $Y = Y(X)$. Assuming diminishing returns on scale, this production function is concave and assumed to be twice differentiable. The farmer can buy inputs and sell outputs according to the positive price p which is again a m-vector.
It shall be simplified that the farmer's utility $U_f(\Pi(X), \dots)$ depends only on

[8] see (Buchanan & Stubblebine, 1962) for a more formal definition

[9] Price-taking means that the farmer has no market power and has to accept the market price as given.

his profits $\Pi(X)$. If the farmer tries to maximize his profit $\Pi(X)$, he is faced with the following optimization problem:

$$\max_X \Pi(X) = \max_X (Y(X) \cdot p - X \cdot p) \tag{2.1}$$

Profit maximization requires that the farmer employs input factors up to the moment when the marginal revenue from an input j equals its price $p_j = \frac{\delta Y}{\delta X_j} \cdot p$. Due to the diminishing returns on scale he will apply more nitrogen for low fertilizer prices, and less nitrogen for high fertilizer prices.

Now, we shall turn to the perspective of a social planner who tries to maximize social welfare. The social welfare function is a function of farmers profits Π and damages D. The damages depend on the magnitude of pollution E, which in turn depends on the farmer's choice of inputs and management practices X. It is assumed that the damages cost function is convex to X, assuming increasing impact of pollution. Hence

$$\Omega = \Pi - D(E(X)) \tag{2.2}$$

The production function of the farmer and the social damage costs are now nonindependent over X. Actions undertaken by farmers influence not only their own utility, but via the environmental damages also the utility of others. It is a classical example of a non-pecuniary external effect.
When the social welfare shall be maximized

$$\max_X \Omega = \max_X (\Pi - D) = \max_X (Y(X) \cdot p - X \cdot p - D(E(X))) \tag{2.3}$$

the optimality requirement $\frac{\delta Y}{\delta X_j} \cdot p = p_j - \frac{\delta D}{\delta X_j}$ diverges from the profit maximization requirement of the farmer. Because $\frac{\delta D}{\delta X_j}$ is positive, the marginal benefit of additional nitrogen application is lower for the social planner than for the farmer. Due to the diminished returns on scale (concavity of the production function) the social planner will use less polluting inputs than the farmer.
Yet, under the current market framework the farmer only considers his private costs and leaves the social costs of his behavior out of consideration. He will use access fertilizer to minimize the risk of nutrient deficiency, deploy manure less often to save work time, neglect soil erosion, grow plants which are less able to absorb nutrients from the soil or undertake other non-sustainable practices. The behavior of the farmer thus prevents the market to deliver an efficient allocation which is par to the social benefit achieved under a social planner.
This situation is illustrated in Figure 2.2.2: The social marginal costs of nitrogen application are the sum of the private marginal costs of buying nitrogen fertilizer and the damage costs. The optimality condition states that marginal costs and marginal benefits of nitrogen application are equal. The social optimum thus lies at O^*, while the private optimum lies at O°. The private quantity of nitrogen application thus lies above the socially desirable level.
Insufficient internalisation will not only cause inaccurate allocation of input factors at one point in time, but also lower incentives for investments into the mitigation of pollution. The social return on investment in efficient production machinery or precision farming technology is far higher than the privately appropriable profits. Thus, both static and dynamic efficiency lack behind their optimal potential.

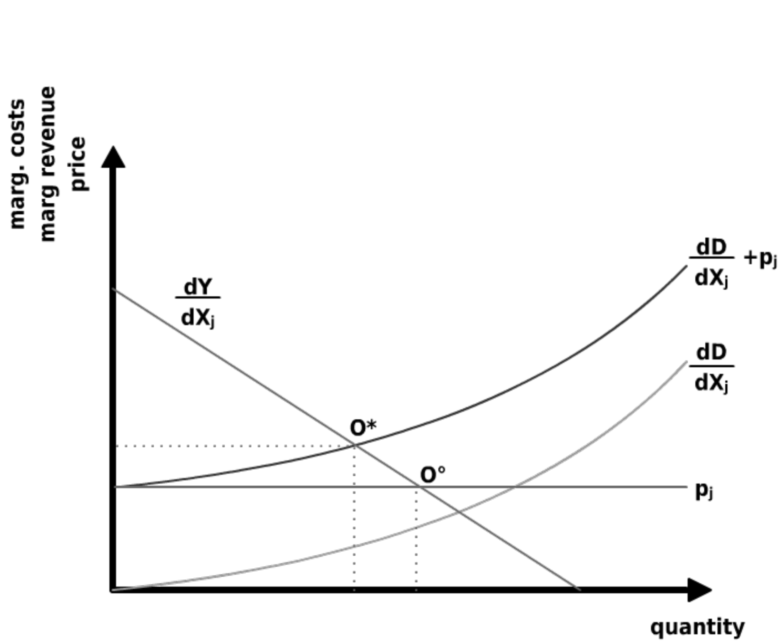

Figure 2.2: Because the farmer does not have to bear the entire costs of his agricultural activity, he has an incentive to extend his agricultural activity beyond the social optimum O^*, leading to excess pollution and a too low market clearing price in the private optimum O°.

2.2.3 Origins of Market Failure

In the last paragraphs it became clear that the current markets are unable to handle the problem of nitrogen externalities. This section shall tackle the question why the market for nitrogen is defective. More precisely, it is not the market for nitrogen, which is defective, but it is the market for the by-products of nitrogen application which fails.

Imagine a functioning market for the by-products with well defined property rights – either pollution rights or rights to an unpolluted environment. In this fictitious scenario, the polluting and the aggrieved parties can conduct negotiations over the level of pollution. If the aggrieved party possesses rights to an unpolluted environment, it can forbid pollution or demand compensation for the environmental damage from the polluter. Also, if the polluting party possesses pollution rights, it can demand compensation for its lost profits when production is curbed. Coase (1960) stated that independent of the division of pollution rights between polluting and aggrieved parties, the introduction of property rights and the subsequent negotiations will lead to a pareto-optimal outcome with the same, optimal level of pollution.[10] The question remains why such a market does not come about in reality.

Tietenberg (Tietenberg, 2002) points out four conditions which are essential to

[10] Yet, this result was incomplete. In the long-run, it is of significant importance, whether the pollution rights are assigned to the polluter or to the aggrieved party: the sale of his pollution rights offers a new income source and increases the firms profits. Firms, whose average costs would otherwise exceed the profits will enter the market if they can sell their pollution rights. Also, it provides dynamic incentives to increase the polluting activity, because the aggrieved parties thus have to increase their compensation payments to the polluter (Taschini, 2009).

an efficient property structure: universality, exclusivity, transferability and protection.

- Universality means that all property rights are private and well-defined. In the case of externalities, one could argue that this is the case: as farmers are not prosecuted for their nitrous oxide emissions, they can emit as much as they want and have the right to pollute. But as no limit to pollution exists, no right for a certain amount of emissions exists and pollution rights can hardly be said to be well-defined. As property rights are not scarce, no market price will develop: when one farmer agrees to curb his production and sells his emissions, another farmer may simply extend his production indefinitely and the level of emissions can not be controlled.
  Also, well-defined property rights might be difficult to achieve because the extent of pollution is largely unknown. As mentioned above, the processes of pollution are long-term, complex, multifaceted and depend on local conditions. If pollution has to be quantified, this can be connected to high measuring costs.

- Exclusivity means that the costs and benefits of a pollution right will exclusively concern the owner of the right. This criteria is also not fulfilled: the damages that the farmer's activities create have to be borne by others. Also, when someone who is affected decides to pay the farmer to reduce, pollution he cannot appropriate the full benefits of this action. As nobody can be excluded from changes in the world climate, greenhouse gas mitigation brings benefits to everybody even when it is only undertaken by single individuals. This leads to a classical prisoner's dilemma, in which free-riding is the dominant strategy: individuals who compensate farmers for low-emission practices are unable to appropriate the whole benefits of their action and only experience a marginal improvement of their situation. If they maximize their personal utility, they will therefore pay lower compensations or even no compensation at all. In the ideal case, the compensations should reach the point at which the sum of marginal benefits across all affected parties equals the marginal opportunity costs of low-emission practices; yet this level will not be reached.

- Transferability means that property rights can be transferred from one actor to another, and that this transfer is based on a free decision of both actors. This characteristic can also be seen as critical for nitrogen pollution. The problem is, that polluters and aggrieved parties do not live necessarily in the same time period. Some of the humans affected by climate change are not even born yet. This time-lag leads to a situation where it is impossible that both negotiators agree to a deal at the same time. This situation is of special subtlety, because polluters have to decide now whether they should mitigate pollution or not, and future generations can not be excluded from the decisions made in the past.
  If there are members of the same generation who want to trade, transferability might still fail due to negotiation costs and contract costs. These are of special concern if traded quantities are small, and when the process of trading is unusual and unstandardized.

- Protection of property rights means that property rights are accepted by all parties concerned and can be enforced by the owner. This may raise problems in countries where no consolidated legal system exists. Protection of property rights also requires monitoring and controlling eventual infringements. Thus, at least for certain spot controls, costs for measuring emissions occur again.

As can be seen, all criteria of Tietenberg for an efficient property structure can be seen as problematic or even fail completely. This explains very well why it is so difficult to establish a market for pollution in this case.

2.3 State Intervention

As there exists a substantial market failure to the disadvantage of present and future generations, one might argue that the state should defend the interests of the aggrieved parties. However, market failure is no sufficient precondition for state intervention. An evaluation of different policy instruments has to be done to see whether state intervention can actually improve the situation and which kind of actions would be suited best. Policy instruments discussed in the scientific debate comprise directives, taxes, subsidies, tradable permits, moral suasion and socialization. This thesis will only debate the first four policy options. Socialisation is such a strong policy intervention, that it can probably not be justified just by the existence of external effects; moral suasion is an instrument which tackles the preference structure rather than the incentive structure of an agent, and it is thus difficult to compare to other policy instruments. A short discussion of both policies can be found separately in Annex .
The first subsection will present the four remaining instruments, and explain which approach is taken to internalize the external effect. Yet, efficient internalization is often not the only criteria of policy makers when choosing their agenda. A second point of major importance are the costs of implementing and administrating this policy (e.g. for information gathering, controlling and re-evaluating). As nitrogen pollution is a non-point source emission, these policy implementation costs are especially high and thus of special importance. A third important issue are the distributive effects of a policy: there might be a trade-off between reaching a sustainable level of pollution and equity-issues. Because food production satisfies one of the most central needs of humanity, this issue is of vital importance when considering the effects of nitrogen regulations. Finally, the policy intervention might have unpredictable outcomes on both distributive and environmental parameters. The fourth subsection will thus discuss the implications of considering uncertainties for the aim of an accurate policy.
This work will not discuss whether the proposed policy measures are actually enforceable under the current constellation of political power. The question whether a policy measure is beneficial or not should certainly be answered prior to the discussion how it could be implemented. This work moreover helps further research to clarify who is affected in which way by the policy discussed and to identify, who thus may have an interest in taking part in the political processes.
In the following, the words 'regulation' and 'regulating' shall be used as synonyms for 'policy instrument' and 'policy instrument

2.3.1 Policy Options and Internalisation

Policies to combat external effects usually have the aim to reach a behavior of all agents which is equal to their behavior when maximizing social welfare. In the following, the most important policy instruments shall be presented and the way they internalize the external effect shall be explained using a simple model. Finally, we will give some examples of realized implementations of the instrument.

Directives

Directives, as all other subsequent policies, do not intend to change the preferences of an agent, but limit their action space. They prescribe or forbid certain production inputs and management activities. Directives seem to be a simple possibility to combat external effects: when farmers have no choice as regards the level of polluting activities, they cannot exceed the desirable amount in their pursuit of profit.

Returning again to our model, let $e_1, \ldots, e_r$ be an enumeration of all polluting entries of X, and let $n_1, \ldots, n_r$ be an enumeration of all non-polluting entries of X. The regulating authority now fixes the level of all polluting inputs $X_{e_1,\ldots,e_r}$, such that the farmer can only optimize his profit over $X_{n_1,\ldots,n_r}$

$$\max_{X_{n1},\ldots,X_{nr}} \Pi = \max_{X_{n_1},\ldots,X_{n_r}} Y(X) \cdot p - X \cdot p \tag{2.4}$$

The regulation authority should choose $X_{e_1,\ldots,e_r}$ according to the first derivative of Ω to X: $\frac{dY}{dX_j} \cdot p = p_j + \frac{dD}{dX_j}$ should be fulfilled to reach efficient allocation.

Directives were often used against nitrogen pollution in the past, and are probably still the main environmental instrument in this field so far. An example is the German "Düngeverordnung",[11] which among other things regulates when and how fertilizer can be applied to agricultural lands, defines technical standards for agricultural machines, demands farmers to take soil samples for nutrient measurement and requests them to calculate nitrogen balances.

Taxation

It was A.C. Pigou, who first introduced the idea to internalize external effects via a tax (Pigou, 1960; Laffont, 1972). In recent years, environmental taxes were also analyzed for the context of nitrogen pollution (Xiang *et al.*, 2007). The idea behind a nitrogen tax is to increase the price of pollution or pollution-related inputs and management practices. This serves as incentive to lessen polluting activities and to substitute polluting inputs and practices with other non-polluting activities, that are not taxed and thus cheaper. The impacts of an input tax on production and pollution can be illustrated by returning to our simplified model: suppose the government introduces a tax that taxes inputs X with a tax scheme $T_+(X)$ being a m-vector. The farmer's profit function becomes now

$$\max_X \Pi = \max_X Y(X) \cdot p - X \cdot p - T(X_{e_1}, \ldots, X_{e_r}) \tag{2.5}$$

[11] The full name of the DüngeV is "Verordnung über die Anwendung von Düngemitteln, Bodenhilfsstoffen, Kultursubstraten und Pflanzenhilfsmitteln nach den Grundsätzen der guten fachlichen Praxis beim Düngen"

The regulation is efficient and restores the social benefit Ω of a social planner, if the regulating authority holds to the rule $\frac{dT}{dX_j} = \frac{D}{dX_j}$.
Examples for input taxation can be found in Sweden, where nitrogen fertilizer is taxed with 0.198 cent per kg. Denmark also has a tax scheme on fertilizers, while Austria abolished its fertilizer tax in 1994. Finland only had a fertilizer tax for 2 years between 1992 and 1994 (Pearce & Koundouri, 2003; Radulescu, 2004).

Subsidies

Subsidies are rather similar to taxes. They also try to influence the incentives for individuals without making hard constraints. They either pay for activities which decrease the level of pollution, or they pay for a reduction of an activity which is polluting.
In our model, a subsidy for the reduction of a certain activity requires a baseline m-vector B. Subsidy payments S_+ are payed according to the difference between applied inputs and the baseline, such that the profit function of the farmer becomes

$$\max_X \Pi = \max_X Y(X) \cdot p - X \cdot p + S(B - X) \tag{2.6}$$

For efficient regulation, the regulator should set the baseline to the level of X without state intervention. The subsidy scheme should be designed according to the rule $\frac{dS}{dX_j} = -\frac{dD}{dX_j}$.
Agricultural regulation in the United Kingdom is mainly based on subsidies, as the right to pollute is attributed to the farmer (Hanley, 2001). An example is the 'nitrate sensitive area scheme', which was introduced in 1990 to limit nitrate in drinking water. Farmer can agree to restrictions on their farming practices like abandoning arable land, or low nitrogen cropping with less than 150 kg Nr-inputs per year.

Tradable Permits

The Cap and Trade System was developed by the Canadian economist J.H. Dales (Dales, 1968) and can be considered as an advancement of the Coase Theorem. It creates a market for emission certificates which assigns a right to the holder to emit a certain amount of pollution. The state issues a fixed amount of emission certificates. Polluters now have to figure out whether it is cheaper to pollute and buy a certificate, to apply mitigation strategies or to stop production. Firms, which are able to mitigate more emissions than they produce can sell their saved emissions on the market. The price of the emission certificates will now balance out demand and supply for emission certificates, and will rise to prohibitive levels, when the emission cap is reached. Thus, tradable permits allow to reach a precise pollution level that can be set by the policy makers.[12]

[12]The cap and trade system can have flexible characteristics. The certificate supply can be designed elastic to picture the utility function of future generations more adequate. Also upper price frontiers can be set to prevent excessive price advances. Another possible option is to allow emission banking, which means that unused emission rights can be banked and kept for the future. There are also different layouts concerning the original distribution of emissions rights. *'Grandfathering'* describes a distribution of pollution rights in continuation of former pollution. Rights can also be distributed according to other political aims or simply

Certificates which are emitted by the state can either be auctioned in favor of the treasury or distributed to citizens according to political interests. One practice which may increase acceptance among regulated agents is to *'grandfather'* the emissions according to past pollution levels. Yet this practice has been highly criticized in the literature(see e.g. (Cramton & Kerr, 2002)) for being unjust, foregoing the steering effect and causing windfall profits for large polluters.
Returning to our model, the farmer's profit function receives a new component $\widehat{E}(X) \cdot p_z$. While p_z stands for the price of a certificate, $\widehat{E}(X)$ stands for the estimated emissions by the regulation authority of a certain input combination X. The farmer is now confronted with the profit function

$$\max_X \Pi = \max_X Y(X) \cdot p - X \cdot p - \widehat{E}(X) \cdot p_z \tag{2.7}$$

For effective regulation, the state has to limit the number of certificates, such that the scarcity price p_z of the certificates comes to a level which guarantees that $\frac{\widehat{E}(X)}{X_j} \cdot p_z = \frac{dD}{dX_j}$.
The Cap and Trade System is the core instrument of todays GHG mitigation policy. The Kyoto Protocol also includes agricultural emissions, but trade does only take place on state level. How each state realizes its mitigation share internally is not further defined by the protocol. In 2005, the European Union created an GHG emission trading system in their 'directive 2003/87/EC'. The emission trading scheme was devolved to a firm level, but this trading scheme up to now excludes agricultural emissions.

2.3.2 Policy Implementation Costs

Up to now, a fictitious world was assumed where the regulation authority has complete information and can realize a policy without costs. These assumptions are not realistic: the regulating authority has incomplete information, and gathering information is cost-intensive. It includes the costs for choosing a policy, implementing and administrating it and finally for controlling compliance and fining infringements. Costs occur both on the regulator side (e.g. estimating the damage cost function and administrating the tax system) and on the farmer side (e.g. collecting the relevant data and delivering it to the agency). Policy makers have to take these costs into account when choosing their favorite policy. While all policy options mentioned above can perfectly internalize the external effect under complete information, information costs diverge substantially.

Up to now, a fictitious world was assumed where the regulation authority has complete information and can realize a policy without costs. This section will alter this condition.
The section is divided into three sections: The first part will have a look at the regulation base; it becomes clear, that nitrogen-pollution can only be based on a regulation of inputs and farming activities. Then, it shall be analyzed why the optimal base cannot be a comprehensive one, which regulates all inputs and farming activities which are pollution-relevant. Finally we will have a look, how information assymetries between farmers and regulation authority may alter the

proportional to the population. Another option that also raises revenue for the state, is the auction of certificates.

advantages of different policy instruments.
A regulation which tries to influence nitrogen-pollution has to take into account the particularities of nitrogen-pollution. It is a non-point pollution, which occurs arbitrarily in space and time. While direct measurement would have prohibitive costs (Griffin & Bromley, 1982), pollution can be approximated statistically on the basis of sample measurements (Xiang *et al.*, 2007), and attributed to inputs and management factors which create the pollution. Griffin & Bromley (1982) explain, that agencies can also regulate efficiently via taxes, subsidies and regulations on inputs and management practices, if they dispose over the relevant information about emission factors and input application.
Yet, gathering information about site-specific emission factors, and farmer-specific application of inputs and farming practices certainly generate costs. They comprise costs for legislation, dissemination of laws and taxes, measurement costs of regulated items, control costs and administrative costs for both firms and central-planners. It can be assumed, that implementation costs rise with the number of regulated items and regulated agents.
As mentioned above, there is a large number of management practices and inputs which influence the pollution on a farm, starting from various industrial fertilizers, livestock production and manure application, growing of legumes and cover-crops, but also including how they are applied, how soils are treated and which moisture conditions exist. Furthermore, the number of farmers who had to be regulated is excessive(USDA, 2009b; eurostat, 2009).
As Smith and Tomasi indicate, regulators should keep in mind these policy implementation costs when choosing the type of policy intervention. The scheme should optimize social benefit under consideration of regulation costs. For a tax scheme, this would be

$$\max_t \Omega = \max_t Y(X) \cdot p - X \cdot p - D(E(X)) - C(t) \tag{2.8}$$

under the condition of private profit maximization

$$\frac{dY}{dX} = p + \frac{dT}{dX} \tag{2.9}$$

with C being the implementation costs that depend on the taxed inputs t.
When it is taken into consideration, that costs rise with the number of taxed items, it is obvious that for the optimal tax base $\frac{d\widehat{T}}{dX_j} < \frac{dD}{dX_j}$, the level of internalization is lower than in a case without regulation costs.

The intricacy of information costs rises, when one takes into consideration, that the regulating authority may not possess certain informations, which the farmer possesses. For example, it is widely accepted in the literature that firms know their own production function better than the regulating authority and are thus able to choose a more profitable production technology (see literature in Shortle & Dunn (1986)). Hence, there are assymetric informations between farmers and regulating authorities.
Regulation could be much cheaper, if the central authority could use this information instead of collecting it itself. Unfortunately, farmers may use their information to forward their own instead of the social interest, and will give no or even wrong information the the authority. Shortle & Dunn (1986) show that incentive based instruments like taxes and subsidies are better suited for asymmetric information, as they manage to use indirectly the knowledge of the

farmer on his production function.
Suppose that the production function of a farmer $Y = Y(X, k)$ features a component k, which corresponds to the specialized knowledge of the farmer of his production function. While the farmer knows the precise value og k, the regulation authority can only guess the density function $w(k)$ of k.
The total utility function of the social planner would thus become

$$\Omega = Y(X, k) \cdot p - X \cdot p - D(E(X)) \tag{2.10}$$

The central authority which sets a tax, certificate or subsidy scheme so that $\frac{dT}{dX_j} = \frac{dD}{dX_j}$ can still reach the optimal solution and conserve the incentives for the farmer to choose the optimal mix of input factors, where $\frac{dY}{dX_j} \cdot p = p_j + \frac{dD}{dX}$, even not knowing k.
This is different in the case of directives. If the regulation authority commands certain entries of the vector X, it cannot make use of the farmers knowledge about k. The agency can only guess the density function $w(k)$ of k. It then has to maximize the term

$$\Omega = \int \Big(Y(X, k) \cdot p - X \cdot p - D(E(X))\Big) \cdot w(k)\, dk \tag{2.11}$$

The regulation authority should choose $X_{e_1,\ldots,e_r}$ according to the first derivative of Ω to X: The equation

$$\frac{d \int Y(X, k) \cdot p\, dk}{dX_j} = p_j + \frac{dD}{dX_j} \tag{2.12}$$

should be fulfilled to reach the most efficient allocation which is possible with directives. Yet, in any case where the best-guess according to the density function does not match with the real value, the efficiency of directives lacks behind the efficiency of incentive-based schemes like taxes, subsidies or certificates.

2.3.3 Distributive Effects and Price Vulnerability

Up to the moment, only questions concerning an efficient attainment of the optimal externality level were considered. Yet, as mentioned above, internalization can lead to numerous outcomes with differing distributional effects, depending on how an external effect is internalized. It shall thus be investigated which distributive effects a certain policy has, and who will be positively or negatively affected by a particular state intervention compared to the status-quo.
One may distinguish in this context between five affected parties, which partly overlap: agricultural producers, consumers, taxpayers and people who are affected by climate change. A tax on agricultural inputs will have the following effects:
The taxpayer of a nitrogen tax is either the farmer or the input producer. If the possibility is available, such taxpayers will aim to circumvent the tax by mitigation practices or the substitution of polluting inputs against less polluting ones. Nevertheless production cost increases for farmers are unavoidable, and farmers will not profit from the tax. Some farmers will be more affected than others: in particular producers of taxed inputs like manure, farmers with high nitrogen input requirements and producers of superior food products like meat

will be burdened by a tax. But as food demand is relatively inelastic to price changes (USDA, 2009a), farmers can shift a lot of their costs to consumers and will not have to curb their production significantly.
As the tax can be shifted to consumers, food prices are likely to rise due to the introduction of a tax. As was already pointed out by the statistician Ernst Engel in the 19th century(Engel, 1857), food is a relative inferior good. Absolute expenditure for food is almost fixed, while food consumption as a share of total expenditure falls with rising income. In developing countries, food constitutes the major part of total expenditure (Huthakker, 1957; Mellor, 1978; Ravallion, 1990). Thus, food price increases are a relatively high burden for low-income households especially in developing countries, which will lead to increasing poverty, undernourishment and hunger (Food and Agriculture Association (FAO), 2008)(Ivanic & Martin, 2008).
The tax revenue from an environmental tax is often referred to as the *'Double Dividend'*, i.e. additionally to the improvement of resource allocation (Perman *et al.*, 2003, Chapter 6). Because the tax revenue can be used for other non-environmental purposes or for balancing out the distributive effects of the tax, environmental steering taxes may be considered a so-called **'no-regret'** option: even if the environmental benefits are in doubt, an environmental tax reform may be desirable (Bovenberg, 1999, p. 421). Yet, the more efficient the steering effect of the tax is, the lower the tax income. Thus, there is a certain trade-off between the two dividends.
People who will be affected by climate change will certainly profit from the tax. Again, it depends largely on the steering effect the tax can develop, how many emissions can be saved. If the tax burden is lower than the mitigation costs, the measure will remain without effect.
Subsidies, although using a similar internalisation principle as taxes, have a totally different distributional outcome. Subsidies offer a new, additional type of income to farmers. This may make some kind of production profitable that would not be lucrative otherwise. Thus farmers profit from subsidies and may expand production. Because of the inelastic demand, benefits will also be passed on to consumers via lower food prices. Thus subsidies will on the one hand relief the burden on low-income households. On the other hand, the treasury will have large expenditures that in turn have to be financed by the taxpayers. Windfall profits of ultimate subsidy recipients, who would have adopted a mitigation practice in any case, are unavoidable and costly to the tax-payer. Furthermore, payed subsidies will rise when farmers adapt to the desired steering effect and skim off the subsidies. An adjustment of the baseline may lower the fiscal burden, but may also decrease dynamic incentives if it is anticipated.
Finally, people affected by climate change will profit less from subsidies than from taxes: because subsidies lower food prices, increase food demand and extend farmers' profits, farmers will increase their production and stay longer on the market. As there are no production techniques without at least a small amount of nitrogen pollution, subsidies can only foster the substitution of polluting measures by less polluting measures. Thus it is not clear whether the production increase will outbalance the pollution reduction or not. In any case, the effects of subsidies on emission reduction will be less than the one of taxes (Kamien *et al.*, 1966).
The cap-and-trade system has similar effects as a tax. Depending on the magnitude of the emission aim, it can have exactly the same effects on farmers and

consumer income as a tax: when the scarcity-price of a certificate equals the tax rate, the tax burden on the farmer is the same for both schemes. Yet, the question of the original distribution of certificates alters the distributional outcome substantially: only in the case when certificates are auctioned by the state are outcomes equal to a tax. In contrast, grandfathering will leave the treasury without income and will not improve the consumers' situation compared to a tax: Cramton & Kerr (2002) diagnose for the energy market – true for the food market – that the price of certificates and thus the food price only depends on the price of mitigation; distribution of certificates to farmers will only create windfall profits and not affect food prices.

2.3.4 Accuracy

So far, it was not considered that policy makers can do mistakes when choosing the optimal policy. Given the substantial uncertainties and lack of information that constraints the work of the regulating authority, these mistakes can not be excluded. Thus, a policy should be designed in such a way that mistaken assumptions do not endanger the benefits of state intervention.

Usually, there are two aspects in particular where policy accuracy is of importance: the quantity of mitigated pollution, and the cost burden for ultimate tax payers. The trade-off between these two accuracies is a topic frequently debated by economists (Weitzman, 1974; Laffont, 1977; Taschini, 2009). Some policies can reach good accuracy concerning the pollution level. For example, the cap-and-trade system limits total pollution effectively and reaches this cap precisely. In this case, the instruments have however low accuracy concerning the burden for farmers and consumers, because the price level for pollution certificates is flexible and adjusts freely to limit pollution. The price level cannot be foreseen by the regulating authority if it does not know the production functions of the farmers.

Other policies can reach good accuracy concerning the burden for farmers and consumers. In the case of a pollution tax, an upper limit of the burden can well be estimated by multiplying current fertilizer application with the tax rate. Yet, because the production functions of the farmers are unknown to the regulation authority, the environmental steering effect of the tax remains uncertain.

Finally, there are also policy instruments which are neither accurate concerning the mitigated emissions, nor concerning the burden on ultimate tax payers: input and management directives neither allow for an accurate estimation of opportunity costs of the forbidden action; nor do they allow for an accurate forecast of the directive's steering effect.

Policy makers thus have the choice whether to prefer a regulation which accurately meets an emission target or whether to choose a regulation safeguarding food prices. Weitzman (1974) now argues that it depends on the slopes of the mitigation cost curve and the damage cost curve whether a quantity or a price policy should be favored. If the damages accrued as a result of missed pollution level are larger than the costs of a wrong level of mitigation, then a quantity instrument should be favored. Otherwise, a price instrument is better suited to a case where failing the right pollution level is of less importance than restricting the burden on farmers and consumers.

As Pizer (1997) explains, marginal damage costs of climate change are rather flat compared to the mitigation costs. This is probably especially true for the agricultural sector: excessive price increases of agricultural products may cause severe and irreparable harm due to hunger and poverty, even if they only occur for short periods. Compared to that, the effects of climate change depend on the accumulated emissions of past periods, so that policy parameters do not have to be accurate from the first point in time, but can be adjusted in the long-run on the basis of observations by a trial-and-error process. Thus, a tax would be more appropriate instrument than emission-trading or directives.

2.4 Policy Efficacy Evaluation

As became obvious in the last section, the main challenges for effective regulation are

- to internalize the external effect (subsection 2.3.1),
- to allow for an efficient allocation of inputs and to preserve incentives for technical progress (subsection 2.3.1),
- to lower implementation costs, e.g. by keeping down the number of regulated parameters and agents (subsection 2.3.2),
- to choose a policy with high accuracy (subsection 2.3.4),
- and to minimize unfavorable distributive effects (subsection 2.3.3),

None of the policies discussed in the last section offers a first-best solution to all these challenges.
Directives hold only a small potential for efficient regulation. It was shown that incentive instruments can preserve an efficient allocation of inputs much better than hard directives. Yet, for certain cases policy implementation costs might be lower, as no fiscal administration is required.
Taxation, subsidies and tradable permits are more promising policy instruments. In theory, they could fully internalize the external effect and still preserve an efficient allocation by regulating all emission-relevant inputs. Yet, with growing number of regulated parameters and agents, policy implementation costs also rise; so there is a trade-off between the targets of efficient allocation, full internalization and policy implementation costs, which can not be reached all at the same time. Furthermore, the target of full internalization also stands in a trade-off relation to the target of favorable distributive effects: higher internalization will either lead to higher food prices (taxes and certificates) or to a higher burden of state budgets (subsidies). Finally policy accuracy also has a trade-off between reaching good accuracy concerning emission targets (certificates) and reaching good accuracy concerning distributive effects (taxes).
It became clear, that no first-best solution is available, as there are certain trade-offs between policy implementation costs and full internalization, between internalization and distributional outcomes, and between emission and distributive accuracy. A second-best solution has to be found that balances out all these aims.

2.4.1 Industrial Fertilizer Taxation - Simple and effective?

A promising second-best approach might be to tax the production of industrial nitrogen fertilizer. The assessment base of this tax would be the quantity of nitrogen fixed by Haber-Bosch synthesis. The tax payer would be the nitrogen producing factory.

Up-stream taxation of inputs, which is executed not on farm-level but on the level of fertilizer production plants, can significantly lower policy implementation costs: First, instead of taxing a mere endless number of emission-relevant inputs and management practices according to their site-specific emission factors, only the single most important input is taxed. Second, instead of taxing millions of farmers, only some dozens of fertilizer plants have to be taxed. There are 2.2 million farmers only in the US (USDA, 2009b); in Europe (27 countries) there are more than 4.1 million farms with more than 5 ha cropland (eurostat, 2009).

Compared to that, there are 41 nitrogen fertilizer plants (Kramer, 2004) in the US. The number of fertilizer plants still decreases (Rabchevsky, 1996), while the capacities of new plants grew to $\approx$1 mio tons per year (United Nation Environment Programme, 1998). Furthermore, the production is concentrated on few countries, with the four largest fertilizer producing countries (China, United States, India and Russia) accounting for 55% of global industrial nitrogen fertilizer production (Kramer, 2004).

Unfortunately, due to the trade-off between lowering policy implementation costs by simplifying the tax base and optimizing static and dynamic efficiency, a distinction between different emission parameters, depending on local conditions, inputs and management practices, entails the option to internalize external effects in a more complete manner. But on the other side, the environmental processes leading to pollution are still so complex and unclear that the development of specific emission parameters is not very fruitful. As was described in section 2.1.2, the destinies of nitrogen are on a regional or global scale, have diverse pollution effects, follow no specific path and can last decades and centuries. Thus, the way how nitrogen is injected into the nitrogen cycle is less relevant than the quantity that gets injected.

Of course, taxation of only one input will increase the utilization of alternative inputs above the optimal level. Still, the utilization of alternative nitrogen inputs is restricted as they are usually by-products of other outputs (legumes or livestock) that would not be cost-effective for their own sake. As a result, total substitution by other inputs is very unlikely.

The taxation of industrial fertilizer will certainly increase food prices. Yet, industrial fertilizer is mainly used in developed countries, while developing countries still use a much larger share of natural fertilizers. This way, an industrial fertilizer tax might burden low-income households of developing countries less than a taxation of all emission-relevant inputs and practices. Furthermore, tax revenue could be redistributed to low-income households to reduce the negative effects of food price increases.

In conclusion, an industrial nitrogen fertilizer tax seems likely to be an adequate policy option to combat nitrogen pollution. In the next subsection, some indicators shall be developed to test this hypothesis.

2.4.2 Indicators for Quantification of Policy Efficacy

To make policy instruments comparable, some indicators have to be developed which represent well the central aims of policy makers, and which are at the same time quantifiable. These indicators shall be estimated in the empiric part of this work. Because the scope of this work is limited, the author will concentrate on only one source of pollution, the greenhouse gas and ozone depleting substance N_2O, and on only one policy option, the taxation of industrial nitrogen fertilizer. As indicators for evaluation, the following five criteria were deduced from the main challenges for effective regulation mentioned in section 2.4:

- **Environmental effectiveness**
  As an indicator for environmental effectiveness, the author defines the N_2O emissions that can be saved at specific cost levels.

- **Static efficiency**
  As an indicator for static efficiency, the author defines the costs at which a certain emission aim can be reached with measures that do not alter current production technology. In this current state, farmers have the possibility to shift their production from one area to another, to shift between crop types and between different types of nitrogen inputs. The nitrogen efficiency of different inputs can not be altered.

- **Dynamic efficiency**
  As an indicator for dynamic efficiency (or innovation efficiency), the author defines the production-cost-savings reached by an improvement of nitrogen efficiency(nitrogen applied per nitrogen incorporated into crops). The higher cost savings by improved fertilizer application are, the higher is the incentive to invest into nitrogen-saving technology.
  An indicator which would usually be more appropriate for evaluating dynamic efficiency would be the cost-savings by improved emission efficiency(emissions per nitrogen incorporated into crops). Yet, as emissions cannot be measured, emission parameters are set by the regulation authority and cannot be influenced by farmers. Thus there is no dynamic incentive for farmers which goes beyond improving nitrogen efficiency.

- **Policy implementation costs**
  As an indicator for policy implementation costs, the author will compare the costs of an industrial fertilizer tax to the costs of a farm-level emission tax which taxes all nitrogen inputs according to their emissions. The difference between these two cost levels may help to judge whether a farm-level tax should be favored to a fertilizer-plant-level tax. If real policy implementation costs are lower than this difference, a farm-level tax should be favored; if real policy implementation costs are higher, a fertilizer-plant-level tax should be preferred.

- **Distribution**
  As an indicator for distributional effects, the author takes the increase in shadow food prices of a region. Higher prices will put pressure on low-income households. Especially in developing economies where food still constitutes a big share of total consumption, price increases may lead to poverty and hunger and should be kept minimal.

Furthermore, the impact on state budgets shall be quantified. Governments can use the tax revenue to forward certain public goals, to lower taxes or to transfer it back to the citizens.

Chapter 3

Model

In the following, an economic model shall be used to estimate the indicators of effective policy for an industrial nitrogen fertilizer, that were developed in the last chapter.
Models can be seen as *"a representation of a selected part of the world (the 'target system')"* (Frigg & Hartmann, 2006, p.2). Yet, a model that fully covers the target-system and agrees with its complexity is an unnecessary model, because the elaboration of such a model would already require full knowledge of the target system. The model would thus give no further insights (Frigg & Hartmann, 2006). Instead, Models should be *'macro-scopes'* that *"reduce, rather than magnify as microscopes do, giving Earth-system scientists an objective distance from their specimens – no longer too close for cognitive comfort."*(Schellnhuber, 1999) They should be an idealized version of the target system, a *"deliberate simplification of something complicated with the objective of making it more tractable"* (Frigg & Hartmann, 2006, p.4). One can distinguish between two forms of idealization:
The *'Aristotelian idealization'* describes the neglect of information that is not relevant for the result of the model. Examples are the color of the harvested vegetables or the farmers name. Although they belong to the target system (in this case the agricultural sector), they can be left out without changing parameters like the agricultural N2O emissions. The *'Galilean idealization'* describes simplifications that are known to distort the model results. Examples are the assumptions that direct soil emissions are equal for all soil types or that yield levels are already known to the optimizer when plants are sown. These alterations are made out of the awareness that a problem is too complicated to be tackled without these assumptions and that the model results still hold a surplus value (McMullin, 1986). The borderline between these two types of idealization blurs under closer examination: can any variable be truly identified as being without influence on the result? Even the religious confession of a farmer might have a structural influence on the farmers' production, as Muslims for example believe pigs to be impure animals.
Models are thus always idealizations and distortions of the target system; errors in the results may thus not only occur but are unavoidable.
Economic models often fail in *"communicating the limitations, weaknesses, and even dangers of its preferred models to the public."* (Colander *et al.*, 2009, p.1) This gives policy makers and the public a 'control illusion', especially when

models are concealed behind difficult mathematical formula and complex computer programs. As Crawford (Crawford, 1996) states, "*this situation bears a disturbing resemblance to computer-assisted intellectual dishonesty. Human beings have always been masters of self-deception, and hiding the essential basis of one's deception by embedding it in a computer program surely helps reduce what might otherwise become an intolerable burden of cognitive dissonance.*" Thus economists should "*make clear the limitations and underlying assumptions of [their] models and warn of the dangers of their mechanic application*" (Colander *et al.*, 2009, p.6).

This study will use a Model of Agricultural Production and its Impact on the Environment (MAgPIE) to estimate the indicators of policy efficacy. To create the transparency which is required out of the above considerations, this chapter will explain the model framework in detail to give the reader an insight into the model's methodology. After the model output is presented and compared to other studies and real measured data in Chapter 4, the underlying assumptions and their impact on the model output shall be discussed in Chapter 5.
This chapter will be structured as follows: The first section will focus on the status-quo of the Model of Agricultural Production and its Impact on the Environment (MAgPIE) as published in (Lotze-Campen *et al.*, 2008, 2009). The second section will describe the model-extensions realized for this thesis. Firstly, a nitrogen constraint was introduced to model the influence of nitrogen scarcity on the farmers' behavior. All crop and livestock activities have specific effects on the nitrogen budgets, which have to be balanced out in the optimization process. Secondly, all agricultural activities were linked to specific N_2O emissions. Thirdly, taxes, both on industrial fertilizer and emissions, were included into the model. The formulas of the model and its extensions can be found in Annex A.1.

3.1 MAgPIE

MAgPIE (Lotze-Campen *et al.*, 2008, 2009) is a global land use allocation model, which is programmed in GAMS (Brooke *et al.*, 2003). It is coupled with a grid-based dynamic vegetation model (LPJmL) (Sitch *et al.*, 2003; Bondeau *et al.*, 2007). Hereby it takes into account regional economic conditions as well as spatially explicit data on potential crop yields, land and water constraints and derives specific land-use patterns, yields and total costs of agricultural production for each grid cell. Since the implementation and validation of MAgPIE is presented in detail elsewhere (Lotze-Campen *et al.*, 2008, 2009), only a short overview will be provided here.
The non-linear objective function of the land-use model is to minimize the total cost of production for a given amount of agricultural demand. Regional food energy demand is defined for an exogenously given population and income growth in ten food energy demand categories (cereals, rice, vegetable oils, pulses, roots and tubers, sugar, ruminant meat, non-ruminant meat, and milk), based on regional diets (FAOSTAT, 2008). The model makes the simplifying assumption that no substitution between these demand categories exists: demand for ruminant meat cannot be substituted by milk or vegetables. Yet, some demand categories can be satisfied by a number of different cropping activities.

There are 20 cropping activities in the MAgPIE model (temperate cereals for food or feed, maize for food or feed, tropical cereals for food or feed, rice, five oil crops, pulses, potatoes, cassava, sugar beets, sugar cane, vegetables/fruits/nuts, two fodder crops) and 3 livestock activities (ruminant meat, non-ruminant meat, milk). Cropping activities within a demand category are perfect substitutes at a fixed caloric ratio (e.g. temperate cereals, tropical cereals and maize can all satisfy equally the demand for cereals). Feed for livestock is produced as a mixture of grain, green fodder, and pasture at fixed proportions. Fiber demand is currently fulfilled with one cropping activity (cotton). Cropland, pasture and irrigation water are fixed inputs in limited supply in each grid cell, measured in physical units of hectares (ha) and cubic meters (m^3). Variable inputs of production are labor, chemicals, and other capital (all measured in US$), which are assumed to be in unlimited supply to the agricultural sector at a given price. Moreover, the model can endogenously decide to acquire yield-increasing technological change at additional costs, if otherwise there is no feasible solution (i.e. land use pattern) under a given set of resource constraints.

Potential crop yields for each grid cell are supplied by the Lund-Potsdam-Jena dynamic global vegetation model with managed Lands (LPJmL) (Sitch *et al.*, 2003; Bondeau *et al.*, 2007). LPJmL endogenously models the dynamic processes linking climate and soil conditions, water availability and plant growth, and takes the impacts of CO2, temperature and radiation on yield directly into account. LPJmL also covers the full hydrological cycle on a global scale, which is especially useful as carbon and water-related processes are closely linked in plant physiology (Gerten et al., 2004; Rost et al., 2008). Potential crop yields for MAgPIE are computed as a weighted average of irrigated and non-irrigated production, if part of the grid cell is equipped for irrigation according to the global map of irrigated areas (Döll and Siebert, 2000). In case of pure rain-fed production, no additional water is required, but yields are generally lower than under irrigation. If a certain area share is irrigated, additional water for agriculture is taken from available water discharge in the grid cell. Water discharge is computed as the runoff generated under natural vegetation within the grid cells and its downstream movement according to the river routing scheme implemented in LPJmL.

Spatially explicit data on yield levels and freshwater availability for irrigation is provided to MAgPIE on a regular geographic grid, with a resolution of three by three degrees, dividing the terrestrial land area into 2178 discrete grid cells of an approximate size of 300 km by 300 km at the equator. Toward higher latitudes, the grid cells become smaller. Each cell of the geographic grid is assigned to one of ten economic world regions (Figure 2): Sub-Saharan Africa (AFR), Centrally-planned Asia including China (CPA), Europe including Turkey (EUR), the states of the Former Soviet Union (FSU), Latin America (LAM), Middle East/North Africa (MEA), North America (NAM), Pacific OECD including Japan, Australia, New Zealand (PAO), Pacific (or Southeast) Asia (PAS), and South Asia including India (SAS). The regions are initially characterized by data for the year 1995 on population (CIESIN et al., 2000), gross domestic product (GDP) (World Bank, 2001), food energy demand (FAOSTAT, 2008), average production costs for different production activities (McDougall *et al.*, 1998), and current self-sufficiency ratios for food (FAOSTAT, 2008). While all supply-side activities in the model are grid-cell specific, the demand side is aggregated at the regional level. Aggregate demand within each

region, defined by total population, average income and net trade, is being met by the sum of production from all grid cells within the region.
Trade in food products between regions is simulated endogenously, constrained by minimum self-sufficiency ratios for each region. This is to say that some minimum level of domestic demand has to be produced within the region, while the rest can be allocated to other regions according to comparative advantages. If, for instance, a region currently has a self-sufficiency ratio of 1.2 for a certain product, then in future projections this may either be kept constant or gradually reduced over time to account for global trade liberalization.
For future projections, the model works on a timestep of 10 years in a recursive dynamic mode. The link between two consecutive periods is established through the land-use pattern. The optimized land-use pattern from one period is taken as the initial land constraint in the next. For the base year 1995, total agricultural land is constrained to the area currently used within each grid cell, according to (Ramankutty & Foley, 1999). In the following periods, additional land from the non-agricultural area can be converted into cropland at additional costs. Trade in food products between regions is simulated endogenously, constrained by minimum self-sufficiency ratios for each region.
MAgPIE is calibrated for the 1995 timestep by adjusting yield levels with an region- and crop-specific calibration factor to meet the harvested area and production of FAOSTAT.

3.2 Model Extensions

For the purpose of this work, the model was extended in three respects:
First, a nitrogen balance constraint was included which determines the cost-optimal mix of crops and the required amount of industrial fertilizer. In a second step, all nitrogen inputs were connected to specific N2O-emissions. Third, the price of industrial nitrogen fertilizer can be raised, simulating an industrial fertilizer tax. Alternatively, a tax on all nitrogen inputs according to their emission factors can also be implemented to estimate minimal policy implementation costs.

3.2.1 Nitrogen

The first part of the model-extension is the implementation of nitrogen scarcity as an additional constraint into the model. The core of this nitrogen modelling approach is the constraint that the long-term nitrogen budget within each MAgPIE-cell has to be balanced out. Nitrogen withdrawn from the agricultural system (e.g. by harvested crops) or lost to the natural environment (e.g. by volatilization or leaching) has to be replaced (e.g. by fertilizer) to allow for permanent production.
Even though the nitrogen budget has to be balanced out within one timestep (10 years), short-term variations of nitrogen stocks are allowed: for example, crop rotations can be used to balance out the withdrawal of a normal plant in one year with the cultivation of a nitrogen-fixing plant in the next year. This also allows for manure transport within a local domain of a cell.
(Insert: balance central to the approach, begründung)

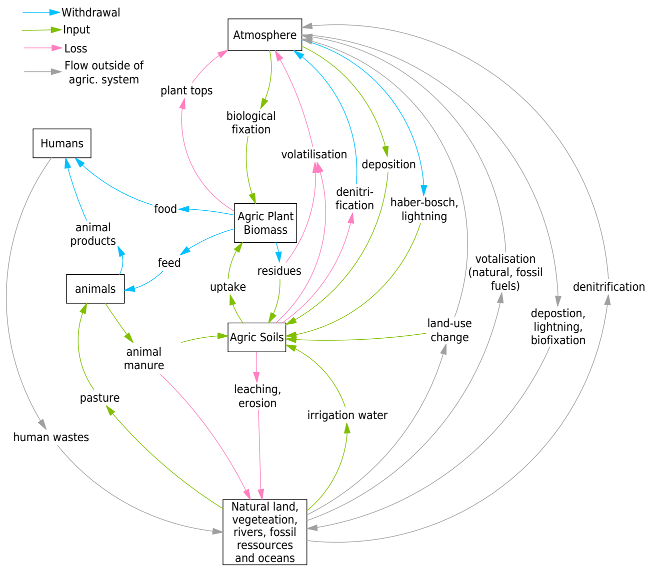

Figure 3.1: The agricultural nitrogen cycle in a simplified depiction.

Figure 3.2.1 depicts a simplified version of the agricultural nitrogen cycle. Nitrogen withdrawals from the agricultural system are depicted as blue arrows, inputs as green arrows and losses as red arrows. Not all inputs could be implemented into the model. At the current state, nitrogen withdrawals implemented into the model are harvested crops for food and feed(W_{Harv}) and crop-residues(W_{Res}). Nitrogen inputs are recycled crop residues(I_Res), biofixation(I_{Fix}), manure(I_{Man}), atmospheric deposition(I_{Dep}) and industrial fertilizer(I_{Fert}). Losses are not further distinguished and represented by multiplying with a factor for nitrogen efficiency (N_{eff}). The disregard of certain inputs will lead to an under-estimation of losses, as will be explained in more detail in section 3.2.1.

Nitrogen withdrawals, inputs and losses were linked to a number of endogenous variables of the model, like cultivated area(A), harvested quantities(Q), cropping(C) or livestock(L) activities and economic regions(R), leading to the constraint

$$
\begin{aligned}
&W_{Harv}(Q,C) + W_{Res}(Q,R,C) \leq N_{eff}(R)* \\
&(I_{Res}(Q,R,C) + I_{Fix}(A,R,C) + I_{Man}(L,Q,R) + I_{Dep}(A,R) + I_{Fert}) \qquad (3.1)
\end{aligned}
$$

The optimization process searches for the cheapest combination of nitrogen-withdrawals and nitrogen-inputs which can satisfy the global demand for food. Every cell obtains a specific nitrogen-mix depending on its production strengths and weaknesses. Because the constraint will restrict the action space of the optimizer, every agricultural activity of the nitrogen balance which has an impact receives a nitrogen shadow price that is flexible and cell-specific. This will affect the relative profitability of agricultural activities and thus the structure of agricultural production.

Crop Harvest: W_{Harv}

One of the main nitrogen removals is the harvest of crops. The amount of removed nitrogen depends on the harvested quantity and the nitrogen content of the crop. While the harvested quantity is determined endogenously by the model, the nitrogen contents of the cropping activities were taken from various publications: Fritsch (2007); Beale & Long (1997); Corbeels *et al.* (2003); Lodhiyal & Lodhiyal (1997); Jørgensen & Schelde (2001); Jackson (1997); Robertson & Rosswall (1986); Devendra (1987); Lewandowski & Schmidt (2006); Neve & Hofman (1998); Preston (1977). See table A.1 in Annex A.3 for the values adopted.
The values presented are only a crude approximation of real values, which may diverge depending on breed, management-conditions and harvest date. Many values were taken from non-representative case studies, as no comprehensive evaluations were found. In cases with multiple studies or diverging values for crops within a crop category, an average value was taken.

Plant residues: W_{Res}

Beyond the harvested crops, the residues of a plant are an important by-product and make up a large share of total biomass. The nitrogen that is taken up by plant residues is difficult to estimate, as no data is collected by public statistics (Smil, 1999). Thus crop to residue ratios were used to derive the quantity of crop residues from the endogenously estimated crop output. When possible, region and plant specific crop to residue ratios were taken from Wirsenius (2000); when data was incomplete, it was completed with plant specific ratios from (Krausman *et al.*, 2008), (Fritsch, 2007), (Lal, 2005), (Jingura & Matengaifa, n.d.). The nitrogen content of residues were drawn from Fritsch (2007); Beale & Long (1997); Corbeels *et al.* (2003); Lodhiyal & Lodhiyal (1997); Jørgensen & Schelde (2001); Jackson (1997); Robertson & Rosswall (1986); Devendra (1987); Lewandowski & Schmidt (2006); Neve & Hofman (1998); Preston (1977). See table A.1 for the values adopted.

Plant Residue Recycling: $\mathbf{I}_{Res}$

Residues have different functions in an agricultural system. A part of the residues is used for animal feed and bedding or burned as household fuel in rural areas. An important share is furthermore burned on the fields to control diseases or weeds, to reduce the need for agricultural chemicals and to minimize fire hazards. Most often the residues are recycled to the soils and used as organic fertilizer and nitrogen input.
In accordance with Smil (1999), it was assumed that 35 % of the residues are burnt in developing countries and 15 percent in developed countries.[1] These estimates exceed the proposals of the IPCC 1997, which are according to Smith

[1] It was guessed that all countries in EUR, NAM, PAO and FSU possess a modern agricultural sector and are thus classified as "'developed"', while all other regions were assumed to belong to the category "'developing"'.

et al too conservative. Furthermore, combustion efficiency was set in Smith et al to 80 % and the release of N_2 in the combustion process to 35%, leading to a return of 72% of the nitrogen from burned residues. Of the residues which were not burned, it was assumed that 23% and 33% are fed to livestock in developed and developing countries respectively (Smil, 1999). The rest of the nitrogen serves as nitrogen input to agricultural soils.

Biofixation: I_{Fix}

Biofixation of plants depends on many factors like crop-type, moisture, temperature, light, soil acidity, phosphorus availability and mineral N availability (Mulongoy, 1992).
The implementation of biofixation for this study is mainly based on Smil (1999). It was assumed that nitrogen fixation depends mainly on the crop-type and the area under cultivation, and not on the production quantity. For cereals, non-leguminous oil-crops, fiber crops, roots and tubers, an annual rate of $5\frac{kgN}{ha}$ was employed, which shall meet the fixation by free living non-Rhizobium diazotrophs. In rice-fields free-living cyanobacteria and the cultivation of Azolla pinnata (a floating fern) can fix $\approx 33\frac{kgN}{ha}$. Endophytic diacotrophs, which live inside sugar cane roots, leaves and stems, can fix $\approx 111\frac{kgN}{ha}$. For the MAgPIE category ‘pulses’, which includes most leguminous crops, the area-weighted average fixation is $\approx 49\frac{kgN}{ha}$. The legumes soybeans and groundnuts feature nitrogen-fixation rates of $\approx 80\frac{kgN}{ha}$. Finally, leguminous cover-crops which are represented by our fodder-crop categories are assumed to fix $\approx 100\frac{kgN}{ha}$ annually. According to Smil (1999), these rates may hold an error rate of $\pm 25\%$.

Manure: I_{Man} and F_{PRP}

Manure is often recycled to agricultural soils. The amount of nitrogen used as fertilizer was estimated on the basis of IPCC (2006). When no default values were available in IPCC 2006, older values from IPCC 1996 were used. The calculation starts with the estimation of manure excretion, which depends on the animal race and regional specific body weights. The share of manure managed under a specific animal waste management system (AWMS) was disaggregated regionally and animal-specific, using the default values of IPCC 2006, and complemented with default values from 1996, when no data existed in the new methodology. Manure management was differentiated into 11 categories: *anaerobic lagoons*, *liquid systems*, *daily spread*, *solid storage*, *drylot*, *digester*, *pit storage*, *pasture range and paddock (grazing)*, *used as fuel chicken layers*, and *other systems*. For each AWMS, a specific share of nitrogen returned to agricultural soils was used, as some part of manure is used for ulterior purposes or lost via leaching or volatilization.[2]
Animal bedding is often mixed with manure and used as fertilizer. Thus, in the case of cows and pigs under the AWMS 'solid waste' and 'other systems',

[2]In some cases, default values for losses were not available for all animals managed under a certain system. In these cases, it was assumed that losses are as high as for dairy cows managed under the same system. In the case of 'other systems', the default values for deep bedding were used. Only in the case of poultry, we used the average between 'wet' and 'dry layers' for the AWMS 'drylot', 'liquid slurry' and 'other systems'.

nitrogen in bedding was added to manure nitrogen.
The calculations were done with 9 animal types (Dairy Cattle, Non-Dairy Cattle, Sheep, Goat, Market Swine, Breeding Swine, Poultry, Buffalos and Others) that were finally assigned to the three region specific livestock product categories using FAOSTAT data (FAOSTAT, 2008) on animal stocks from 1995. The IPCC 2006 regions were assigned to their closest MAgPIE counterpart, and the IPCC animal breed categories were assigned to the three MAgPIE livestock production types weighted by FAO 1995 livestock numbers.
The composition of animal breeds and of AWMS within a MAgPIE category is thus assumed to be fixed and cannot be influenced by the optimization process. As FAO does not distinguish between market and breeding swine, it was assumed that breeding swine account for 10% of total swine.

Atmospheric deposition: I_{Dep}

Soils can receive considerable amounts of nitrogen through atmospheric deposition. Once entering the atmosphere due to volatilization of fertilizers, waste discharge, power generation, traffic, industry and open biomass burning, the nitrogen in the form of NO_x and NH_3 is scavenged by the rain to the soils or deposited in a dry state.
The total amount of atmospheric deposition was taken from emission and transport modelling results of (Dentener *et al.*, 2006). Regional average depositions for MAgPIE's ten world regions were aggregated, assuming homogeneous per ha depositions within a region. The deposition rates of 2000 were used for 1995, and the optimistic scenario results for 2030 were used to account for future changes in nitrogen deposition. Deposition rates are listed in table A.3.
Taking average rates of atmospheric deposition will probably lead to an underestimation of actual rates of atmospheric deposition: volatilization of reactive nitrogen mainly occurs by volatilization of fertilizer in agricultural areas, or by fossil fuel combustion in populated areas (which are often close to agricultural areas). Even though NO_x and NH_3 can cover large distances before being deposited on the ground, atmospheric deposition on natural vegetation is thus significantly lower than on agricultural soils (Smil, 1999). This bias could be reduced by using more spatial explicit datasets in future studies.

Industrial Fertilizer: I_{Fert}

The model allows farmers to buy industrial fertilizer products at an exogenous price. As the current model was calibrated to meet the current demand, fertilizer costs were already priced in. Thus, a marginal price close to zero was used for 1995. For simulations of 2005 and the future, only the amount by which the price for fertilizer increases (for example due to taxes) has to be paid by the farmer.
It was assumed that production capacities can easily be expanded in the long run and that supply changes will have no significant influence on prices. According to (Bumb & Baanante, 1996), this assumption seems plausible, as supply faces no major constraints: the raw material (natural gas) is still available in sufficient quantities, and the demand for fertilizer will not have a large effect on natural gas prices. Fertilizer producing companies in both developing and developed countries have funds available for investment, and the technology of

production is available on a wide basis.
Even though the price of industrial fertilizer is not heavily determined by the quantity of demand, it remains difficult to predict its future levels. As the production costs depend largely on the price of natural gas, the fluctuations of past gas prices were passed on to fertilizer prices (See fig. A.2 in Annex A.2). Future price levels thus remain as uncertain as the price of fossil fuels. For this model, we assumed the inflation corrected price to be stable. From 1995 to 2005, this assumption actually meets real price changes: the inflation corrected and consumption weighted price basket of anhydrous ammonia, nitrogen solutions, urea, ammonium nitrate and ammonium sulfate with US-farm prices stayed at approximately 640 $US\$_{1995}$ per ton Nr (own calculations based on fertilizer prices of United States Department of Agriculture (2009), fertilizer quantities of International Fertilizer Association (IFA) (2009) and inflation rates of International Monetary Fund (IMF) (2009)).

Losses to natural systems and nitrogen efficiency: N_{eff}

Losses of reactive nitrogen to natural systems (non-agricultural land, riverine and coastal systems, sea, atmosphere) are diverse. They include volatilization in the form of NO, N_2O, N_2 and NH_3, leaching in the form of NO_3^-, soil erosion and losses from plant tops (Smil, 1999; IPCC, 2006). Losses are extremely difficult to measure on field-level and show large variations, which makes bottom-up estimates very problematic.
For this implementation, a top-down approach is used to estimate losses to natural systems: total inputs and total withdrawals are aggregated on a regional basis, using FAO information from 1995 for area and production data and IFA data on fertilizer consumption in 1995 instead of endogenous model outputs (see table 3.2.1).
Let the output-to-input ratio be defined as *nitrogen efficiency* N_{eff}, which represents how much of the inserted nitrogen is actually incorporated into plant biomass and does not get lost to natural systems. In field studies, these fertilizer recovery rates were estimated to lie between 11% and 82% Mosier (1995); Tandon (1993); Strong (1995); Smil (1999) and Smil (1999) estimates that the global average lies between 35 and 65 percent, with a best guess of 50%. This study calculates a global nitrogen efficiency of 54%, a rate which lies approximately in the middle of this range. Yet, regional nitrogen efficiencies are sometimes too high to be supportable by bottom-up studies. This is the case for Africa (84%), Latin America (76%) and Pacific Asia (86%), regions with high rates of deforestation. Land-use change is a large contributor to the global nitrogen budget. According to Vitousek et al, land-use change (including deforestation, biomass burning and wetland drainage) releases $\approx 70\frac{TgN}{yr}$, one third of global anthropogenic reactive nitrogen. Even though only smaller fractions of this nitrogen will serve as input for agricultural production, the disregard of this input gives the false impression that these regions can reach the same agricultural outputs with lower inputs. Furthermore, soils have not necessarily a balanced nitrogen budget. Unsustainable nutrien depletion of soils plays a large role in agriculture, and it is estimated that 22% of global cropland already have been depleted (yet nitrogen-depletion only is one of multiple reasons for that) (Chen *et al.*, 2002). As information about the proportion of reactive nitrogen remaining on agricul-

tural land after land-use change was not available on a global scale, it has to be assumed that nitrogen input by land-use change will make up a constant portion of total future nitrogen inputs. A fixed rate of nitrogen efficiency thus stands for a continuation of the status-quo concerning the proportion of nitrogen losses and non-regarded nitrogen inputs to total nitrogen withdrawals. This will distort the model results because regions with high rates of current land-use change are assumed to have high nitrogen efficiency also for industrial fertilizer, decreasing the costs of applying taxed fertilizer.
Uniform regional rates of nitrogen losses also do not account for large variations of losses between farming intensity, crop-type, climate and fertilizer-type: crops with low fertilization rates (<150 kg per ha) have rather high recovery rates (60-65%); forages and legumes may have nitrogen uptake rates of 75% and 65%, while rice has rates around 35%; and plants in humid climates may have a recovery rate of 55% compared to 35% in rainfed crops in drier climates (Smil, 1999); nitrogen inputs from manure has higher rates of losses than industrial fertilizer application, while biofixation losses are extremely low (IPCC, 2006). These effects were neglected in this study and should be included into further research.

	Withdrawals *Tg N*			**Inputs** *Tg N*						**N_{eff}** *%*
	Harvested Crops	**AG Residue**	**BG Residue**	**AG Residue Recycling**	**BG Residue**	**Biological Fixation**	**Manure**	**Atmospheric Deposition**	**Industrial Fertilizer**	**Nitrogen Efficiency**
AFR	3,0	2,8	0,2	1,4	0,2	2,4	2,3	0,7	1,2	**0,74**
CPA	9,5	8,0	0,8	4,0	0,6	3,4	3,1	1,8	24,2	**0,49**
EUR	7,5	2,8	0,8	1,9	0,7	3,6	4,1	1,6	12,1	**0,46**
FSU	6,0	1,3	0,4	0,9	0,3	7,3	2,4	0,6	2,5	**0,56**
LAM	5,3	4,0	0,4	2,0	0,3	4,6	1,8	0,6	2,7	**0,80**
MEA	1,4	0,8	0,1	0,4	0,1	0,8	0,7	0,1	2,4	**0,53**
NAM	11,3	4,3	0,8	2,9	0,7	6,8	2,0	1,0	12,8	**0,62**
PAO	2,3	0,4	0,1	0,3	0,1	3,1	0,4	0,1	1,3	**0,54**
PAS	4,3	2,0	0,2	1,0	0,1	1,5	0,8	0,5	4,0	**0,82**
SAS	7,6	4,1	0,5	2,1	0,4	6,6	2,6	3,3	13,3	**0,43**
Total	**58,4**	**30,6**	**4,3**	**17,0**	**3,4**	**39,9**	**20,2**	**10,3**	**76,5**	**0,56**

Table 3.1: Estimated regional nitrogen withdrawals, nitrogen inputs and nitrogen efficiency, calculated on the basis of various sources (see section 3.2)

3.2.2 N_2O emissions

Anthropogenic N_2O emissions are calculated on a cell-basis including both direct and indirect emissions from agricultural soils and emissions form animal waste management systems (AWMS). If not stated otherwise, the management and emission factors are consistent with the Tier 1 methodology of the 2006 Guidelines for National Greenhouse Gas Inventories (IPCC, 2006). It is likely

that these IPCC inventories will be the basis for future international treaties on climate change (Schlamadinger *et al.*, 2007a). In contrast to IPCC (2006), this implementation does not account for N_2O emissions from the management of organic soils and from nitrogen mineralization due to the loss of organic carbon by land-use change, which forms an important part of total emissions. Soil emissions from nitrogen input through compost, sewery wastes and organic amendments like brewery waste were also not considered.
The endogenous variables for the calculation of the emissions are the cell-specific quantities of nitrogen inputs as described in section 3.2.1 and the livestock production in the cell. The nitrogen ratio of residues was calculated in a manner partly dissenting from the IPCC methodology (see 3.2.1).

Total agricultural N_2O emissions N_2O_{Tot} are composed of direct soil emissions N_2O_{Dir}, indirect soil emissions N_2O_{Ind}, emissions from animal waste management systems(AWMS) N_2O_{AWMS} and emissions from pasture ranges and paddocks N_2O_{PRP}.

$$N_2O_{Tot} = N_2O_{Dir} + N_2O_{Ind} + N_2O_{AWMS} + N_2O_{PRP} \tag{3.2}$$

Direct soil emissions occur as a by-product of the nitrification process (the aerobic microbial oxidation of ammonium to nitrate) and as an intermediate product in the process of denitrification (the anaerobic microbial reduction of nitrate to non-reactive nitrogen in the form of N2) (IPCC, 2006). Beside other factors like temperature, precipitation, soil texture and C availability (Smil, 1999), which are not regarded in this methodology, direct emissions mainly depend on the amount of reactive nitrogen available in the soil. The IPCC Guidelines assume a linear relationship between the amount of nitrogen inputs from plant residues, industrial fertilizers and manure.

$$N_2O_{dir} = (I_{Fert} + I_{Man} + I_{Res}) \cdot EF_{1;1FR} \tag{3.3}$$

Two different emission factors were used: EF_1 for normal crops and EF_1FR for flooded rice fields. According to the 2006 methodology, nitrogen inputs by biological fixation have no significant effect on total emissions, which is a major update to the 1996 Guidelines for National Greenhouse Gas Inventories (IPCC, 1996). The 2006 methodology further has significant lower emission factors than the old methodology (1%(2006) compared to 1.25%(1996)).

Indirect soil emissions are emissions which were not released on the soils on which the nitrogen was applied, but which have their origins in nitrogen leaving the agricultural land either by volatilization or leaching, and which are transformed later on to N_2O.
When nitrogen is applied to soils, parts volatilize in the form of NH_x and NO_y. These compounds may be deposited later on on soils or surface waters, where they again enter into a process of nitrification and denitrification. Again, a linear relationship between nitrogen inputs and N_2O-emissions is assumed:

$$N_2O_{ind} = (I_{Fert} \cdot Frac_{GASF} + I_{Man} \cdot Frac_{GASM}) \cdot EF_4 \tag{3.4}$$

It is assumed that only nitrogen applied with manure and industrial fertilizer have significant rates of volatilization. Yet, the fractions $Frac_{GASF}$ of volatized

fertilizer and $Frac_{GASF}$ of volatized manure differ.
Another pathway of indirect emissions is the leaching of parts of the applied nitrogen with surface water or with the flow through soil macropores or drain pipes. This nitrogen is partly transformed into N_2O in the process of nitrification and denitrification in the groundwater or in riverine systems. Although leaching is especially high when nitrogen inputs exceed the biological demand, a linear relationship between applied nitrogen and N_2O-emissions was again assumed:

$$N_2O_{ind} = (I_{Fert} + I_{Man} + I_{Res}) \cdot Frac_{LEACH-H} \cdot EF_5 \tag{3.5}$$

Leaching is relevant for nitrogen inputs via industrial fertilizer, manure and residues, and also occurs on pasture range and paddocks. The fraction of inputs which leaches, $Frac_{LEACH_H}$, was assumed to be equal for all three inputs. Divergent from the IPCC 2006 methodology it was assumed that leaching occurs on all agricultural soils and not only on soils where the soil water holding capacity is exceeded in rainy periods. This simplification was made because no regional explicit data was available.

N_2O_{AWMS} is produced by the nitrification and denitrification of the organic nitrogen content in livestock manure and urine during storage. Direct emissions occur depending on the amount of manure handled in a specific manure management system. Furthermore, indirect emissions occur through volatilization in the form of $NO_x - N$ and $NH_3 - N$, which is later on partly transformed to N_2O. The fraction of nitrogen $Frac_{GasMS(S)}$ which volatilizes also depends on the type of AWMS. No leaching of nitrogen in AWMS was considered. The amount of nitrogen $NEXC_T$ handled under a specific AWMS S was calculated regional specific as explained in section 3.2.1.

$$N_2O_{AWMS} = \sum_S NEXC_S \cdot EF_{3(S)} + NEXC_S \cdot Frac_{GasMS(S)} \cdot EF_4 \tag{3.6}$$

N_2O_{PRP}-emissions occur when manure nitrogen is not used for the fertilization of agricultural soils, but is excreted on pasture ranges and paddocks. The amount of indirect emissions released can be considered equal to the indirect soil emissions from agricultural soils. Yet, the direct emission factor for cattle, poultry and pigs EF_{3PRP}^{CPP} is higher than the emission factor for sheep and other animals EF_{3PRP}^{SO}, because of a more even urine distribution and lower soil compaction during grazing.

$$N_2O_{PRP} = NEXC_{PRP} \cdot (EF_{3PRP}^{CPP;SO} + Frac_{GASM} \cdot EF_4 + Frac_{LEACH-H} \cdot EF_5) \tag{3.7}$$

Finally, all emissions in the form of N_2O-N are converted to N_2O emissions by multiplying with the molecular weight ratio $\frac{44}{28}$. To obtain the 'Global Warming Potential' (GWP) of these emissions, which makes the impact of N_2O comparable to CO_2 emissions, they are multiplied with 298 to receive the emissions in carbon-dioxide-equivalents CO_2eq.

3.2.3 Taxes

The tax on industrial nitrogen fertilizer was implemented as a unit tax on industrial nitrogen fertilizer. The product of fertilizer consumption and fertilizer tax was added to the total cost function, which is MAgPIE's goal function. The second tax included into the model is a tax on all nitrogen inputs according to their emission factors. This second tax was implemented to estimate the minimum policy implementation costs (see section 2.4.2). For this tax, a new variable input *emission allowance* was introduced on N_2O emissions. Each agricultural activity requires emission allowances with the price t_e according to their IPCC N_2O emission estimate. Costs for purchasing emission allowances add to the total cost function.

Chapter 4

Model Outputs and Evaluation

While the last chapter described the model framework, this chapter will now display the model output. While baseline-runs of the standard model are described extensively in Lotze-Campen *et al.* (2008, 2009), this chapter will explicitly focus on the results from the model extension of this thesis. First, the results for the nitrogen balance and the N2O emissions shall be presented for the 'business as usual' scenario. Then, the impact of a tax shall be pointed out. The last section will compare the model results of the 'business as usual' results to results of other models and to real measured data. The model was executed using the GDP and population growth datasets from the 'pessimistic' SRES A2 scenario (IPCC, 2000). In this scenario, world population rises to approximately 8 billion people in 2025 and the world economy per capita grows in average by 1.3% per annum. For the following scenarios, the area of agricultural land expansion was fixed to 1% of agricultural area. The trade balance between regions was not allowed to diverge more than 5% from current levels. It was assumed that global warming has no impact on agricultural production.
MAGpIE was run for three timesteps (1995, 2005, 2015) and solved with CONOPT (Drud, 1994), a generalized reduced-gradient algorithm which can find local optima of large-scale nonlinear programs. Model runs were made only with best-guess values. The impact of probability distributions and fuzziness on model results was not regarded quantitatively.

4.1 Nitrogen Balance

The model results indicate that nitrogen application rises in the future with increasing demand. Withdrawals by crops and residues rise from ≈75 Tg Nr in 1995 to ≈ 110 Tg Nr in 2015. Industrial fertilizer application rises in the same time-range from ≈60 Tg Nr to ≈100 Tg Nr. Compared to that, manure stays at ≈ 22 Tg for the whole period, and nitrogen inputs from biofixation even fall slightly by ≈3 Tg Nr to ≈ 34 Tg Nr.
The amount of nitrogen withdrawals, losses and inputs for each region in 1995 are illustrated in figure 4.1. Central and South Asia (CPA and SAS) are the

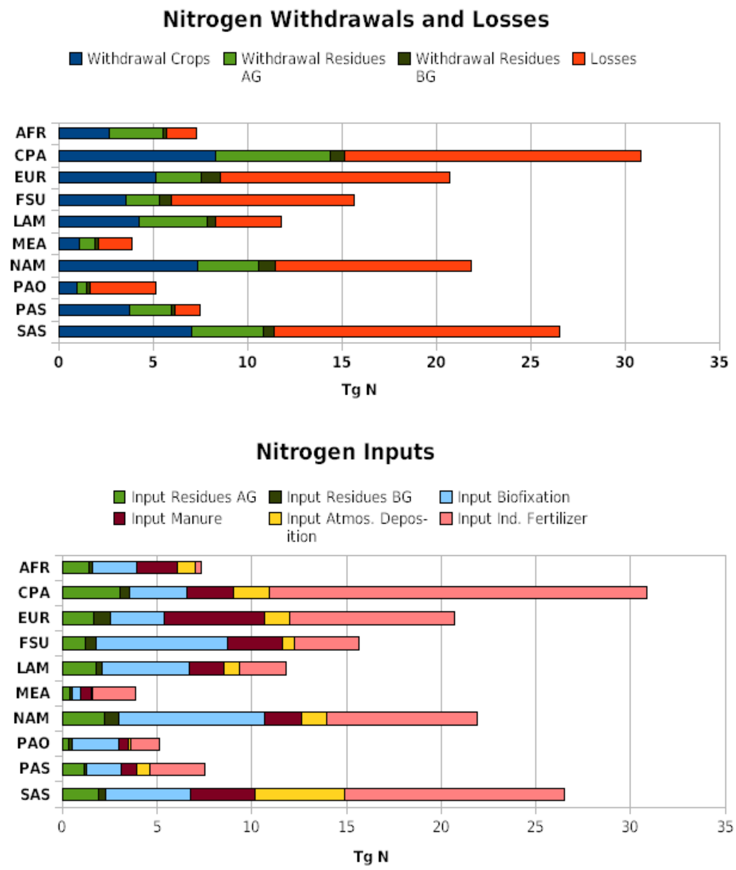

Figure 4.1: Nitrogen withdrawals, losses and inputs per region (own calculations)

regions with the highest nitrogen withdrawals and also use large amounts of industrial fertilizer. The Middle East and Pacific Asia(MEA and PAS) also use a comparatively large share of industrial fertilizer, yet have only a small share of the world nitrogen budget. Compared to that, Africa, the Former Soviet Union, Latin America and North America (AFR, FSU, LAM, NAM) have large amounts of nitrogen fixed biologically. Africa uses almost no industrial fertilizer, and in Latin America, Oceania and Japan (LAM, PAO) as well industrial fertilizer application is low. Manure recycling is especially high in Europe (EUR), but also in AFR.

4.2 N2O Emissions

It was estimated with the model, that global annual agricultural N_2O-emissions rise from $\approx$ 1530 million tons (Mt) of CO2eq for 1995 to $\approx$ 1700 MtCO2eq for 2005 and $\approx$ 1770 MtCO2eq for 2015.
In absolute terms the regions with the highest emissions are AFR, CPA, LAM

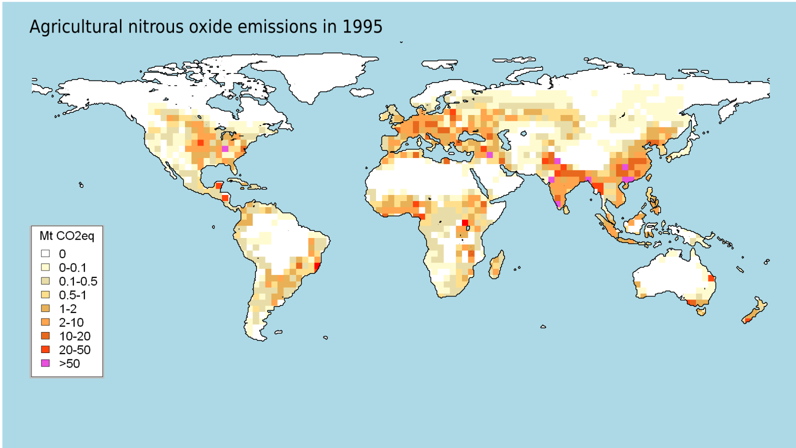

Figure 4.2: Spatial allocation of N2O emissions in 1995 in Mt CO2eq.

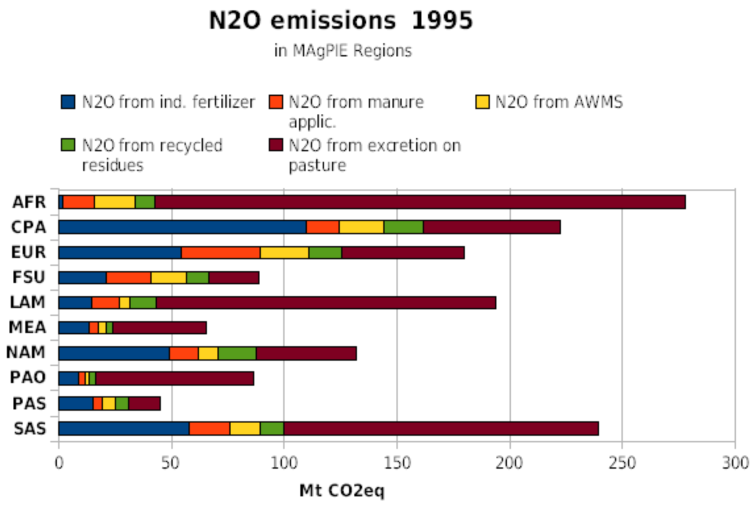

Figure 4.3: Regional N2O emissions and their sources in 1995 in Mt CO2eq (own calculations).

and SAS. In AFR, LAM, MEA, PAO and SAS emissions mainly stem from animal grazing on pasture, while in CPA, EUR, FSU, NAM and PAS the emissions are almost equally distributed between Industrial Fertilizer Application and excretion on pasture. Plant residues only have a minor impact on total GHG emissions (see figure 4.2).
While emissions from industrial fertilizer rises substantially from 1995 to 2005, there are only small increases from 2005 to 2015. Emissions from livestock production fall in the first timestep, but rise again in 2015. Emissions from recycled residues continue to rise throughout time (Figure A.2 in Appendix A.2 illustrates the main sources of N2O emissions in the ten MAgPIE world regions).
N2O emissions were calculated spatial explicitly for each MAgPIE cell. The map in figure 4.2 plots the emissions as estimated for each cell. The hot-spots of GHG discharge are in South and South-East Asia, in Europe, in the South-East of North America, in the South-East of Latin America and in Central and South-East Africa.

4.3 Tax Impact

In the following, unit taxes(tax rate) of 160(25%), 320(50%), 480(75%), 640(100%), 800(125%), 960(150%), 1280(200%), 2560(400%), 3040(475%) and 3200(500%) US $\$_{1995}$ were applied to the model. All tax impacts were estimated for the year 2015. There are mainly three major effects occurring to agricultural production structure:
First, the crop mix within a demand category is changed in such a way that nitrogen withdrawals are diminished. Crops with high nitrogen-content are substituted against crops with lower nitrogen-content. For example, for a 100% tax, tropical cereals production for food is reduced by 15% and substituted with temperate cereals and maize. Cassava production is reduced by 6% and substituted with potatoes, where production goes up 9%.
Second, plants which are able to fix nitrogen biologically receive an increasing part of the production mix. For example. sugar cane production goes up 2%, substituting sugar beet, which decreases by 2%.
Third, trade flows of agricultural products change. For example, non-ruminant meat production is shifted from CPA to AFR for taxes of 200% and higher. A shortage of industrial fertilizer substitutes in CPA makes the production of non-ruminant meat very expensive: non-ruminats are unable to digest grass from pasture land, and agricultural fodder production requires large amounts of nitrogen-inputs. AFR seams to command over cheap industrial fertilizer substitutes and thus has a comparable advantage in producing non-ruminant meat. Another example is the shift of sugar beet production from SAS to FSU, while sunflower production is shifted from FSU to SAS. FSU has a comparable shortage of cheap nitrogen substitutes and a comparable advantage in producing low-protein sugar beets, while SAS has a comparable advantage in producing nitrogen-rich sunflowers.
Fourth, for very high tax rates (>500%), more ruminants are kept than required for food demand. Most regions produce only their required minimum self-sufficiency ratio, while SAS is producing more than the world food market

requires. In this case, ruminants are only held for the purpose to transport nitrogen from the pasture land to agricultural land, and avoid purchase of taxed nitrogen fertilizer.
Fifth, as a result of these shifts, the industrial fertilizer application goes down. In the case of a 100% tax, industrial nitrogen fertilizer application is reduced by approximately 4 Tg Nr(3.7%).
These changes have an effect on the global nitrogen budget, as can be seen in figure 4.3.

In the following, we will discuss the impacts the tax has on the indicators

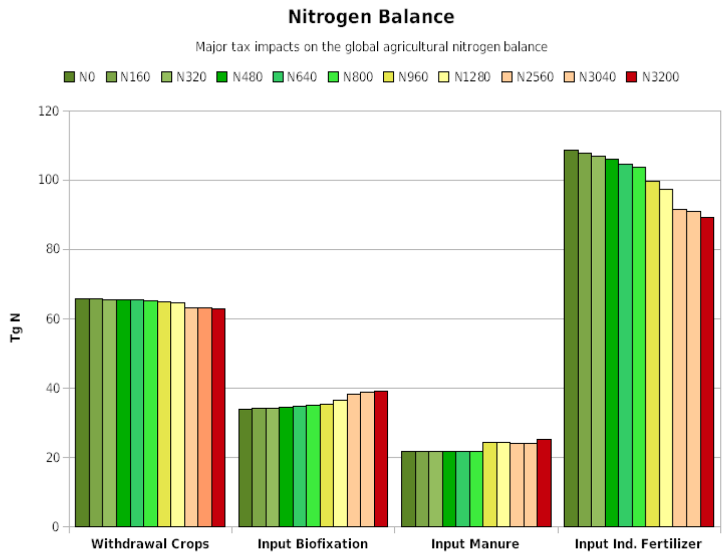

Figure 4.4: Tax Impact on the Nitrogen Budget (own calculations). Other inputs and withdrawals remain largely stable.

developed in the theory chapter:

4.3.1 Environmental effectiveness

A nitrogen tax can lower N2O emissions. Only considering static effects (no changes in nitrogen efficiency or production technology), a tax rate of 100% reduces the emissions from 1522 to 1487 Mt CO2eq. Up to a tax rate of 475%, the emissions can be reduced to approximately 1400 Mt CO2eq. Tax rates above 500% boost the emissions in the model, as animal production is increased to transport nitrogen from pasture area to croplands (see figure 4.3.1 for the impact of a tax on emissions from different sources). For tax rates higher than 475%, emissions rise continuously and might even be larger than in the baseline-scenario.
As the map in figure A.2 in Annex A.2 shows, emission abatement is concentrated on few points, mainly in SAS and FSU, regions with nitrogen shortages. On the other side, in some parts emissions might actually increase, as emission-intensive production are sourced out to regions with a comparative advantage

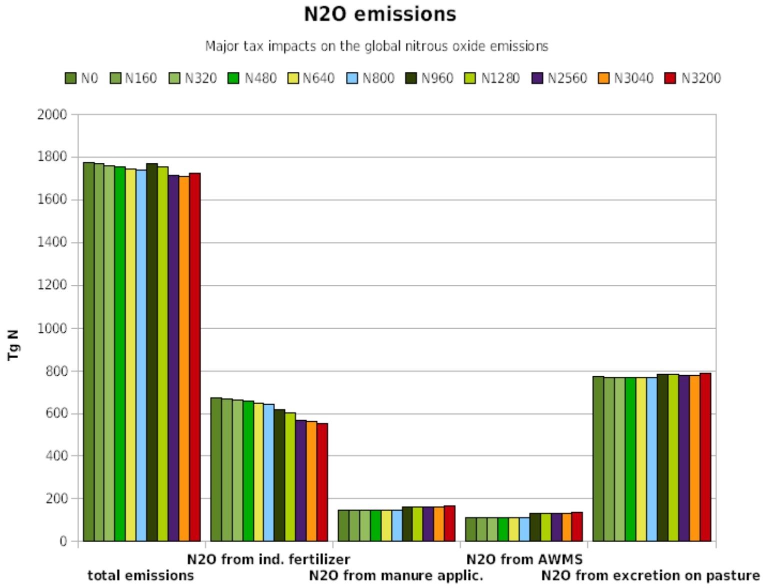

Figure 4.5: Changes in N2O emissions due to the introduction of an industrial fertilizer tax in 2015 at different tax rates (own calculations)

in producing nitrogen substitutes (mainly AFR).

4.3.2 Static efficiency

The indicator for static efficiency was defined as the costs at which a certain emission aim can be reached with the current production technology. These costs equal the increase in production costs minus the tax income that is collected by the state. Policy implementation costs and monitoring costs are not considered.

The model results for marginal and total abatement costs are depicted in figure 4.3.2. With a tax rate of 125%, approximately 30 million tons of CO2eq can be mitigated at costs below 100 US$1995 per ton CO2. Tax rates above 125% first diminish the effects of the policy; abated emissions go almost back to zero with a tax rate of 150%. These higher emissions occur at the same time when trade dynamics shift non-ruminant meat production to AFR. Because livestock productivity is low in AFR, and AWMSs are less developed, AFR has higher emissions per ton meat that other world regions, and a trade-shift leads to increasing emissions.

For higher tax rates, mitigation rises again, and at prices above 200 US$1995 reaches the effects of a 125% tax. The maximum mitigation potential lies at a global tax rate of 475%, where ≈60 million tons of CO2eq can be saved. Higher taxes lead again to both higher costs and higher emissions. This characteristic results again from cattle breeding the purpose of manure-fertilization in SAS.

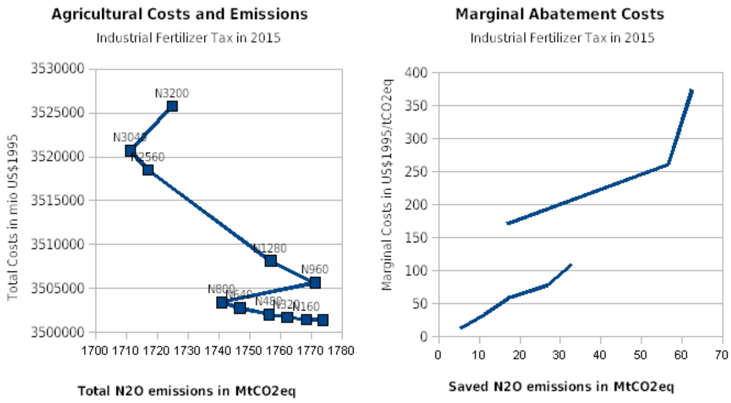

Figure 4.6: Changes in N2O emissions due to the introduction of an industrial fertilizer tax in 2015 at different tax rates (own calculations)

4.3.3 Dynamic efficiency

The indicator for dynamic efficiency was defined as the benefits of increasing nitrogen fertilizer efficiency, or more precisely, as the marginal regional cost decrease by a marginal improvement of the regional nitrogen efficiency $-\frac{\delta costs}{\delta L_{leak}}$. These cost decreases might serve as incentives for individual actors to improve their nitrogen efficiency (e.g. by precision farming or better animal waste management) or as an incentive for collective actors like states to establish policies to improve nitrogen fertilizer efficiency (e.g. educational campaigns or micro-credits).

For illustration one can calculate for a farmer the net benefit of improving nitrogen efficiency by one percentage point. First, $-\frac{\delta costs}{\delta L_{leak}}$ is divided by the total nitrogen inputs (including residues, manure, biofixation and industrial fertilizer) and also divided by hundred (for one percentage point) to receive the benefit of one-percental efficiency-improvement per ton nitrogen input (Table 4.1 lists these dynamic incentives for different world regions. Let us assume a European farmer with 100 ha cropland and total nitrogen inputs of 400 kg/ha. When an industrial fertilizer tax with a tax rate of 100% is introduced, his incentive to increase his nitrogen fertilizer efficiency by one percentage point (which means reducing his nitrogen inputs by 4kg/ha) is more than 500 Dollar.

Assuming that a 100% industrial fertilizer tax would increase the nitrogen efficiency by 5 percentage points in all world regions, global N2O emissions would go down by almost 120 MtCO2eq. The impact of a nitrogen fertilizer tax would thus more than triple compared to the static model without efficiency improvements. Furthermore, the average price of mitigating one ton of N2O goes down to only 8.50$ compared to 51$ without efficiency improvement.

As soon as farmers start to increase their nitrogen efficiency, they will require less industrial fertilizer. Then the tax will burden them less and the incentives for further efficiency improvements are diminished. After an efficiency improvement of 5 percentage points, the benefits of improving fertilizer efficiency further fall by 10-20%, depending on the region.

Some results for different tax rates and efficiency improvement levels are listed in table table 4.1.

Tax Rate[1]		50%	100%	100%	100%	100%	100%	200%
Efficiency increase		-	-	2,5%	5,0%	7,5%	10,0%	-
Dynamic Incentive[2]	**AFR**	12	24	22	21	20	19	36
	CPA	8	16	14	13	12	11	30
	EUR	7	14	12	11	10	9	27
	FSU	6	11	10	10	9	8	22
	LAM	4	9	8	8	7	6	17
	MEA	8	16	14	13	12	11	32
	NAM	5	10	9	8	8	7	19
	PAO	3	7	6	6	5	5	14
	PAS	5	10	9	9	8	8	19
	SAS	11	21	19	17	16	14	41
Total Emissions[3]		1762	1747	1697	1653	1610	1572	1757
Saved Emissions[3]		12	27	77	121	163	202	17
Perc. Change		1%	2%	5%	7%	10%	13%	1%
Average Costs		25	51	16	9	6	4	398
of Mitigation[4]		25	51	16	9	6	4	398

[1] assuming a fertilizer price of 640 US$1995
[2] Dynamic Incentive for improving nitrogen efficiency by 1% in US$1995 per ton applied Nr
[3] in MtCO2eq
[4] in US$1995 per MtCO2eq

Table 4.1: Dynamic incentives and improved-efficiency scenarios (own calculations)

4.3.4 Policy Implementation Costs

To see whether the second best policy of an industrial fertilizer tax can actually be justified by policy implementation costs, the results shall be compared to a scenario in which all emission-relevant inputs (not only industrial nitrogen fertilizer) are taxed according to their emissions.
As was explained in the theory chapter, an all-input tax should be less costly in reducing emissions than an industrial fertilizer tax in the absence of policy implementation costs. The difference in costs can be interpreted as the minimum feasible cost difference of policy implementation between the two taxes, which would still justify a tax on fertilizer-plant-level. If policy implementation costs are lower than this difference, a farm-level all-input tax would be better than an industrial fertilizer tax on the fertilizer-plant-level.

Figure 4.3.4 shows the cost differences for an industrial fertilizer tax and an all-input emission tax, assuming constant nitrogen efficiencies for both. The cost difference is low for low tax rates. Yet, tax rates of more than 125% lead to a cost gap between the two taxes, which slightly increases over time. Emission reductions above 65 Mt CO2eq can only be reached with an all-input emission tax.
For low tax rates, there would be no considerable efficiency loss as a result of reducing the number of taxed items. For high tax rates, global mitigation costs for the same mitigation target might diverge by 10 billion dollars. For illustration, lets assume again a farmer with 100 ha and 200 kg ‘new’ fixed

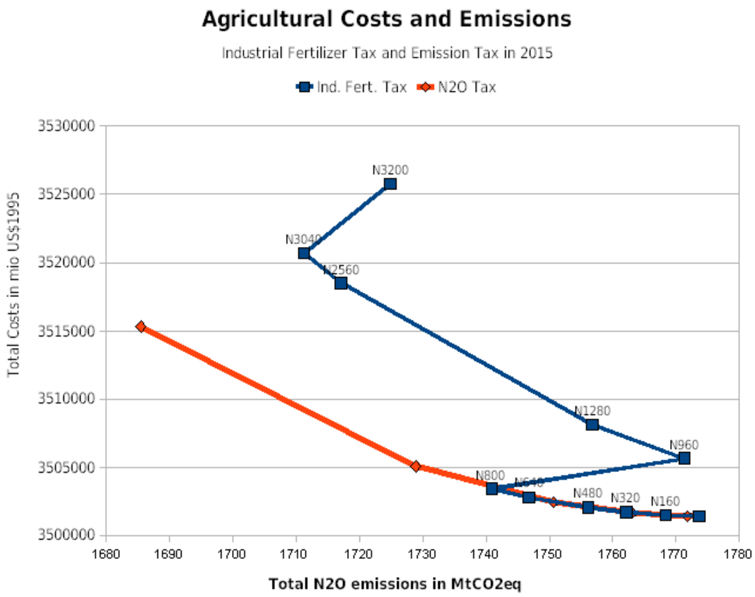

Figure 4.7: Average and total abatement costs for an industrial fertilizer tax and a N2O emission tax

nitrogen inputs per ha. His N_2O-emissions of $\approx$ 1t CO2eq,[1] could not justify a nitrogen-fertilizer tax with a high tax-rate, if extra policy implementation costs of a farm-level tax were higher than 200-300 US$$_{1995}$.
The cost difference decreases when dynamic efficiency improvements can be reached and reduce the costs of both taxes.

4.3.5 Distributional Effects

As indicator for the distributional effects of the industrial fertilizer tax, food price changes and tax income were observed.
The impact of fertilizer taxation on production costs is rather low, raising the costs only by 1-2%. Yet, as food prices are determined by the costs of the last unit produced, the change of fertilizer price might affect this last differently than the tons which can be produced more cheaply.
Food price changes are homogeneous between different crop types. Prices for cereals, roots&tubers and fibers and especially oilcrops rise; prices for rice and ruminant meat mostly fall. Regions are affected differently by price changes, some regional prices might even develop into an opposite direction. Rates of changes range mostly between +15 and -10%. For ruminant SAS shows over-proportionate price decreases, while AFR has over-proportionante price decreases in Non-Ruminante Meat(see Figure A.2 in Annex A.2.
To make the impact on average food prices more visible, the prices were weighted

[1]Assuming average N2O emissions of 4.6% of 'new' fixed nitrogen(not recycled) according to Crutzen *et al.* (2007).

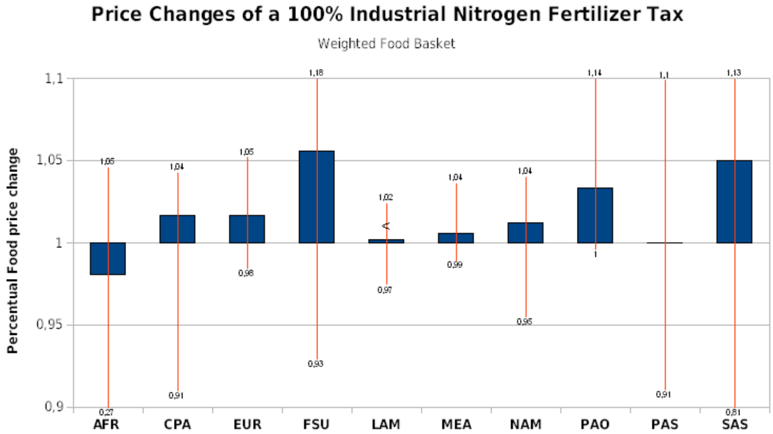

Figure 4.8: Price changes of demand-quantity-weighted food baskets for world regions (own calculations). The red line indicates the minimum and maximum price changes of a single demand category (tax rate = 100%, 2015).

with average demand of world regions in GJ food energy. Average price changes now become more modest and range -2% for AFR und +6% for FSU (see figure 4.3.5). The least developed world regions (AFR, PAS) are affected hardly or even positively; while transition countries (CPA, FSU, LAM, MEA, SAS) seem to be far more affected. Developed regions(EUR, NAM, PAO) are moderately affected.
The lower prices in AFR are counterintuitive: while production quantity and production costs increased in AFR, prices fell. This has to do with a different growing pattern: groundnuts and soybeans (which are able to fix nitrogen) are planted on a larger area with lower yields, while non-fixing plants like cassava is produced more intensively on smaller area. While the average costs of this production pattern are higher, the costs of the last units produced are smaller. The global tax income, defined as the quantity of industrial fertilizer times the tax, goes up from approximately 34 billion US$−1995 at a tax rate of 50% to 67 billion US$$_{1995}$ at a tax rate of 100 % and 234 billion US$−1995 at a tax rate of 400%. If the tax was raised in the countries where the fertilizer is applied, roughly two thirds of the tax income would go to developing countries (AFR, CPA, MEA, LAM, PAS, SAS), especially in Asia.

4.4 Result Comparison

Comparing model results with other studies may indicate the dimension of accuracy. If available, model results should be compared to real measured data. For many values, this data is not available - because the data cannot be collected because technical limitations do not allow for direct measurements (e.g. no emission measurement by remote sensing), because data simply has never been collected by statistical authorities; or because the model analysis a fictional scenario (like the introduction of a global industrial fertilizer tax) which never existed in reality.

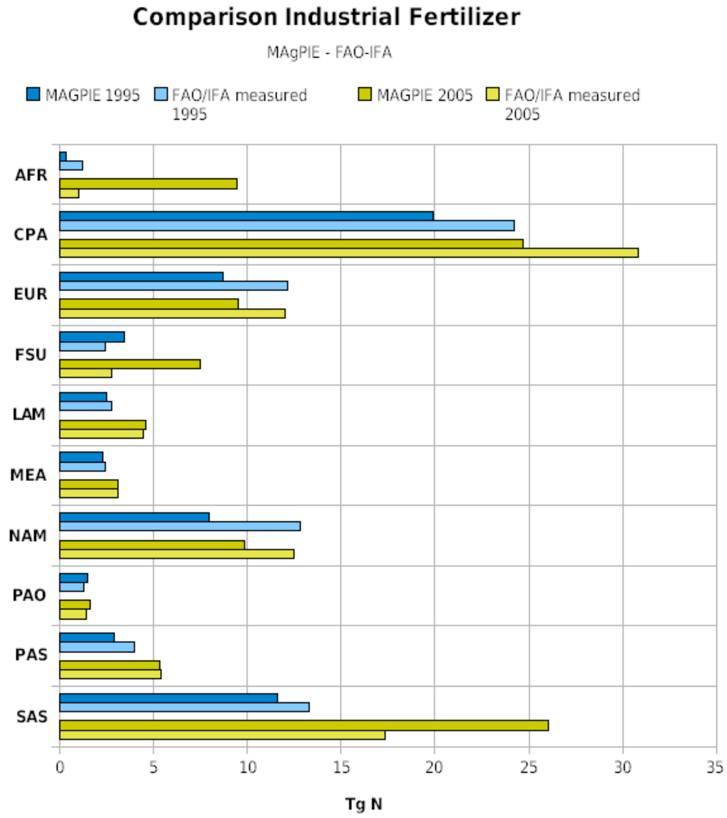

Figure 4.9: Result Comparison: Industrial Fertilizer Consumption per region in 1995 and 2005, recorded by International Fertilizer Association (IFA) (2009) and estimated by own calculations

In such cases, results can only be compared to other models. This procedure can help to improve model accuracy within the model framework; however, model comparison cannot exclude errors within the underlying theory. For example, when the IPCC Guidelines for National Greenhouse Gas Inventories are inaccurate, a model comparison of two models based on these guidelines might deliver similar results which are yet both faulty.

For this study, the results of the baseline scenario were compared to collected statistical data about industrial fertilizer consumption in 1995 and 2005 and to FAO projections for 2015, to nitrogen flow estimates from Smil (1999) in 1995 and to N2O emission estimates of US-EPA (2006a) in 1995, 2005 and 2015.

While most of the inputs and withdrawals are not recorded in official statistics, this is not the case for nitrogen fertilizer, where consumption is published in statistics of the International Fertilizer Association (IFA). In figure 4.4 we compared the regional fertilizer consumption for 1995 and 2005 as recorded by IFA with our own calculations.

A FAO study tried to estimate fertilizer consumption in 2015 using three differ-

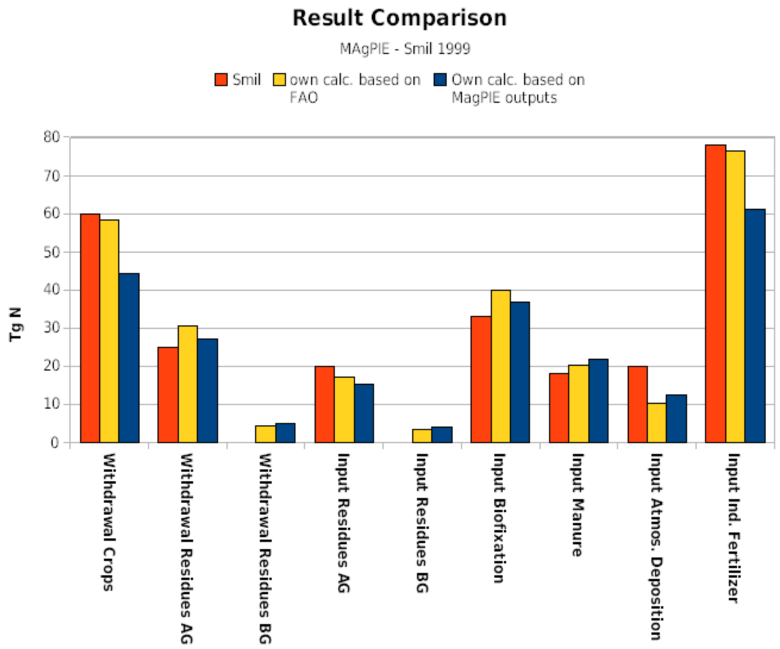

Figure 4.10: Result Comparison: Nitrogen fertilizer withdrawals and inputs according to Smil (1999), according to own calculations based on FAO-data on agricultural area and production, and according to own calculations based on MAgPIE outputs.

ent methodologies that are based on regressions. Depending on the methodology used, they derived a world industrial fertilizer consumption of 88.0, 100.4 and 106.3 Tg Nr. The estimate of this implementation lies with 105.8 Tg Nr within the range of these results.

The model-results of 1995 were compared with those of an influential study of Vaclav Smil (Smil, 1999) (see figure 4.4). Smil uses a similar methodology to calculate the global nitrogen budget. For biofixation, this implementation even uses the plant-specific nitrogen fixation of Smil(see section 3.2.1),[2]. For the estimate of returned crop residues some parameters of Smil were used, too (see section 3.2.1). First, Smil's estimates were compared to calculations based on the nitrogen-methodology of this thesis, but using FAOSTAT data on outputs and on area harvested instead of MAgPIE outputs; here, the values of Smil and of this study are quite similar. Second, the estimates were compared to calculations based on the nitrogen-methodology of this thesis using MAgPIE outputs for the area harvested and agricultural production: here, the withdrawals by harvested crops are significantly lower than the two other estimates. The low estimates for industrial fertilizer input simply reflects this underestimation of nitrogen-withdrawals.

It is difficult to compare the emission results to other studies, as the most

[2] different levels of nitrogen fixation occur due to different data of harvested area and different crop categories

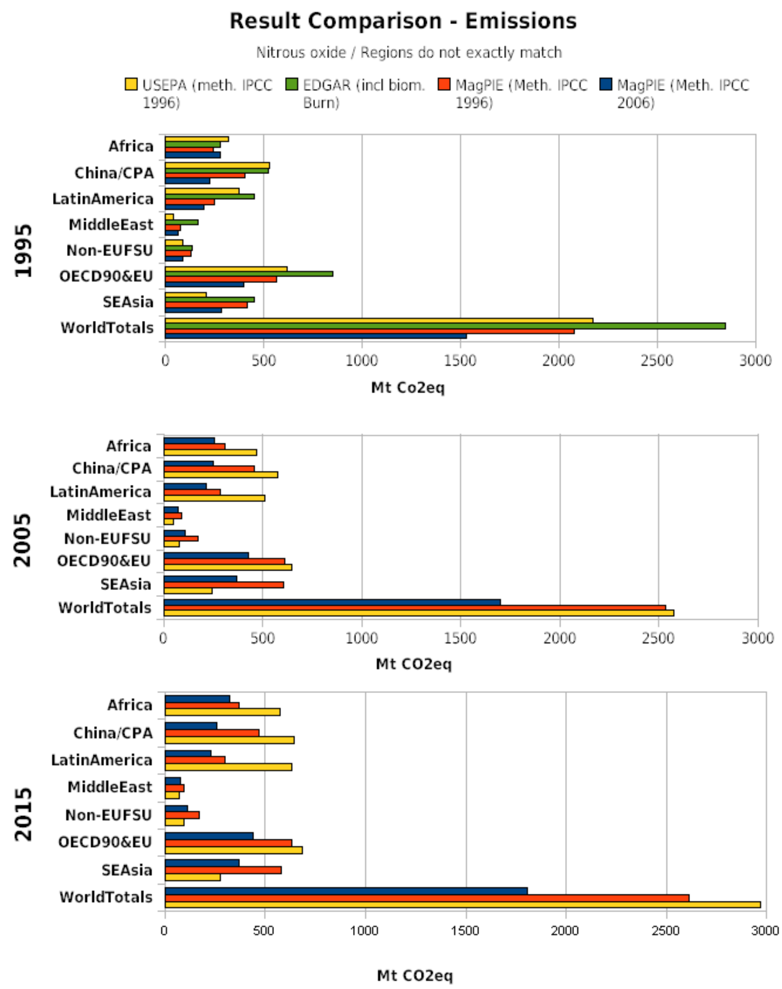

Figure 4.11: Comparison of N2O emissions in 1995 between (US-EPA, 2006a), and own estimates with the IPCC 1996 and the IPCC 2006 methodologies.

important studies in the past still used the 1996 Guidelines for National Greenhouse Gas Inventories (IPCC, 1996). There were significant changes between this ten-year old methodology and the current one from 2006 (IPCC, 2006), which was used for this implementation. For example, biofixation is now no longer considered to have an impact on GHG. Further, the parameter for direct emissions was lowered from 1.25% to 1% of applied nitrogen and may even lie as low as 0.9%. The amount of nitrogen volatilizing in AWMS was augmented significantly, and the emission factors for AWMS were changed. Most of these changes lead to lower estimates of GHG emissions in agriculture (IPCC, 2006). To make this study comparable to other studies, the 1996 methodology was also implemented.[3]
In figure 4.4 one can see a comparison between the estimated emissions of MAgPIE with the 1996 and 2006 methodology, the emissions estimated by an influential study of the United States Environmental Protection Agency (US-EPA, 2006a), that assembles all country-reporting on GHG, and the EDGAR emission database, which uses a methodology similar to IPCC (1996), yet with slight adjustments for certain emission factors (Netherland Environmental Assessment Agency, 2009; Olivier & Berdowski, 2001). The results of the MAgPIE model had to be adjusted and aggregated from MAgPIE regions to their closest counterparts of US-EPA regions, to make the results comparable.

The MAgPIE model-results are very similar to US-EPA (2006a), when the old 1996-methodology is used. The estimates mostly diverge for Latin America, where emissions of US-EPA (2006a) are assumed to be higher, and South-East Asia, where emissions of US-EPA (2006a) are lower than estimates by this study. With the current and more precise methodology of 2006, MAgPIE estimates for emissions are much lower. Estimates fall especially in China, South-East Asia and OECD countries, and are in total approximately on third lower than the emissions with the old methodology.
For 1995, the results were also compared to the EDGAR emission database. In contrast to the 1996-methodology implementation of the MAgPIE model, EDGAR also includes burning of savannas into emission inventories. This might be one of the reasons, why EDGAR denotes higher emissions than the MAgPIE model. Yet, also for OECD countries, where no emission-burning occurs, the EDGAR database registers much higher emissions than both US-EPA (2006a) and the results of this thesis.

Finally, one may compare the estimates of this study to the top-down emission estimate of (Crutzen *et al.*, 2007). His estimate for 1995 lies at $\approx$2000-2700 Mt CO2eq of agricultural N_2O-emissions. While the estimate of this study using the old methodology lies within this range(yet at the lower end), the estimate using the new IPCC 2006 methodology is significantly lower.

[3]See Popp *et al.* (in review) for a detailed description of the inclusion of the 1996 methodology in MAgPIE. Yet, while Popp *et al.* (in review) uses a regression to determine the amount of industrial fertilizer applied, the amount of industrial fertilizer for this implementation was determined with the balanced-budget approach described in section 3.2.1

Chapter 5

Discussion

In the last two chapters, the design of the MAgPIE model and of the model-expansion were made transparent, and the model outputs were illustrated and compared to other studies.
Yet, the model-outputs should not stand on their own, because the underlying assumptions, theory and data characterizing the model bias the results. It is the aim of this chapter to explain the main limitations of the model and to embed the outputs in a broader context. The first section discusses the optimization approach used in the model, and how this approach biases the results. Then, the implications of a limited action-space available to the optimizer are explained. The scope of the model is subject to the third section, and the material-flow approach of the model-extension is discussed in the fourth section. The last section of this chapter shortly explains the existing data uncertainties and reflects on the main data sources of the model. Under consideration of these model limitations, the next chapter finally presents the conclusions of the study.

5.1 Optimization Approach

MAgPIE uses the model-framework of an optimization model. This 'social planner' approach minimizes total cost of production to reach a given amount of agricultural demand. As the target system of the MAgPIE model is the agricultural sector of the real world, this approach implies that a cost-minimizing world is a good approximation of the world we live in. Underlying this approach is a distinct view of social institutions (markets, states, cultures): First, it is assumed that social institutions possess an internal dynamic toward an efficient outcome; second, it is assumed that they can also reach this outcome and restore full efficiency.
It can be illustrated that these assumptions are simplifications by discussing the example of markets, which are probably among the most important social institutions. It is assumed that the 'invisible hand' of the market will coordinate the individuals in such a way that every single agent who is maximizing his own utility is unconsciously channeling the resources into their most efficient deployment (Smith, 1776). This way the economy as an entity is assumed to behave like an individual agent, a social planner optimizing his target function. Yet, this theory is far from being uncontroversial:

First, one might criticize that individuals are not the rational cost-minimizing agents who they are implicitly assumed to be in the model. They might act irrationally or bounded rationally, as many behavioral studies show.[1]. Furthermore, both consumers and farmers may have other preferences than just low-cost food (see for example (McGregor *et al.*, 1996)).

Second, it can be criticized that one cannot draw conclusions from single individuals' preferences to the behavior of a social organism (Colander *et al.*, n.d.). The market mechanism that may deliver this coordination is certainly not omnipresent. Many regions of the world do not have well-established markets, especially in the agricultural sector, where self-sufficiency is still dominant in many less-developed regions. Even where markets exist, these markets do seldom fulfill the conditions required for an optimal allocation of resources.[2] As Stiglitz & Greenwald (1987) state, "like the emperor's new clothes, we may not be able to see the invisible hand because it is not there; or perhaps more accurately, because it is so invisible, we do not see how palsied it is". Markets do not only fail sometimes in obvious cases like the labor market, but "certainly they must be mal-performing in more subtle ways much of the time"(Stiglitz & Greenwald, 1987, p.477). Market failure is thus almost omnipresent and models should try to include other socio-economic, political, institutional and cultural dynamics which determine the outcome of the 'macro sphere' (Colander *et al.*, n.d., p.8.). Simply assuming some kind of stochastic irrationality or implementing some hard constraints as done by recent studies will not sufficiently reproduce the real world. Yet, an appropriate micro-foundation is not available up to date and remains a task for further research (Colander *et al.*, n.d., p.9).

A consequence of the optimization modelling approach is that inefficiencies cannot be explained, but have to be included into the model via restrictive constraints or calibration. The neglect of any other kind of macro-sphere also makes the model incapable of simulating important phenomena like short-term famine, food-price speculations, and distortions of competition due to political lobbying, and dynamics of trade-politics.

It is also an open question, if the current calibration is sufficient, when a new institution (a tax system) is introduced. In the model, the tax will change the optimal allocation pattern also for small price changes. This might not be realistic, as low tax levels may not even be visible to the farmer. Furthermore, institutional, psychological or other barriers may hinder the change of behavior, especially if the incentive to do so is not large enough. For example, agricultural research and plant breeding has a large return on investment, but is not undertaken in sufficient extent (Nagy & Furtan, 2008; Ayer & Schuh, 1972; Evenson, 2000). The reaction on a tax might thus be lower in the real world than in this economic model.

Furthermore, a model which does not account for other dynamics might not appropriately judge the impact of policy options: in the context of imperfect

[1]see for example (Lo *et al.*, 2005; Coates & Herbert, 2008) for emotional and hormonal reactions of stock-market traders

[2]According to Downing *et al.* (2001b), these requirements are: all markets are perfectly competitive; markets are comprehensively established in the sense that all current and future property rights are assigned; marketed goods are exclusive (ownership is singular and well defined) and transferable (goods can be bought, sold, or given away); the underlying social and legal systems guarantee that property rights are (reasonably) secure; there are no transaction costs involved in creating and/or maintaining any current or future market; there is perfect and complete information on all current and future markets.

markets, 'imperfect' policy instruments might have a better performance than instruments designed for perfectly competitive markets (see for example Goulder & Schneider (1999) for counter-intuitive effects of subsidies on other policy interventions).

Despite these biases, the optimization approach can be considered a 'Galilean idealization' which may still allow for good insights into the target system. Even though current social institutions may fail to reach optimality concerning the efficient allocation of resources, there are mechanisms which lead to continuous efficiency improvements: either through conscious adaptation of best practice and the development of rational rules (Vanberg, 1984), or through evolutionary cultural evolution (Hayek, 1973, 1978, 1988).

The advantage of the optimization approach is that it is not bound to particular institutions like the market. Centrally planned economies or subsistence farming may have a similar function: to reach an increasingly efficient outcome (even though some institutions may do a better job in reaching this target than others). While markets play a large role in developed countries, they are less central to many developing countries. Therefore, a global model should be detached from particular institutions, which is the case for the MAgPIE model. Nevertheless, the MAgPIE model allows to simulate market parameters: shadow prices for agricultural products, water or nitrogen can be obtained over the cost changes when constraints are marginally tightened.[3] This allows estimates of prices even in regions where no markets exist.

5.2 Action space of the optimization process

The optimization outcome is heavily determined by the pre-defined action space of the optimizer. The action space of the ideal-type social planner is set up by all parameters which can be influenced by individual agents and delimited by a set of constraints.

When a model is built, the definition of this action space significantly influences the results: the fewer variables can be influenced by the model's social planner, the smaller is the optimization potential; the fewer constraints are introduced, the higher is the optimization potential. Again, idealization leads to the effect that influential parameters and constraints are not completely depicted. As Friedrich Hayek discussed already, the true possibilities which are open to an economy cannot be fully identified by a top-down analysis (Hayek, 1945).

In the MAgPIE model, there are numerous idealizations of the action space. To name a few:

- As will be explained in further detail later, the current model optimizes the global land-use patterns of the world assuming 2178 production areas (cells) with homogeneous yields. A better spatial resolution of the model would increase the action space of the social planner considerably. One cell has currently the size of approximately 300 by 300 km. If the cell size is reduced to 1km^2 (which comes closer to the area over which a farmer makes decisions) one previous cell now equals 90'000 cells. The social

[3] For example, it can be observed what effect an extra-cubic-litre in a cell would have on total agricultural costs. This cost change equals the price of a cubic-litre of water in a competitive market.

planner could now relocate production to the most productive parts of each previous cell and satisfy the demand on the basis of the cultivation of a much smaller area than in the former case, where all parts of a cell had the same yield.

- In the MAgPIE model, each crop type has a fixed yield within one grid cell. Yet, in reality, farmers have the possibility to enhance or slacken their production by using more or less factor-intensive production techniques. They can also substitute between different types of input factors. If the social planner in this model could not only choose the types and amount of production, but also the intensity and production technology, his action space would increase considerably.
- In the current nitrogen implementation, MAgPIE's social planner can only influence the mix of nitrogen inputs. He cannot influence the nitrogen efficiency. Yet, as described earlier, there are multiple ways of diminishing reactive nitrogen losses. For example, the model holds the shares of animal waste management systems (AWMSs) constant. Yet they may change both over time, for example when countries adopt more 'efficient' ways of animal production (e.g. factory farming) (Fiala, 2008); they may also change due to the impact of a policy measure, which burdens different types of AWMSs differently.

These simplifications bias the results. Although the points mentioned above seem to indicate that the model is rather underestimating the action space of its social planner and thus limits his optimization potential, an uncalibrated version of MAgPIE can settle the global demand with an agricultural cultivation area which is just a small fraction of real measured area. This can be attributed to the fact that not enough constraints where included; it can also be attributed to the supposition that the current world is not an ideal one and markets do not necessarily deliver efficient outcomes, as was explained in section 5.1.

5.3 Computability

The model is based on polynomial integer inequations with a non-linear polynomial target function to calculate the optimal feasible solution. The problem belongs to the NP-hard complexity class (Garey & Johnson, 1978). There is – most probably – no efficient algorithm for finding the global optima of such a problem. Solver algorithm can only deliver local minima. For solving MAgPIE, the solver algorithm "CONOPT" was chosen, which is specialized on solving models with non-linear constraints (Drud, 1994).
As was discussed in the previous section, the model could deliver more precise results if it was more disaggregated. However, there is a trade-off between higher precision and computability of the model. Even though data for higher resolutions is available, the calculation capacity of current computers and the mathematical algorithms for solving the optimization problem are not sufficient for reaching a result within an appropriate timeframe.

5.4 Scale and scope of the model

The MAgPIE model simulates the global agricultural sector with a focus on the supply side, and the model-extension for this study only simulates nitrous oxide emissions. The model thus simulates only particular subsystems of the real world. Even when interactions may be considered low, no subsystem is isolated from the total system. The ceteris paribus condition for the rest of the system is usually a 'Galilean idealization', which biases the results. Furthermore, even when a subsystem is simulated in the model, this does not imply that it is included exhaustively. These aspects are discussed in the following for three aspects that seem most important to the author: interactions of the agricultural sector with the economy as a whole and with the climate, the importance of demand side simulation, and the interactions between different types of pollution.

- The agricultural sector is rather small in industrialized countries and may have low impact on the economy and society as a whole; yet this is certainly not the case for many developing countries. Here, 75% of the population still lives in rural areas, and the agricultural sector is the dominant one in the economy (World Bank, 2008). Shifts in agricultural production will have effects on income, population growth and diets; factors which were assumed to be exogenous in the MAgPIE model. To simulate these interdependencies, a coupling of the MAgPIE model with the Remind-Model (Bauer *et al.*, 2008) is planned, but not yet realized.
  Even though MAgPIE is a model with a focus on agriculture and climate change, the impact of climate change on agriculture was up to now not included into this study. Agricultural production will probably change significantly due to climatic change and also because of the in recent times frequently debated effect of carbon dioxide fertilization (Lobell & Field, 2008). The interrelation between the agricultural sector and the climate shall be included into the MAgPIE model in the near future.

- The current state of the MAgPIE model is mainly focused on the supply-side and neglects the demand-side. One large improvement of the MAgPIE model would be a finer disaggregation of the demand for agricultural products:
  First, up to now food calorie demand is set constant and price-inelastic. It is not considered that caloric intake, especially in developed regions, is well above the metabolic requirements (Smil, 2002) and certainly also price-elastic (USDA, 2009a). On the other side, the price-inelastic demand curve in the model defines away hunger and malnutrition due to food price increases.
  Second, substitution between plant and livestock products can only take place within demand categories, leaving only small possibilities for substitution. Yet, in the long run, significant changes in diets are thinkable. A shift from livestock products to a vegetarian or vegan diet in particular offers large potential: direct and indirect emissions from livestock production, including nitrogen applied on fodder crops, animal waste management systems, manure application on agricultural soils and pasture land, methane emissions from ruminants or soil degradation, cause a large part

of total agricultural emissions (for a comprehensive study of livestock's tremendous impact on natural systems see Food and Agriculture Organization (2006)). A MAgPIE simulation of diet shifts was undertaken by Popp *et al.* (in review), indicating that 51% of anual agricultural non-CO2 emissions could be saved in 2055 if livestock production was reduced by 25%. However, one also needs to consider that livestock forms an integrative part of agricultural production systems in many developing countries. Especially on poor soils, livestock is often the only possibility for attaining sufficient proteins.
Third, other nutrients beside calories are neglected. Yet, the selection of a diet certainly depends not only on the caloric value of the food, but on a balanced combination of all nutrients. While the total demand for food energy was satisfied in the baseline run of the model, the nitrogen withdrawals were far lower than estimates based on measured production data (see section4.4). This indicates, that proteins represent a second important factor for food demand. As long as they are not considered, crops with high energy content but low nitrogen content will be overrepresented.

- The current model only includes the effects of the agricultural system on N2O emissions. While this focus has no effects, as long as the model only forecasts business-as-usual results (at least if no feedbacks on agricultural productivity or demand occur), it has an impact on the evaluation of policies. There might be synergetic or antagonistic effects, when other pollution types are included.
As was explained in the theory chapter, there are numerous other negative effects of abundantly available reactive nitrogen. These have to be included into cost-benefit-analysis of policy measures, as the joint effect might lower the costs of pollution abatement significantly.[4] Furthermore, the production changes caused by a nitrogen tax may even influence other agricultural non-nitrogen externalities. E.g. the methane-emissions occurring in rice-paddies more than outbalance the global warming benefit of lower N_2O emissions in rice paddies (Hou *et al.*, 2000).
Finally, the model may also have effects on emissions in other sectors, especially in the Land-Use sector. Higher nitrogen prices may increase land-expansion out of two reasons: During the conversion process of forests or wetlands to agricultural area or pasture, the decomposition of biomass sets free reactive nitrogen which accumulated over years. It is integral part of many agricultural systems to use these one-off nitrogen injections, for example by slash-and-burn or shifting cultivation practices. Furthermore, extensive agriculture on large areas with low yields can increase the insertion by atmospheric deposition, natural and anthropogenic nitrogen fixation. Land-expansion, deforestation and wetland drainage on the one-hand set off further GHGes, but also have a number of other impacts like destroyed livelihoods and biodiversity losses.
If further types of pollution are included into the analysis, the effectiveness of policy instruments may thus both rise or fall.

[4] Yet, the connection between different forms of nitrogen pollution may be very different and even antagonistic in some cases: fixing one form of Nr pollution often creates another." (Sutton *et al.*, 2009, p.3)

5.5 Material Flow Approach

The model approach underlying the nitrogen extension is a classical 'bottom-up'-approach. It tries to cover the most important agricultural activities which deliver or detract nitrogen to/ from the field. The idea that all nitrogen withdrawn had to be inserted first is based on a physical relationship. Because every activity is connected to an (opportunity) cost, the model allows for cost calculations for different input-output mixes. The approach allows for a high disaggregation and a detailed description of the production function and can be enriched (at least in principle) with locally available micro-data, which is difficult to integrate into top-down models (Perman *et al.*, 2003, Chapter 6).
A usual critique of top-down models is that they do not account for changes in relative prices, if adjustments in the macro-system occur (Perman *et al.*, 2003, Chapter 6). Yet this critique is not valid for the model used in this thesis, as the material-flow approach is integrated into the optimization model, which adjusts shadow prices endogenously. This way it combines the accuracy of the bottom-up approach with dynamic macro-effects.
Another critique is that input-output relationships of certain activities are held fix, allowing for no innovation(Perman *et al.*, 2003, Chapter 9). For example, livestock manure is always managed in the same way within a region. Certainly, the model could be more disaggregated, accounting for several AWMS-options for every region. Nevertheless, it could not handle innovation, learning effects or other dynamic technology improvements. Yet, the inadequate forecast of endogenous technological change is a problem which exists in most current economy-climate models, both top-down and bottom-up (van der Zwaan *et al.*, 2002).
Furthermore, the neglect of certain agricultural flows might bias the model. For example, in our case, Nr losses and Nr-inputs from land-use change were not considered. Because it was assumed that the difference between inputs and withdrawals equals the nitrogen losses, nitrogen-efficiency is over-estimated in regions with land-use change. Thus, these regions have, in the model, too low costs for expanding their nitrogen-inputs and react less to nitrogen-tax increases. Also, the assumption of an uniform rate of loss for all nitrogen-inputs is a rather crude simplification. Losses for manure are higher than losses from industrial nitrogen fertilizers, while losses from biofixation occur almost only indirectly, over crop residues (IPCC, 2006).

5.6 Data quality

One aspect of modelling, which is of special importance, is data collection. The process of data collection simplifies the target system to a set of parameters. For example, it creates a combination of numbers and units out of the actually harvested cereals of a country. This practice relies on a large number of assumptions. These begin with the definition of the measured parameter: What is a homogeneous good? How can wheat, rye and triticale be merged into one parameter? Is a cereal harvested at milk-ripe comparable to a cereal harvested at full-ripe? They continue with the method of quantification used: is the quantity produced actually measured, estimated statistically or the result of an expert-guess? Furthermore, measurement errors are unavoidable. In social science, the data quality can not only be diminished by systematic physical measurement errors, but also by deliberate lies and the suppression of information: e.g. tariffs and taxes give incentives to underestimate the produced or exported quantities, sales prices are often guarded as corporate secrets, and governments might polish macroeconomic statistics to improve their legitimacy. Furthermore, economic data often comes from unique phenomena, and measurement cannot be repeated as in the natural sciences, when for example the velocity of light is estimated (Morgenstern, 1963).

Compared to natural science, data quality in social sciences is of low quality, and in most cases no data exists at all (Lucht & Jaeger, 2001). When creating a model, it is thus not sufficient to find an elegant theoretic idealization of the target system with a low number of parameters, but one also has to find real data which can be inserted for these parameters. If this data is not available in sufficient quality, creative workarounds are required. Often the input data of one model is adopted from the output data of another model, which again depends on the output of yet another model. Yet, together with this data, the latter model also inherits all assumptions of the previous model.

In the case of the MAgPIE model, data sources mainly include FAOSTAT (2008), the LPJ model() and the GTAP model(). FAOSTAT is a global agricultural database, which publishes datasets mainly received from country statistics offices, which in turn are based on local surveys, censuses, administrative records or other data collection processes (Kasnakoglu, 2004). Accuracy for FAOSTAT figures is rather high for high-income countries, with errors mostly smaller than 5%. For low-income countries, data gaps had to be filled, which lowers accuracy significantly. Furthermore, shadow markets (especially home gardens and backyard plots) exist, which are not depicted in official data. Lastly, there seems to be an incentive to submit underestimations of farmland figures. China's real farmland area is estimated to be 50% larger than its official claim.(Smil, 1999).

The LPJ model delivers the cell-specific yields for major crop types, which are derived from 'crop functional types'(CFT). These CFTs simulate potential yield levels, mainly based on hydrological, climatic and weather conditions. One of the main biases of the model is that it does not include soil degradation and management factors (plant breeding, fertilizer and pesticides, machinery) apart from irrigation into the calculation of potential yield levels. While yields are underestimated in regions with intensive management (e.g. the US Central Plains or the Australian wheat belt), global yield levels are overestimated (Bondeau *et al.*, 2007). The model-output was calibrated with FAO data to meet the FAO yield levels per country before it was implemented into MAgPIE.

The GTAP model is currently the largest trade data base available. It is difficult to judge the accuracy of the datasets, because model structure and data sources are rather intransparent. According to Mitra-Kahn (2008), GTAP uses its market power to restrict output and increase data prices, which severely limits the scientific value of the project. Still, it is known that in order to balance out the social accounting matrices of the model, data has to be adjusted, which may significantly alter the results. For example, Mitra-Kahn (2008) reports a case of Mozambique's economy, where this 'benchmarking' process led to an adjustment of agricultural producers income by 58 million dollar, which corresponds to a change of 4.4 US$ per agricultural producer in a country where 38% of the population lives from less than 1$ per day.

The data used to estimate nitrogen flows in agriculture is not based on a single

database, but on numerous single studies. Smil (1999), who executed a similar investigation of nitrogen flows, estimated that aggregated data of flows may have a range of $\pm 25\%$. The uncertainty range for most of the values of this study should reside in the same scale. Yet, there is one parameter where accuracy has to be considered far lower, namely nitrogen efficiency. As was already discussed in section 3.2.1, nitrogen efficiency is over-estimated for countries with high land-use change, and it does not account for variations in nitrogen-efficiency between different types of inputs and different types of crops. This leads to a systematic bias of model results. E.g., it under-estimates the efficiency of nitrogen-fixing plants, which may lead to an underproportionate shift from industrial fertilizer to biofixation when a tax is imposed, and thus to lower emissions. Beyond, it over-estimates the efficiency of the least developed countries with high rates of land-use change. This leads to an under-estimation of the impacts a nitrogen-fertilizer tax has on production costs and food prices.
Finally, the emission parameters derived from IPCC (2006) are subject to large uncertainties. The main emission factor for direct emissions of Nr applied on agricultural soil, EF_1, ranges over several orders of magnitude in field studies(Smil, 1999), and even for aggregated values uncertainty of N_2O-emission ranges from 0.3 to 3% of applied Nr. The best guess value was revised from 1.25% in the 1996 methodology to 1% in the 2006 methodology and may even lie at 0.9%. Uncertainties for other emission parameters reside in a similar or even larger range (IPCC, 1996, 2006). The fixed emission factors may bias the results, when nitrogen-efficiency improvements occur: According to (Snyder *et al.*, 2009), emissions depend less on the total quantity of nitrogen applied, but on the share of nitrogen which is not incorporated into plant biomass and lost to the environment. This share will also be reduced due to efficiency increases.

This chapter made clear that model-outputs are subject to numerous limitations, biases and uncertainties. If one takes into account that 'macro-scopes' like the MAgPIE model reduce the complexity of the global agricultural system to nothing but a few equations, it seems obvious that this has to be the case. On the other side, such models have a value beside one of pure amusement (which they certainly have, too). They can help to structure thoughts about a certain subject and to formulate problems in a logical (mathematical) language; they can impart an impression of the scale, in which best-guess values reside; and they can become archetypes for more sophisticated future models, which are definely required in consideration of the large global threats which await humankind within the next centuries.
Despite the large limitations and uncertainties of the model, certain soft conclusions can be drawn from this study, which will be revealed in the following chapter.

Chapter 6

Conclusion and Further Research

This study indicates that nitrous oxide emissions from agriculture increased strongly within the last decade and will continue to do so in the near future. Hence, the agricultural sector will continuously be one of the main drivers of global warming and stratospheric ozone depletion.
So far, current institutions failed in regulating this externality. Yet, there are instruments at hand which could help to reach a sustainable level of pollution. One of these instruments is the taxation of industrial nitrogen fertilizer. By increasing the price of nitrogen fertilizer, this instrument intends to shift nitrogen inputs from polluting chemical fertilizers to less polluting substitutes, and gives incentives to improve nitrogen-efficiency.
This study investigated the advantages and disadvantages of the industrial nitrogen fertilizer tax, using five indicators: environmental efficacy, static and dynamic efficiency, policy implementation costs and distributive effects.

The results of this study indicate, that this policy may be effective in reducing nitrous oxide emissions. However, estimates are highly uncertain:
On the one hand, when dynamic improvements of nitrogen-efficiency are taken into account, emission savings can turn out much higher. First, emissions will decrease because less nitrogen inputs are required; second, emissions depend largely on the share of nitrogen which is not incorporated into plant biomass and which is lost to the environment. This share will also be reduced due to efficiency increases.
On the other hand, when side-effects of the tax on deforestation, land use change and livestock demand are taken into account, the tax might as well have less positive or even negative impacts on the environment. The tax creates increasing pressure to take new land into cultivation, because more extensive cultivation patterns can make better use of natural fixation and atmospheric deposition, and because the land-conversion offers one-off nitrogen inputs from the decomposition of biomass. As ruminants can obtain nitrogen from pasture land and recycle it on agricultural land, the tax will lead to a substitution of other food for ruminant meat and milk. Both land-use change and livestock production are main-contributers to the greenhouse effect and key drivers of other types of pollution.

Even though its potential on N_2O-reduction is limited, nitrogen fertilizer taxation might well be a low-hanging fruit for GHG and ODS abatement from a mitigation cost perspective, especially if the large concurring effects on other types of nitrogen-pollution are taken into consideration, which make abatement of reactive nitrogen inputs much more beneficial. Furthermore, efficiency improvements do not only decrease emissions, but at the same time they lower also the costs for the purchase of nitrogen fertilizer. However, mitigation costs rise or fall with the uncertain quantity of emission-savings.

Into full mitigation costs, also the costs of policy implementation have to be included. The one-input tax has most probably far lower policy implementation costs than tax or certificate schemes on farm-level, because the number of regulated agents and regulated items is much lower. Yet, the model results show clearly that industrial nitrogen fertilizer taxation is far from being a first-best solution: emissions may even increase with higher tax levels. The origin of these inefficiencies lies in tax-avoiding behavior: nitrogen from manure is not taxed and may be a cheap substitute for taxed industrial fertilizer. This is the case for ruminants, which gather nitrogen from pasture areas, but also for non-ruminants, if they are not fed by crops grown with industrial fertilizer. This may lead to a large increase of emissions.

The largest deficit of industrial nitrogen fertilizer taxation is the clear trade-off between mitigation aims and food security. The model indicates, that food prices might be severely affected by the introduction of a tax. This is especially true for transition countries, which still use nitrogen fertilizers extensively with low efficiency. Price changes were estimated to be lower or even negative in the least developed regions Africa and Pacific Asia. Yet, as calculated production costs rise also in these countries, this indicates a loss in producers' rent (price minus production costs), which may also have negative effects on rural populations earning their income in agriculture. Furthermore, price and cost changes in Africa, Latin America and Pacific Asia might well be underestimated, as deforestation is not covered by the model as a nitrogen source. This biases the nitrogen-efficiency upwards, so that it becomes cheaper to produce with industrial fertilizer than it is in reality.
Even if it is taken into consideration that nitrogen-efficiency improvements and the demand-reaction on price increases will dampen the negative impact, a residual effect will remain.

In the author's opinion, further research has to concentrate on five important issues:
First, the high shadow price of nitrogen efficiency eventually indicates that the model with its limited action space under-estimates actual emissions. The action space of the model has to be enlarged. Disaggregation has to include agricultural activities with diverging nitrogen efficiencies, and eventually allow for endogenous nitrogen efficiency improvements.
Second, it also has to be better understood, why inefficiencies may persist in the real world. It is a difficult, yet an essential task to understand the dynamics behind them and to be able to forecast the future evolution of these inefficiencies.

Third, the impact and interactions with other types of pollution has to be depicted more comprehensively. This includes other types of nitrogen pollution (e.g. eutrophication or air pollution), other agricultural greenhouse emissions (mainly methane from rice and livestock production) and the impact of the agricultural sector on other sectors (especially land-use change and fertilizer producing industry).
Fourth, the demand-side has to be better understood. This may happen by introducing price-elasticities of demand, and by analyzing the effect of food prices on malnutrition and hunger.
Fifth, it remains the task of further research to quantify and compare comprehensively the advantages and disadvantages of the industrial nitrogen fertilizer tax in respect to other policy instruments. The inclusion of further indicators, for example concerning concerning the feasibility and enforceability of policy options or the neutrality in terms of effect on competition may be included into the analysis. This will hopefully help policy makers to select the most sustainable solution.

Some policy implications may also be deduced from this study.
First, an industrial nitrogen fertilizer tax will hardly reduce emissions below the level of 1995, even under optimistic assumptions. This will probably also be the case for other supply-side instruments. Hence, the situation requires additional demand-driven shifts toward a more vegetarian diet to actually reach a sustainable state. While these shifts are probably non-regret options, they have to be undertaken soon, because preference-shifts on the demand-side are slow. Thus, policy makers should soon take action on this issue.
Second, if a nitrogen fertilizer tax shall be introduced, the precise tax-level should be carefully chosen. Distorting effects of too high tax rates diminish the positive environmental effects of the policy and make it an inefficient instrument.
Third, also for moderate tax-levels, the distributional effects of a tax are substantial. As food price increases were one of the main reason, why the number of people on earth who suffer malnutrition crossed recently the one billion line(Alston *et al.*, 2009), the author believes that policies should in no case tolerate that this trend is reinforced. A win-win situation can only be reached if tax income is consequently distributed to low-income households.

Appendix A

Annex

A.1 MAgPIE – Model description

The following is a formal description of the MAgPIE model (in black font color) and the model extension (in red font color). The model is written in GAMS (Brooke *et al.*, 2003) and solved with (Drud, 1994).

A.1.1 Variables

x	level of activity (21 crop activities (ha), 3 livestock activities (ton), 2 land conversion activities (ha), 3 input purchase activities (US$), industrial nitrogen fertilizer (t Nr), emission certificates (t CO2eq))
yld_tc	technological change variable

A.1.2 Parameters

C	total costs of production
c	production costs per activity unit (US$1995); now also including tax and certificate costs
tcc	technological change costs
wat_tc	water-saving rate ($0 = wat\_tc = 1$)
d_food	demand for food energy (GJ)
y_prd	production output(from crops and livestock) per activity unit (t product)
y_food	food energy delivery (from crops and livestock) (GJ)
y_feed	feed energy delivery (from crops and residues) (GJ)
y_fodd	green fodder energy delivery (from crops) (GJ)
y_land	land delivery (i.e., from conversion activities) (ha)
y_wat	water delivery (i.e., from irrigation activities) (m3)
y_input	variable input delivery (i.e., labor, chemicals, capital) (US$)
req_feed	feed energy requirement (i.e., per ton of livestock output) (GJ)
req_fodd	green fodder energy requirement (i.e., per ton of livestock output) (GJ)
req_land	land requirements (i.e., cropland, pasture) (ha)
req_wat	water requirements (m3)
req_input	variable input requirements (i.e., labor, chemicals, capital) (US$)
req_share	area to be considered for rotational constraints (ha)
$land\_const$	available land (cropland, pasture, non-agricultural land) (ha)
wat_const	available water discharge for irrigation (m3)
max_share	maximum crop share in average rotation (percent)
n_inp_fert	nitrogen inputs per activity by industrial nitrogen fertilizer (t Nr)
n_inp_area	nitrogen inputs per area by atmospheric deposition and biofixation (t Nr)
n_inp_prd	nitrogen inputs per production unit by manure and residues (t Nr)
n_inp_with	nitrogen withdrawals per production unit by crops and residues (t Nr)
$emis$	N_2O emissions of nitrogen input activities(t CO2eq)

A.1.3 Indices

i	number of economic regions (10)
j	number of grid cells per region (total: 2178 grid cells (3 degree by 3 degree))
k	number of activities (21 crops (kcr), 3 livestock (kli), 2 land conversion (klc), 3 input purchases (kin))
l	number of food energy demand categories (10)
m	number of agricultural land types (3) (cropland, pasture, non-agricultural land)
n	number of rotational constraints (10)

A.1.4 Goal function

Cost minimization (Total costs of production; sum for all i regions)

$$\min_{x,yld\_tc} C \tag{A.1}$$

$$C = \sum_i \sum_j \sum_k x_{i,j,k} \times c_{i,k} + \sum_i yld\_tc_i \times tcc_i \tag{A.2}$$

Costs for tax payments and certificate purchase now also enter the goal function

A.1.5 Global constraints

Food energy demand (minimum constraint; for all l demand types):

$$\sum_i \sum_j \sum_k x_{i,j,k,l} \times y\_food_{i,j,k,l} \times yld\_tc_i \geq d\_food_{i,l} \tag{A.3}$$

(similarly for fiber)

Emission Constraint (only for valid for all-input tax to calculate policy implementation costs)
Emission certificates (which are part of the activities x) have to equal the emissions of nitrogen inputs. As biofixation and atmospheric deposition have no emissions in the model, area-dependent nitrogen-inputs are not considered in the constraint.

$$\sum_i \sum_j \sum_k x_{i,j,k} \times y\_prd_{i,j,k} \times yld\_tc_i \times n\_inp\_yld_{i,k} * emis_{i,j,k} = 0 \tag{A.4}$$

A.1.6 Regional constraints

(for all i regions)
(Note: all k activities are included in all constraints, in order to reduce the number of indices; however, many of the parameter values may be zero.)
Minimum trade balance (regional supply $\geq$ regional demand $\times$ self-sufficiency rate):

$$\sum_j \sum_k x_{i,j,k} \times y\_food_{i,j,k} \times yld\_tc_i \geq d\_food_{i,l} \times self\_sufficiency_{i,l} \tag{A.5}$$

(similarly for fiber)

Feed energy balance (regional demand $\leq$ regional supply):

$$\sum_j \sum_k x_{i,j,k} \times (req\_feed_{i,k} - y\_feed_{i,j,k}) \times yld\_tc_i \leq 0 \tag{A.6}$$

Green fodder balance (regional demand $\leq$ regional supply):

$$\sum_j \sum_k x_{i,j,k} \times (req\_fodd_{i,k} - y\_fodd_{i,j,k}) \times yld\_tc_i \leq 0 \tag{A.7}$$

Input purchase balances (regional demand$\leq$regional supply; for all kin inputs):

$$\sum_j \sum_k x_{i,j,k} \times (req\_input_{i,k,kin} - y\_input_{i,j,k,kin}) \leq 0 \tag{A.8}$$

A.1.7 Cellular constraints

(for all j cells)
Land constraints (for initially available cropland and pasture):

$$\sum_k x_{i,j,k} \times (req\_land_{i,k,m} - y\_land_{i,j,m}) \leq land\_const_{i,j,m} \tag{A.9}$$

Land conversion constraint (for non-agricultural land to be potentially converted into cropland and pasture):

$$\sum_k x_{i,j,k} \times y\_land_{i,j,m} \leq land\_const_{i,j,\text{"}non-agri\text{"}} \tag{A.10}$$

Rotational constraints (for all n constraint types):

$$\sum_k x_{i,j,k} \times req\_share_{i,k,n} \leq max\_share_{i,n} \times land\_const_{i,j,\text{"}cropland\text{"}} \tag{A.11}$$

Water constraints:

$$\sum_k x_{i,j,k} \times (req\_wat_{i,k} - y\_wat_{i,j})/(1 + (yld\_tc_i \times wat\_tc)) \leq wat\_const_{i,j} \tag{A.12}$$

Nitrogen constraints:
Nitrogen inputs are either activity specific (industrial nitrogen fertilizer) area-specific(biofixation and atmospheric deposition) or production-output-specific(manure, residues). Nitrogen inputs $\times$ nitrogen efficiency $\geq$ nitrogen withdrawals.

$$\begin{aligned}\sum_k x_{i,j,k} \times y\_prd_{i,j,k} \times yld\_tc_i \times n\_inp\_prd_{i,k} \times n\_eff_i + \\ \sum_k x_{i,j,k} \times n\_inp\_area_{i,k} \times n\_eff_i + \\ \sum_k x_{i,j,k} \times n\_inp\_fert \times n\_eff_i + \\ \geq \sum_k x_{i,j,k} \times y\_food_{i,j,k} \times yld\_tc_i \times n\_with_{i,k}\end{aligned} \tag{A.13}$$

A.2 Figures

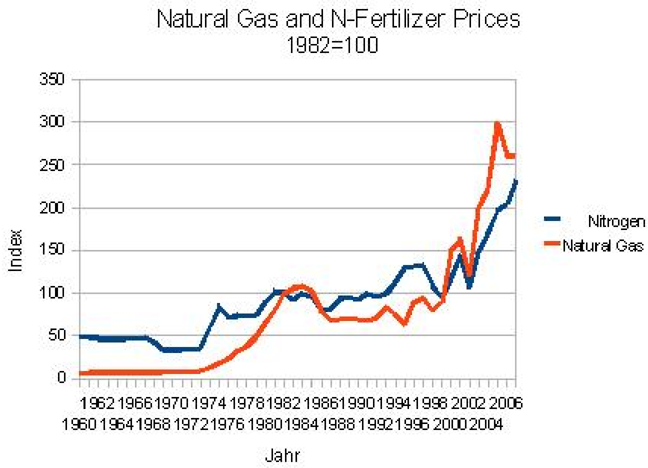

Figure A.1: Prices of natural gas and nitrogen fertilizer in comparison (own calculations based on fertilizer prices of United States Department of Agriculture (2009), fertilizer quantities of International Fertilizer Association (IFA) (2009) and inflation rates of International Monetary Fund (IMF) (2009))

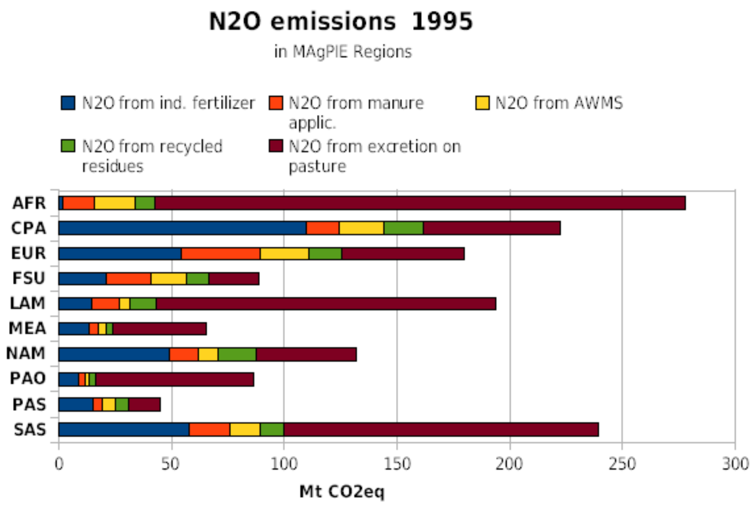

Figure A.2: Regional N_2O-emission in 1995, disaggregated into different emissions sources (own calculations).

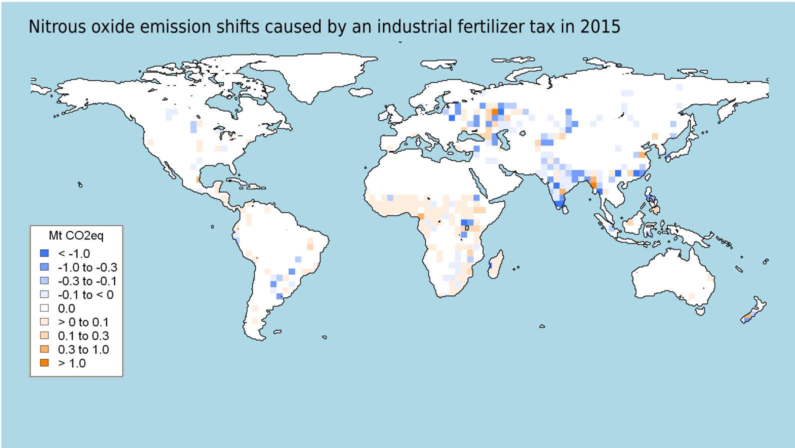

Figure A.3: Change in N2O emissions due to the introduction of an industrial fertilizer tax in 1995 with a tax rate of 100%(own calculations)

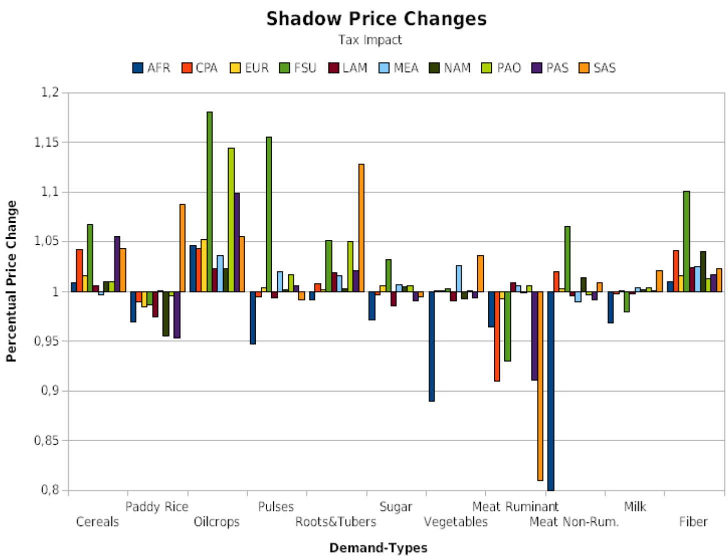

Figure A.4: Price changes of demand categories as an effect of industrial fertilizer taxation (own calculations, tax rate = 100%, 2015)

A.3 tables

		Temperate cereals	Maize	Tropical cereals	Paddy Rice	Soybeans	Rapeseed	Groundnuts in Shell	Sunflower	Other oil crops	Pulses	Potatoes	Cassava, Sweet Potatoes	Sugar Cane	Sugar Beets	Vegetables& Fruits	Gras	Cotton& Fibre
Nr in HC		1,72	1,51	3,50	1,80	4,40	3,35	4,30	2,91	3,50	4,10	0,35	0,30	0,22	0,18	0,30	0,55	2,90
AG biomass per HC	**AFR**	2,30	3,50	3,50	1,50	1,50	2,30	1,50	2,30	1,50	0,40	1,00	0,90	0,70	0,70	0,80	1,30	2,10
	CPA	1,70	3,50	3,50	1,50	1,50	2,30	1,50	2,30	1,50	0,40	1,00	0,90	0,70	0,70	0,80	1,30	2,10
	EUR	1,00	1,20	1,20	1,20	1,20	1,90	1,20	1,85	1,50	1,00	1,00	0,90	0,70	0,50	0,80	1,30	2,10
	FSU	1,50	1,90	1,90	1,20	1,50	1,90	1,20	1,85	1,50	1,00	1,00	0,90	0,70	0,50	0,80	1,30	2,10
	LAM	1,50	3,00	3,00	1,20	1,50	2,30	1,50	2,30	1,50	0,40	1,00	0,90	0,70	0,70	0,80	1,30	2,10
	MEA	1,50	3,00	3,00	1,20	1,50	2,30	1,50	2,30	1,50	0,40	1,00	0,90	0,70	0,70	0,80	1,30	2,10
	NAM	1,20	1,20	1,20	1,20	1,20	1,90	1,20	1,85	1,50	1,00	1,00	0,90	0,70	0,50	0,80	1,30	2,10
	PAO	1,20	1,20	1,20	1,20	1,20	1,90	1,20	2,30	1,50	1,00	1,00	0,90	0,70	0,50	0,80	1,30	2,10
	PAS	1,50	3,00	3,00	1,00	1,20	2,30	1,20	2,30	1,50	0,40	1,00	0,90	0,70	0,70	0,80	1,30	2,10
	SAS	1,70	3,50	3,50	1,50	1,50	2,30	1,50	2,30	1,50	0,40	1,00	0,90	0,70	0,70	0,80	1,30	2,10
Nr in AG biomass		0,50	0,70	0,62	0,55	1,50	0,70	1,47	1,00	0,53	1,50	0,20	2,99	0,47	0,40	2,00	0,55	1,30
BG per HC		0,24	0,22	0,22	0,16	0,19	0,22	0,20	0,22	0,26	0,19	0,20	0,20	0,07	0,20	0,10	0,80	0,77
Nr in BG biomass		0,90	0,70	0,60	0,24	0,80	0,70	1,40	0,70	0,37	0,80	1,40	1,00	1,00	1,40	1,30	1,60	1,00

Table A.1: Nitrogen content of harvested crops(HC), aboveground (AG) biomass and belowground (BG) biomass; ratio of AG and BG biomass to HC. All values in %, biomass is dry biomass. For sources, see section 3.2.1

	Region	Ruminants for Meat	Non-Ruminants for Meat	Ruminants for Milk
Nr Excretion recycled on agricultural land	AFR	0,019	0,410	0,011
	CPA	0,323	0,009	0,023
	EUR	0,146	0,018	0,013
	FSU	0,214	0,057	0,020
	LAM	0,008	0,025	0,030
	MEA	0,021	0,090	0,002
	NAM	0,039	0,019	0,007
	PAO	0,025	0,033	0,002
	PAS	0,325	0,021	0,010
	SAS	0,261	0,157	0,010
Nr Excretion on pasture	AFR	2,172	0,194	0,128
	CPA	0,860	0,013	0,013
	EUR	0,304	0,002	0,005
	FSU	0,225	0,018	0,006
	LAM	0,871	0,061	0,021
	MEA	1,083	0,052	0,055
	NAM	0,247	0,003	0,001
	PAO	1,028	0,002	0,014
	PAS	0,676	0,017	0,005
	SAS	1,216	0,086	0,045

Table A.2: Nitrogen excretion on croplands and pasture land, in t Nr per t livestock product. For Sources, see 3.2.1

Region	Atmospheric deposition in kg/ha 2000
AFR	5.92
CPA	11.67
EUR	9.01
FSU	3.14
LAM	5.61
MEA	3.29
NAM	6.34
PAO	2.93
PAS	11.46
SAS	23.11

Table A.3: Atmospheric deposition from NO_x and NH_y. For sources, see section 3.2.1.

A.4 MAgPIE regions

Sub-Saharan Africa (AFR) Angola, Benin, Botswana, Burkina Faso, Burundi, Cameroon, Central African Republic, Chad, Congo (Dem Republic), Congo(Republic), Côte d'Ivoire, Djibouti, Equatorial Guinea, Eritrea, Ethiopia, Gabon, Gambia, The, Ghana, Guinea, Guinea-Bissau, Kenya, Lesotho, Liberia, Madagascar, Malawi, Mali, Mauritania, Mauritius, Mozambique, Namibia, Niger, Nigeria, Rwanda, Senegal, Sierra Leone, Somalia, South Africa, Sudan, Swaziland, Tanzania, United Rep of, Togo, Uganda, Western Sahara, Zambia, Zimbabwe, Cambodia

Centrally planned Asia (incl. China) (CPA) China, Hong Kong, Laos, Mongolia, Taiwan, Viet Nam

Europe (incl. Turkey) (EUR) Albania, Austria, Belgium-Luxembourg, Bosnia and Herzegovina, Bulgaria, Croatia, Cyprus, Czech Republic, Denmark, Estonia, Falkland Islands (U.K.), Finland, France, Germany, Greece, Greenland, Hungary, Iceland, Ireland, Italy, Kerguelen (F.S.A.T.), Latvia, Lithuania, Luxembourg, Macedonia,The Fmr Yug Rp, Montenegro, Netherlands, Norway, Poland, Portugal, Romania, Slovakia, Slovenia, Spain, Sweden, Switzerland, Turkey, United Kingdom, Yugoslavia (Fed Rep of)

Former Soviet Union (FSU) Armenia, Azerbaijan, Republic of, Belarus, Georgia, Kazakhstan, Kyrgyzstan, Moldova, Republic of, Russian Federation, Tajikistan, Turkmenistan, Ukraine, Uzbekistan

Latin America (LAM) Argentina, Belize, Bolivia, Brazil, Chile, Colombia, Costa Rica, Cuba, Dominican Republic, Ecuador, El Salvador, French Guiana, Guatemala, Guyana, Haiti, Honduras, Jamaica, Mexico, Nicaragua, Panama, Paraguay, Peru, Suriname, Trinidad, Uruguay, Venezuela

Middle East/North Africa (MEA) Algeria, Egypt, Iran, Islamic Rep of, Iraq, Israel, Jordan, Kuwait, Lebanon, Libyan Arab Jamahiriya, Morocco, Oman, Qatar, Saudi Arabia, Syrian Arab Republic, Tunisia, United Arab Emirates, Yemen

North America (NAM) Canada, Puerto Rico, United States of America

Pacific OECD (Japan, AUS, NZL) (PAO) Australia, Japan, New Zealand

Pacific Asia (PAS) Brunei, Fiji, Indonesia, Korea (Dem People's Rep), Korea, Republic of, Malaysia, New Caledonia, Papua New Guinea, Philippines, Singapore, Solomon Islands, Thailand, Vanuatu

Southern Asia (incl. India) (SAS) Afghanistan, Bangladesh, Bhutan, India, Myanmar, Nepal, Pakistan, Reunion, Sri Lanka

A.5 Further Policy Options

A.5.1 Moral Suasion

Moral Suasion has the aim to dissolve the conflicting interests between agents whose utilities are connected through the polluting activity. For this purpose, policy makers try to influence the preference structure of individuals in such a way that it futhers the public interest. These programs can either target producers, consumers or stakeholders.

Taking the example of the farmer in the model mentioned above, the aim of a moral suasion campaign would be to raise the sympathy of the farmer for the posterity. Thus, the policy tries to include the social damage costs into the utility function of the farmer U_f, such that $U_f = (\Pi, \dots)$ becomes $U_f(\Pi, D, \dots)$. The farmer now aims to reduce polluting activities himself. If the farmer acts as if the foreign interest was his own, the externality is fully internalized.

For inveterate economists, the idea of farmers furthering interests of other agents seems absurd, but seeing farmers just as profit-maximizers without other preferences would do them wrong and oversimplify the picture. This notion is supported by a survey of Canenbley et al., which states that farmers continue to understand themselves as professionals of sustainable acquaintance with nature and of the production of environmental goods (Canenbley *et al.*, 2004, p. 44). This professional ethics supports the responsibility of farmers also concerning off-site effects of their work. As they do not want to infringe upon their professional self-perception, they will undertake environmental tasks also without direct governmental control.

Still, even if farmers take into consideration the interests of future generations, it is unclear to which extent. Being affected oneself gives probably a greater consciousness of distress than the empathy with others. Furthermore, price pressure and falling profits force farmers to rethink which environmental services they are willing and able to provide without compensation (Canenbley *et al.*, 2004, p. 44). They enter into a dilemma between their professional autonomy and the need for governmental support. It becomes difficult for them to stick to

their self-perception and not to exchange it for profit-maximizing "‘industrial"’ culture which is rather contradictory to a sense of community, sustainability and responsibility(Canenbley *et al.*, 2004, p. 8).
Compared to the incentive-based policies, moral suasion has rather different implementation costs and largely depends on the approach of implementation. If an environmental awareness campaign addresses the farmer, it can make use of the large biological and ecological expert knowledge of the latter and his information about the state of local sites. No costs for market transactions, negotiations and contract setting incur, and no governmental control is necessary. Yet, this is not the case if consumer preferences are changed: here principal-agent settings require monitoring and controls of farmers.
Yet, emission reductions can hardly be foreseen, as preference shifts are long-term developments which are complex and multi-causal. Badly executed campaigns might even counteract their intended aim to persuade farmers. The latters’ professional ethics makes them skeptical of ‘lay’ interference from the state or ecologists, as they perceive such interventions as constraint on their creativity and freedom of action (Canenbley *et al.*, 2004, p. 43f).
One good example for moral suasion is the organic farming movement. Organic farmers decided to abstain from industrial fertilizers and pesticides, because they could not reconcile those practices with their conscience. Later, the organic movement became increasingly demand-driven, with consumers accepting higher prices for organic products. It would be disproportionate to assume that the organic movement was initiated and arranged by the state, trying to dissolve a market failure. It rather seems to be a product of diverse cultural, economic and political movements. However it was certainly facilitated by the state, for example by increasing market transparency through labeling, e.g. the German *‘Bio’*-label or the European *‘Ecolabel’*.
Embedding values of sustainability into the professional self-image of farmers and into the identity of the consumers possesses large (but limited) potential to raise the efficiency of the agricultural system, and it can be combined to other, incentive-based instruments.

A.5.2 Socialisation

If a social planner could deliver a more efficient solution than the market, it would stand to reason to institutionalize the state as central planner of the economy. Yet, central planners are no social planners: Even democratic states do not necessarily act altruistic. They are administrated by political elites and bureaucrats which try to forward their own interest, and democratic control is not omnipresent. Furthermore, markets feature certain functions which bureaucratic designs do not hold: a price system as indicator for scarcity, the control function of competition and performance and innovation incentives. These features help to lower transactional costs and informational asymmetries and thus give market provision efficiency advantages over state provision.
Still, Brada & King (1993) compare the efficiency of private and state farms in Poland during the Cold War, and does not find significant differences. Also Johnson (1982) comes to the conclusion, that centrally planned agriculture could keep up with international standards in the 1960s and 1970s. Finally, Carter & Zhang (1994) examine nine centrally planned economies that accounted for one

quarter of global agricultural land, to come to the conclusion that the agricultural downturn in production in the 1980s was not due to inefficiencies in the production system, but due to lower inputs - mainly fertilizer. China, which started privatization reforms in the 1980s, even lost in efficiency compared to its planned neighbors.

Yet, there is also no evidence that state farms acted in an environmental more sustainable way. Sumelius *et al.* (2005) comes to the conclusion, that excessive use of plant nutrients was also a common problem in the centrally planned states. The transition process also shows no clear evidence that planned economies used nitrogen more or less efficiently. Indeed, from 1989 to 1999 artificial nitrogen fertilizer input in Eastern Europe and Central Asia decreased by $\approx$75% from 11,0 Tg N to 2,5 Tg N (International Fertilizer Association (IFA), 2009). But this has to be seen in the context of a substantial decrease of agricultural gross production which also halved from 175 to 93 Billion international dollar (FAOSTAT, 2008). As some natural nitrogen sources remain constant, and nitrogen from the previous extensive use still remained in the circle, no clear trend of nitrogen efficiency can yet be asserted.

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