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Damage Costs of Nitrogen Fertilizer in Europe and their Internalization

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ABSTRACT *This paper estimates the environmental impacts and damage costs ('external costs') of synthetic nitrogen fertilizer and discusses options for reducing these impacts, including their consequences for farmers and for producers of fertilizer. The damage costs of the fertilizer life cycle that could be estimated are large, about 0.3 €/kg_N (compared to the current market price of about 0.5 €/kg_N); much of that is due to global warming by N_2O and CO_2 emissions during fertilizer production and N_2O emissions from fertilized fields. Policy options for internalizing these costs are discussed, and the consequences of reduced fertilizer input on crop yield are explored. If the damage costs were internalized by a pollution tax or tradable permits that are auctioned by the government, the economic consequences would be heavy, with a large revenue loss for farmers. However, if it is internalized by tradable permits that are given out free, the revenue loss for farmers is small. The loss for fertilizer producers increases linearly with the amount of external cost that is internalized, by contrast to the loss for farmers which increases quadratically but is very small for a damage cost of 0.3 €/kg_N. Expressed as a change in the fertilizer-dependent part of the farmers' revenue (crop yield × crop price – fertilizer used × fertilizer price), the decrease is less than 0.5% for most crops; the losses are larger only for crops with low €/ha revenue. Averaged over wheat, barley, potatoes, sugar beet and rapeseed, the loss to farmers is about 0.1% in the UK and 0.4% in Sweden. The revenue loss for fertilizer producers is larger, about 8% in the UK and 14% in Sweden.*

Introduction

This paper estimates the environmental impacts and damage costs (also called external costs) of synthetic nitrogen fertilizer and discusses options for reducing these impacts, including their consequences for farmers and for producers of fertilizer.

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Considering the life cycle of synthetic nitrogen fertilizer, the following potentials for damage can be identified:

- global warming due to the production of fertilizer;
- damages due to air pollutants emitted during the production of fertilizer;
- global warming due to the application of fertilizer;
- eutrophication due to leaching of applied fertilizer;
- pollution of drinking water due to leaching of applied fertilizer; and
- damages due to release of volatile substances (especially NH_3) from applied fertilizer.

Acidification of soils should not arise if good farming practices are followed.

Among the policy options for abating the damages, taxes and tradable permits are examined in detail. Using data for crop yield as function of fertilizer input, the consequences of reduced input are evaluated. However, in view of the site-specific nature of some of the damage mechanisms, the simple generic analysis of the present paper can provide only a rough first assessment of economic consequences and must not be used for the formulation of specific policies.

Damage Costs Due to Nitrogen Fertilizer

Impacts due to Fertilizer Production

The upstream impacts are mainly due to the emission of greenhouse gases and other air pollutants during production of fertilizer and its raw materials, ammonia and nitric acid. Apart from greenhouse gases only NO_x and NH_4NO_3 emissions make a significant contribution. The emission rates in Table 1 are typical of current technologies (Gruncharov & Bozadjiev, 2000). For the damage cost per kg_{Nox} 3.4 €/kg_{NO_x} is taken as calculated for LCA applications in Europe by the ExternE (2004) project series. The toxicity of ammonium nitrate particles is assumed by ExternE (2004) to be half that of ordinary combustion PM_{10}, and its damage cost is estimated to be 6.0 €/$kg_{NH_4NO_3}$.

For greenhouse gas emissions the value of 7.0 kg CO_{2eq}/kg_N is taken from Wood & Cowie (2004, their Table 5) as the average value for European nitrogen fertilizer production, based on a review of eight life cycle assessments. It is made up mainly of CO_2 emissions from ammonia production, and N_2O emissions from nitric acid production, but also includes other emissions of these gases and of CH_4 in the production of fertilizers. The greenhouse gases are expressed in terms of CO_2

Table 1. Emissions and damage costs due to fertilizer production

Pollutant	kg_{poll}/kg_N	€/kg_{poll}	€/kg_N
NO_x	2.4 E−3	3.4 €/kg_{NO_x}	0.01
NH_4NO_3	3.7 E−3	6.0 €/$kg_{NH_4NO_3}$	0.02
CO_{2eq}	7.0	0.019 €/kg_{CO_2}	0.13

equivalent, using their global warming potential (GWP). The final stage of fertilizer manufacture requires electricity input, about 0.2 kWh/kg_N and mostly from base load plants burning coal or lignite, with the emission of PM_{10}, NO_x and SO_2. The damage cost of these pollutants is typically in the range of 0.01 to 0.04 €/kWh (ExternE, 2004), implying a contribution of about 0.002 to 0.008 €/kg_N. Since this is rather small compared to the uncertainties of the estimates in Table 1, it is not considered. For the damage cost of greenhouse gases we take the value of 19 € per tonne of CO_2 of ExternE (2004).

Global Warming Due to N Fertilizer Use

Soil bacteria decompose nitrates and emit N_2O, a powerful greenhouse gas. Its global warming potential (GWP) is about 310, meaning that 1 kg of N_2O causes as much warming as 310 kg of CO_2. Thus, despite relatively small quantities that are emitted, it contributes currently about 5% of the total global warming. The emission of N_2O has increased dramatically above the natural background, ever since the massive worldwide use of synthetic nitrogen fertilizer began during the 1960s.

The emission of N_2O from cultivated land is highly variable in space and time (see Appendix A). Variations due to soil management, cropping systems and rainfall are larger than due to the type of mineral fertilizer. According to the Intergovernmental Panel on Climate Change (IPCC) (1995) and Mosier *et al.* (1998) a typical average value is:

$$N_2O \text{ flux (in } kg_N/\text{ha}) = 1.0 + 0.0125 \times \text{N input (in } kg_N/\text{ha}) \quad (1)$$
$$\text{(uncertainty: range 0.025 to 0.225)}$$

Typical input rates are in the range of 100 to 200 kg_N/ha, implying that the anthropogenic contribution is larger than the background flux (the 1.0 in Eq. 1).

All flows of nitrogen-containing compounds are reported in mass units of N (molar weight 14). To convert the N_2O flux from kg_N to kg N_2O, recognizing that one mole of N makes half a mole of N_2O, and multiplying by the molar masses one finds that the incremental flux due to anthropogenic input is:

$$(2 \times 14 + 16)/2\ g_{N_2O}/\text{mol}/14\ g_N/\text{mol} = 22/14\ g_{N_2O}/g_N = 1.57\ g_{N_2O}/g_N$$

Multiplying by the GWP of N_2O one obtains the equivalent CO_2 emission per kg of fertilizer:

$$310\ g_{CO_{2eq}}/g_{N_2O} \times 1.57\ g_{N_2O}/g_N \times 0.0125 = 6.09\ kg_{CO_{2eq}}/kg_N \text{ of fertilizer}$$

At a damage cost of 0.019 €/$kg_{CO_{2eq}}$ the corresponding damage is:

$$6.09\ kg_{CO_{2eq}}/kg_N \times 0.019\ €/kg_{CO_{2eq}} = 0.116\ €/kg_N \text{ of fertilizer.}$$

Health Impacts

The main health impact of concern is methemoglobinemia, a serious and often fatal illness in infants due to conversion of nitrate to nitrite by the body, which can reduce the oxygen-carrying capacity of blood. Adults are not affected. To eliminate this risk, the current legislation for water quality in the EU and in the USA imposes a limit of 50 $mg_{nitrate}/L$ in drinking water (EC, 1998). With this limit the concentration is below the no-adverse-observed-effect-level (NAOEL) for methemoglobinemia reported in the epidemiological studies, hence no health risk is expected from treated water (see http://www.epa.gov/iriswebp/iris/index.html).

The epidemiological studies that established the limit value are somewhat dated and were based in rural USA around 1950. More recent work has established that the risk of methemoglobinemia is greatly enhanced by concurrent microbial infections and the observed cases were probably associated with poor hygiene and inadequate control of microbes in drinking water. The risk of methemoglobinemia is hence likely to be close to zero in countries with good water quality and concentrations below 50 mg/l. However, the risk could be significant in developing countries and in some regions of Eastern Europe. For developing countries a recent study in India highlights the risk (Gupta *et al.*, 1999).

Eutrophication

Eutrophication arises from the gradual increase in the concentration of phosphorus, nitrogen and other plant nutrients in an aging aquatic ecosystem, leading to excessive growth of certain species, especially blooms of algae (e.g. phytoplankton), and creating conditions that interfere with the recreational use of lakes and estuaries, and the health and diversity of indigenous fish, plant and animal populations. Algal blooms hurt the system by clouding the water and blocking sunlight, and by increasing oxygen demand when the algae die and decompose. Although aquatic eutrophication is a natural process in the aging of lakes and some estuaries, human activities can greatly accelerate the problem by increasing the rate at which nutrients and organic substances enter aquatic ecosystems from their surrounding watersheds, due to agricultural runoff, urban runoff, leaking septic systems, sewage discharges, eroded stream banks and similar sources. Atmospheric emissions of NO_x can also contribute to eutrophication. (For a good overview of eutrophication see http://www.epa.gov/maia/html/eutroph.html).

Run-off of nutrients from agriculture is the dominant contribution to eutrophication in most areas, and it has been recognized as an important environmental problem in Europe. However, so far no estimates are available for the damage costs per emitted pollutant. Thus, even though several studies provide monetary information for eutrophication, the link to the emitted pollutant is missing. For example, Pretty *et al.* (2003) estimate that the total cost of freshwater eutrophication in the UK is between \$105 and \$160 million per year. However, this is a total without any link to the individual causes, such as nitrate deposition following atmospheric emissions of NO_x, discharges from water works, inadequate management of animal manure, nitrate effluents from crop land, and phosphate effluent from crop land. In addition, the specific effect of nitrogen on eutrophication depends on the local nutrient balance and is highly site-dependent. If it is assumed that none of these significant causes individually accounts for more than 50% or less than 10% of the total damage estimate, then the eutrophication cost of the 1228 kt_N of N-fertilizer used in 2001 in the UK (Eurostat data) lies in the range of 0.01 – 0.065 €/kg_N, with a central value of 0.03 €/kg_N.

Summary of the Damage Costs

The damage costs ('external costs') that have been quantified are summarized in Table 2, expressed like all results in this paper per kg of N in the fertilizer. NH_3 releases from fields are not considered because for this pollutant ExternE has not yet provided any impact data.

Even without NH_3 emissions the total damage cost is not at all negligible compared to the market price of fertilizer, currently about 0.5 €/kg_N. If it were internalized by a pollution tax, it would raise the price by about 60%. Because of the uncertainties of the damage cost estimates (Rabl & Spadaro, 1999) the total is rounded down to 0.3 €/kg_N, for the assessment of economic consequences in the fourth section.

Policy Options for Reducing the Use of N Fertilizer

Policy Options to Achieve Cost Internalization

Agricultural and environmental policies have been developed in the European Union and in many member countries to limit damages arising from incorrect application of plant nutrients. Both organic nutrients and inorganic fertilizer are targeted by such policies. An interesting case is the German requirement that farmers annually calculate nutrient balances and demonstrate that they do not over- or under-apply manure and fertilizers. Science-based approaches to support good agricultural practice and optimal nitrogen application are promoted by the fertilizer industry.

Interestingly, many of the approaches stress that nitrogen use reductions can be achieved without reductions in agricultural yield (briefly reviewed in the next sub-section). Later the paper considers the relation between use of synthetic fertilizer and crop yield, to determine the socially optimal level of application, all else being equal.

Table 2. Damage costs of nitrogen fertilizer, in €/kg_N

Impact category	**€/kg_N**
Greenhouse gases from fertilizer production	0.13
NO_x from fertilizer production	0.01
NH_4NO_3 from fertilizer production	0.02
N_2O from fertilizer in soil[a]	0.12
NH_3 emissions from fertilizer in fields	Not quantified
Eutrophication[b]	~0.03
Health (infant mortality due to nitrates in drinking water)	negligible if $< 50\ mg_{NO_3}/L$
Total	~0.31

[a]Site-specific, numbers shown are global average.
[b]Very site-specific, number shown is rough estimate of average freshwater eutrophication in UK.

To reach the social optimum, a variety of policy options could be considered, for example:

- a tax per kg_N equal to the external cost;
- tradable permits that are issued free by the government;
- tradable permits that are auctioned by the government;
- restrictions or limits on the use of fertilizer (maximal kg_N/ha);
- incentives to improve fertilization practices or substitute manure.

Taxes (equal to the marginal damage cost) and permits are market-based and achieve the desired reduction of use with greatest economic efficiency because they allow greater flexibility than rigid kg_N/ha limits (the latter force everyone to obey the same limit even though the cost per avoided kg_N is very different between different farmers). Taxes and permits differ in their approach to the social optimum. With taxes the regulator only needs to know the damage cost, but the response of the polluters may be slow and unpredictable (the regulator does not know their abatement costs). Permits achieve a specified pollution reduction immediately, although the optimal reduction level is not known in advance since it involves the abatement costs. But since the resulting permit price is equal to the marginal abatement cost, the regulator can adjust the quantity of permits periodically until the social optimum is reached.

It is important to distinguish, on the one hand, tradable permits that are issued free (henceforth referred to as 'free permits'), and, on the other, permits that are auctioned by the government or taxes. Even though all three are market-based and achieve the desired reduction of use with greatest economic efficiency (greater flexibility than rigid kg_N/ha limits), there is an enormous difference in cost to the polluters. As shown by Desaigues & Rabl (2001), with free permits they pay only the abatement cost (i.e. the cost necessary to reach the social optimum) whereas with auctioned permits or with taxes they have to pay the residual damage cost in addition (i.e. the damage cost of the socially optimal emissions). Since the residual damage cost is usually much larger than the abatement cost, the difference for the polluter is very important. It turns out to be dramatic in the case of fertilizer use by farmers, as shown later.

Transaction costs may reduce the extent of this difference. The cost of verification and monitoring of permit schemes may be significant. The impact of transaction costs on permit schemes has been examined by Cason & Gangadharan (2003) and Netusil & Braden (2001). Cason & Gangadharan (2003) show the influence that initial allocations can have on cost efficiency in the presence of transaction costs, finding that in the case of a market with constant marginal transaction costs the importance of allocating the permits accurately is important. In the case where marginal transaction costs decline the impacts of transaction costs are less important in the long run, as search costs and information costs decline with experience. Although small, region specific markets may affect the extent to which transaction costs would decline. Netusil & Braden (2001) simulate the impacts of transaction costs and trading regimes on a hypothetical tradable permit scheme to address sediments from agriculture, finding that combining transaction costs with bilateral and sequential trading results in less cost saving than under the optimal solution.

However, even with the highest level of transaction costs modeled the abatement costs are lower than under the regulations in force. For a review of the efficiency of different instruments in controlling pollution see Markandya *et al.* (2002).

The policy options for reducing use of nitrogen fertilizer would have economic consequences that should be analyzed carefully before making a choice. The economic consequences are:

(1) direct:
- for farmers: reduced income,
- for producers of fertilizer: reduced sales;

(2) indirect:
- land use and employment effects,
- changed consumer spending due to changed prices,
- effect on international trade.

The impacts of raising N fertilizer prices on land use require the use of complex models linking different uses of land. Such a model is currently being developed under the SENSOR project for the European Commission, building on a macroeconometric model, a land-use allocation model and the CAPRI model for the agricultural sector (SENSOR, 2005). Such modeling is beyond the scope of the current paper, which considers agriculture in a partial equilibrium environment, assuming no changes to land use (e.g. move to agri-tourism or to forestry) or to the land areas used to grow certain crops. This is necessarily a simplified exercise, but the changes shown are largely marginal in terms of applied nitrogen and hence there is not likely to be significant impacts, unless the change in application of N fertilizer or, in the case of a tax, the additional cost of fertilizer makes extensive areas of marginal farm land not viable. The latter is certainly possible in some areas of Europe at the present time, where agriculture is facing a range of stresses including increases of costs and declining prices for products. The sustainability of rural communities may also be brought into question if increasing fertilizer prices lead to reduction in employment in rural areas.

For the purposes of this paper, which has as its main aim the identification of the external costs of N fertilizers and a first assessment of the impacts on farm incomes, these are not taken into account. There is need for a full analysis of the indirect impacts of N fertilizer taxes before they are applied, including use of macroeconometric or computable general equilibrium models and spatially explicit models of land use where available to take into account the multifunctional benefits of agricultural land use.

This paper evaluates only the direct effects of reduced crop yield due to restrictions on fertilizer use. Effects on consumer spending and international trade are likely to be negligible because the changes in crop yield turn out to be very small. Examination of the other indirect effects would exceed the scope of the present paper.

Options for N-fertilizer Use Reduction without Yield Loss

Knowledge-based farming approaches might have a significant potential to reduce N_2O emissions. For example, Mosier *et al.* (1996) suggest that using nitrification inhibitors is an option for decreasing N-fertilizer use and to directly mitigate N_2O emissions. The tilling practice is also significant. Jacinthe & Dick (1997, p. 221) report that for maize/soybean/wheat farming in Ohio, the "seasonal N_2O-N loss from chisel-till plots was generally significantly higher than from no-till or ridge-till plots". In a paper published in *Science*, Matson *et al.* (1998) suggest that Mexican wheat farmers could save between 12 and 17% of after-tax revenues by lowering usage of nitrogen fertilizer, and applying it later in the crop cycle, thus reducing the loss of nitrogen without affecting yield and grain quality. They state: "A knowledge-intensive approach to fertilizer management can substitute for higher levels of inputs, saving farmers money and reducing environmental costs (p. 225)."

Table 3 shows estimates by the IPCC (1995) of the worldwide potential of reducing N_2O emissions by improved agricultural practices, without any reduction of yield. The total potential is 0.68 Mt_N/yr compared to total worldwide N_2O emissions of 3.5 Mt_N/yr. A further method for reducing the need for synthetic fertilizers is to increase the use of sewage sludge (although that may be controversial because it often contains significant traces of toxic metals).

Table 4 gives an indication for the potential of increased use of manure instead of synthetic fertilizer. Such a shift would reduce the emissions of N_2O because most manure would produce N_2O in any case even if not used as fertilizer (see Table 5). However, in Europe most of the produced manure is already used as fertilizer and the potential for greater use is quite limited.

Table 3. Estimates by IPCC (1995) of the worldwide potential of reducing N_2O emissions by improved agricultural practices, without any reduction of yield, in Mt_N/yr

Match N supply with crop demand (Soil/plant testing to determine N needs; minimize fallow periods; optimize split application schemes; reduced crop production in regions with overproduction)	0.24
Tighten N flow cycles (Integrate animal and crop production for manure reuse; maintain plant residue on production site)	0.14
Advanced fertilization techniques (Controlled-release fertilizers; place fertilizers below ground surface; foliar application of fertilizers; use nitrification inhibitors; match fertilizer type to seasonal precipitation)	0.15
Optimize tillage, irrigation and drainage	0.15
Total	0.68

Table 4. Fertilizer use (IPCC, 1995)

Mt_N/yr	Europe	World
Synthetic N	13.6	77.4
Manure N produced	12.3	115.3
Manure N used as fertilizer	11.1	74.9

Table 5. Emissions of N_2O due to different fertilizer types, in Mt_N/yr (IPCC, 1995)

Mt_N/yr	Europe	World
Synthetic N	0.27	1.53
Animal waste	0.22	1.49
N fixation	0.02	0.50
Total	0.51	3.50

Table 5 shows that N fixation by legumes also contributes N_2O, but far less per incremental crop yield because the plants use this source very efficiently; furthermore there is no runoff to pollute drinking water or cause eutrophication.

Up to this point, the paper has reviewed approaches to reduce the use of N-fertilizer that do not lead to reductions in crop yield. However, in view of the high damage cost estimated in this case study, we deem it appropriate to go beyond such measures.

Crop Yield and Optimal N-fertilizer Use

The yield per ha is likely to be lower with a restriction on the quantity of synthetic fertilizer. Crop yield depends on many different parameters, and it is difficult and research-intensive to single out the effect of nitrogen dosage. Fertilizer suppliers regularly carry out trials in order to be able to give advice on optimal usage. Unfortunately, the results are not easily available. Much of the present paper is based on older nitrogen response curves of crops in the UK (England, 1986). Some yield data were also obtained for several crops in Sweden (O'Shea, 2004), and the general trends found could be substantiated by comparison with a recent yield curve for wheat in Germany, communicated by the European Fertilizer Manufacturers Association (EFMA, 2004) and with curves for corn in Illinois, USA (see http://www.cropsci.uiuc.edu/research/pubs/n-rate-2001.html#figure1).

A general feature of all these response curves is that the initial response to nitrogen application is strong, but then diminishes with yield being very insensitive to N-dose changes around the maximum. This is illustrated in Figure 1 with some typical curves. The equations of these curves are:

$$y(q) = a_1 - a_2 a_3^q - a_4 q \quad \text{for the UK (England, 1986)} \tag{2}$$

and

$$y(q) = a + bq - cq^2 \quad \text{for Sweden (O'Shea, 2004) and for Germany (EFMA, 2004).} \tag{3}$$

There are large variations, reflecting in part differences in climate and soil chemistry. By contrast to the yield curve for wheat in Germany (EFMA, 2004), the curves for both the UK and Sweden are based on older measurements of N fertilizer-yield relationships, corresponding to the technologies and practices at the time. However, that does not affect the conclusions here because the key feature is the same for all:

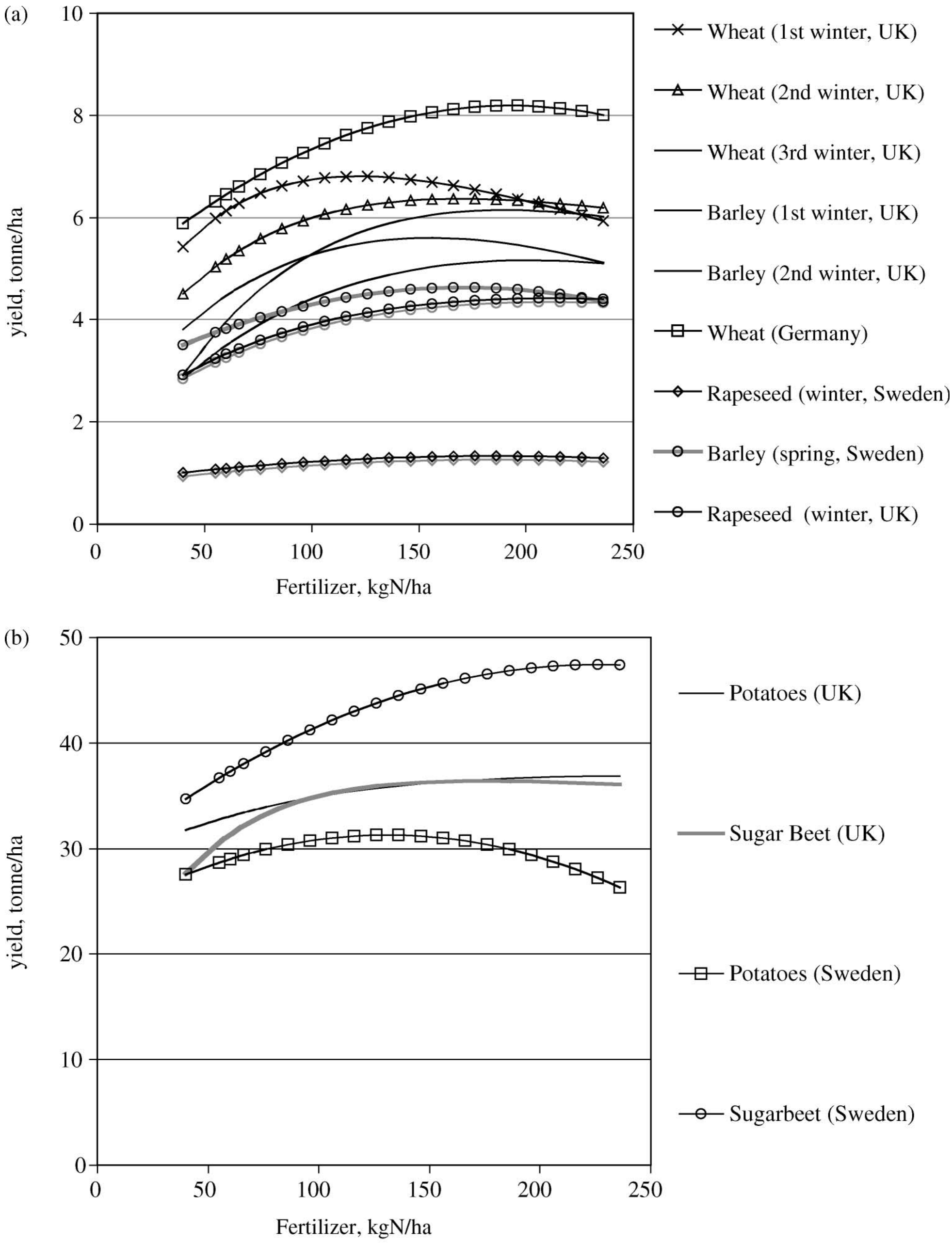

Figure 1. Typical examples of crop yield versus N fertilizer use: (a) yield curves for barley, wheat and rapeseed; (b) yield curves for potatoes and sugar beet.

there is a broad optimum, generally around 100 to 200 kg_N/ha. The changes in fertilizer input due to internalization are not large, and for changes near the optimum the effect on the yield and on the revenue of farmers is small, as shown in the following. In any case, the point of this paper is to indicate the general magnitude of the effects, not to provide specific numbers for detailed formulation of policies.

Given the yield y(q) as function of fertilizer input q, the optimal input is readily found by considering the tradeoff between the cost of incremental input and the value of the incremental yield. Denoting the fertilizer-dependent contribution to the revenue of a farmer by R, in €/ha:

$$R = yp_C - qp_N \tag{4}$$

where

$$p_C = \text{price of crop, in €/kg;}$$

$$p_N = \text{price of fertilizer, in €/kg}_N.$$

The net revenue is lower than R because there are other production costs, such as labor and machinery.

The private optimum level of application of nitrogen fertilizer is estimated using the yield curves in Figure 1. The optimum dosage of nitrogen fertilizer occurs when the revenue obtained from the marginal unit of N applied equals the marginal cost of the N fertilizer. An estimate is made of the impact of increasing the price of nitrogen fertilizer by the external costs to obtain a socially optimal level of N fertilizer application. The socially optimal level of N fertilizer is less than the private optimum as long as the external costs are positive. See Appendix B for the mathematical detail of the estimation of the socially and privately optimal rates of N fertilizer application.

Comparison of Modeled and Observed Fertilizer Use

It is interesting to compare the fertilizer input rates that have been calculated with the rates observed in current practice, using data from the UK (DEFRA, 2003). The results are shown in Table 6. A five-year average was used to even out the factors that would change nitrogen application year-to-year such as the weather and financial circumstances. As the Table shows, the model comes very close in predicting the application to potatoes and sugar beet, but for the other crops the application of N is systematically underestimated compared to the current situation.

That is not surprising because current practice is to use large amounts of fertilizer, much larger than the optimum of the yield curves. One of the reasons lies in the fact that the cost of the fertilizer is small compared to the value of the crops, so the farmer is not very sensitive to the cost of fertilizer.

It is probable that farmers use more than the theoretically optimal amount of nitrogen as a way of maximizing yields under uncertainty since they have difficulty determining the optimum with precision, especially in view of uncertainties about weather and soil conditions. However, that does not change the key feature of the result here, as already pointed out in the previous subsection with the comment about the age of the yield curves. In fact, each farmer has his own 'expected yield curve' y*(q), i.e. what he expects the yield to be taking into account the uncertainties and the local conditions. Thus he optimizes his revenue according

Table 6. Difference between modeled and observed fertilizer use in the UK, at fertilizer price of 0.5 €/kg_N. The observed data are from DEFRA (2003) for the period 1998 – 2002; they do not distinguish between different years of the planting cycle

	Observed kg_N/ha				
	low	***high***	**Average**	**Modeled kg_N/ha**	**Difference**
Wheat	*183*	*197*	190.2		
Wheat (1st winter)			190.2	82.8	130%
Wheat (2nd winter)			190.2	98.6	93%
Wheat (3rd winter)			190.2	127.7	49%
Barley (winter)	*136*	*156*	146.6		
Barley (1st winter)			146.6	92.6	58%
Barley (2nd winter)			146.6	107	37%
Barley (summer)	*95*	*114*	107		
Barley (1st summer)			107	66.8	60%
Barley (2nd summer)			107	91.7	17%
Rapeseed (winter)	*188*	*202*	196.4	121.7	61%
Potatoes	*193*	*172*	178.4	179.6	−1%
Sugar beet (UK)	*104*	*112*	108.2	111.6	−3%

to y* instead of y. The formulas and results are the same, with merely the replacement $y''(q_{op})$ -> $y^{*''}(q_{op})$ in the key result of Eq. B.14. So there is an unknown but small change in the curves. Since the conclusions do not depend on the exact curves, they remain valid.

Another reason for the overestimation in Table 6 may be the age of the yield curve data and the site specificity of yield curve estimation. Over time land loses its natural fertility so greater levels of fertilizer per hectare are needed to keep yields high. In addition, new cultivars of crops have been developed which have a better response to nitrogen fertilizer.

Economic Consequences of Internalization

Losses for Farmers and for Fertilizer Producers

This section calculates the change in the farmers' revenue contribution R, defined in Eq. 4, i.e. the proceeds from selling a crop minus the cost of the fertilizer. For this calculation it is assumed that farmers are currently operating at the optimal input level q_{op} of Eq. B.3 and would be forced to change to the social optimum q_{os} of Eq. B.6. Of course, in reality farmers may take additional criteria into account, resulting in different input levels (see the previous subsection). In addition, the real yield can be quite different from the yield curves assumed here because it depends on many additional variables, such as local soil conditions and the weather in a particular year.

However, the results for the change due to internalization demonstrate that differences between different yield curves have a relatively small effect on the change in the revenue contribution R; in particular the absolute level of the yield curve does not matter because the optimum depends only on the derivative. In addition, the study here is considering the effect of reducing fertilizer input, all else being the same. The results are intended only as a general indication of the magnitude of the economic consequences of internalization, not as specific guidelines. In any case, policies should not be based on simple tax or permits per kg_N, but on the excess fertilizer left in the soil after harvest that can be measured (EFMA, personal communication 2004).

For crop prices the FAOSTAT database was consulted and the producer data of 2001 was used, the most recent available; they were converted to € at the 2001 rate from local currencies. The price of fertilizer was set at 0.50 €/kg_N based on the 2000 price given in Richards (2000), inflated to the 2001 price and converted from Sterling to € at the 2001 exchange rate.

The cost to farmers is estimated according to the value of lost productivity, assuming no changes in prices, and the tax burden in the case of taxes (see Eqs. B.9–B.14 of Appendix B). Results are shown in Figure 2, as percentage reduction of the quantity R versus the amount e of the external cost that is internalized. The crops shown as examples include those mentioned above, using the crop N-response functions of Figure 1. At 0.3 €/kg_N the loss is less than 0.5% in most cases. The higher the absolute revenue per ha, the lower the percentage loss. The highest curves in Figure 2 are for marginally productive crops. In this example it is assumed that permits are issued freely and without transaction costs. Any such costs may reduce the difference between free permits and taxation, an example of which is shown in Figure 3.

With free permits the loss to farmers increases with the square of the damage cost e that is internalized, and it is inversely proportional to the price of the crop. e is

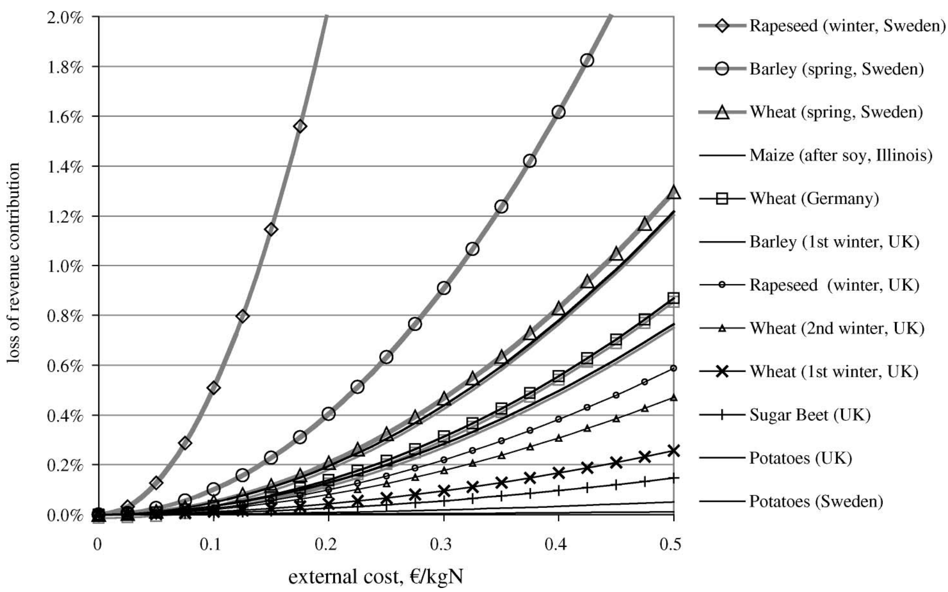

Figure 2. The effect of the internalization on revenue of farmers, as percentage reduction of the quantity crop sales: fertilizer cost versus amount of the damage cost that is internalized by free permits.

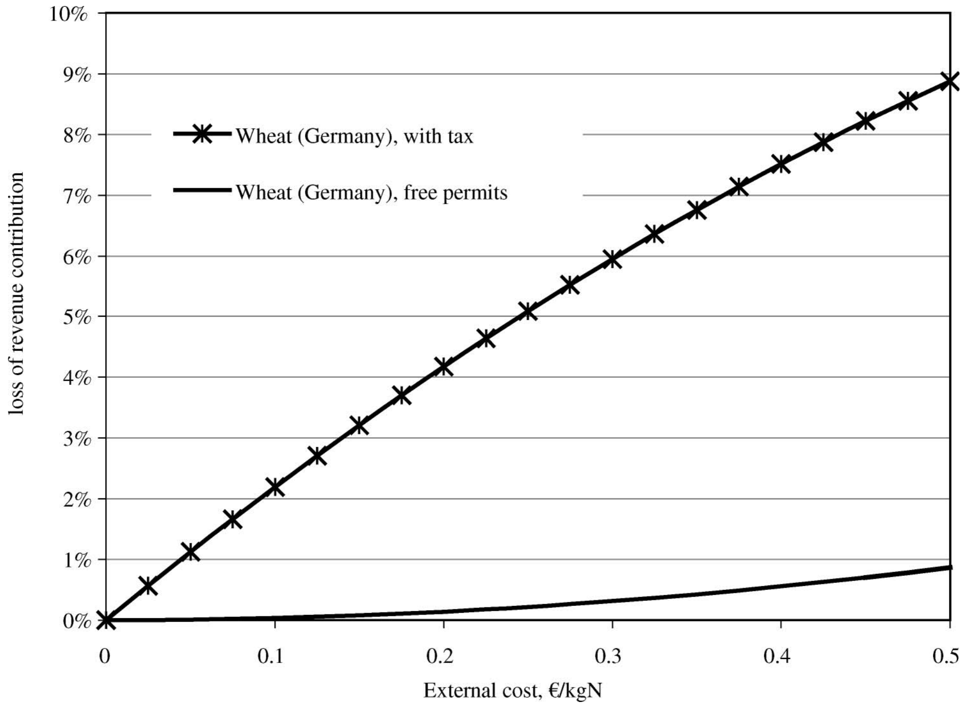

Figure 3. Comparison of the fall in revenues under free permits and under a tax.

sufficiently small so that most crops lie in the slowly varying zone of the curves in Figure 2. If on the other hand the external costs are internalized by a tax, the loss to farmers is that with free permits (Eq. B.14 of Appendix B) plus the tax e on the purchased fertilizer.

For the magnitude of e under consideration here the linear term is much larger than the quadratic one, hence for the farmers the impact of a tax is much more severe than free permits. That is shown in Figure 3. With $e = 0.3$ €/kg_N the loss of revenue contribution would be about an order of magnitude larger with a tax than with free permits. However, the revenue from the tax could be spent in many different ways, for example to reduce general income taxes or to reimburse farmers for their pollution control costs; the effects of the tax would certainly be very different.

It is also of interest by how much fertilizer use would be reduced if costs were internalized. This is illustrated in Figure 4, which shows that the quantity of fertilizer used is reduced linearly with the amount of the external cost that is internalized. The revenue of the fertilizer producers is generally proportional to the quantity sold, and at $e = 0.3$ €/kg_N their loss would be in the range of about 4 to 10% for most crops.

Country Averages

The investigation was extended to the level of entire countries, by combining the crop-specific results with figures from FAOSTAT showing the area A_i of the UK and Sweden used for each crop to get aggregate national costs. The results are shown in

Table 7 for the internalization of a damage cost of e = 0.30 €/kg_N by free permits. In the case of the crops for which several yield functions are given in Table 6, depending on season (e.g. wheat) we have taken the simple average of the respective numbers. The loss of the revenue contribution R amounts to 0.13% for the UK and 0.35% for

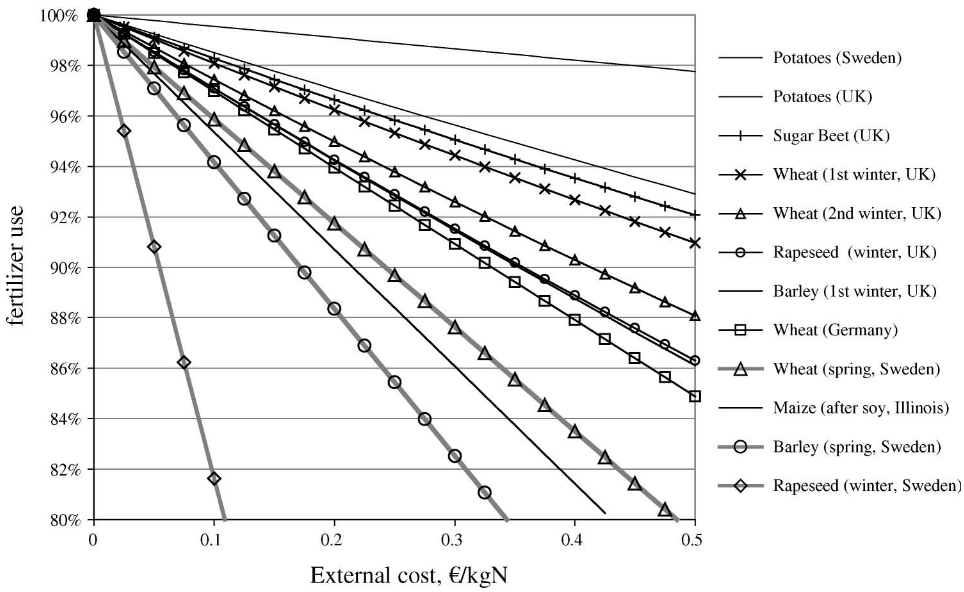

Figure 4. Effect of the internalization on quantity of fertilizer versus damage cost that is internalized.

Table 7. Country level results for UK and Sweden, for the internalization of a damage cost of e = 0.30 €/kg_N by free permits

	Area	**Base R**	**ΔR**	**ΔR/Base R**	**Base q**	**Δq**	**Δq/Base q**
UK	1000 ha	M€	M€	%	1000 t	1000 t	%
Wheat	545	417	0.83	0.20%	75.3	5.7	7.5%
Barley	311	153	0.55	0.36%	41.3	3.7	9.0%
Rapeseed	404	353	0.99	0.28%	72.0	6.7	9.4%
Potatoes	165	1057	0.28	0.03%	36.4	1.9	5.3%
Sugar beet	177	286	0.29	0.10%	25.9	2.0	7.7%
Total	*1602*	*2266*	*2.9*	*0.13%*	*250.9*	*20.1*	*8.0%*
Sweden							
Wheat	200	117	0.47	0.40%	27.5	3.1	11.4%
Barley	398	152	1.38	0.91%	52.6	9.2	17.5%
Rapeseed	22	6	0.13	2.14%	2.7	0.8	30.9%
Potatoes	32	194	0.01	0.00%	4.1	0.1	1.3%
Sugar beet	55	117	0.07	0.06%	11.6	0.5	4.1%
Total	*707*	*585*	*2.1*	*0.35%*	*98.6*	*13.7*	*13.9%*

Source: FAOSTAT (2004) and authors' estimates.

Sweden. The reduction of the quantity of fertilizer is 8.0% for the UK and 13.9% for Sweden.

The impact on income and crop yield clearly depends on the elasticity of demand for fertilizer. Previous studies reviewed in Pearce & Koundouri (2003) show elasticities for nitrogen fertilizer from -0.12 to -0.51 for Sweden and from -0.1 to -1.1 for all countries. With the price of fertilizer ranging from 0.50 to 1 €/kg_N, price elasticities are estimated from -0.23 to -0.60 for Sweden and from -0.12 to -0.24 for the UK. This shows that the sensitivity of the use of nitrogen fertilizers to the price is greater in Sweden than in the UK, reflecting the different crop mixes and relative returns to adding nitrogen fertilizers in the two countries.

Conclusion

The paper has shown that the damage costs of nitrogen fertilizer use that can be readily estimated are large, about 60% of the market price of the fertilizer on average, although highly variable with local conditions. After discussing general policy options to reduce the damage, the paper has evaluated the most important direct economic consequences of a fertilizer tax or a limitation of fertilizer use by means of tradable permits.

If the damage costs were internalized by a pollution tax or tradable permits that are auctioned by the government, the economic consequences would be heavy, with a large revenue loss for farmers. But if it is internalized by tradable permits that are given out free, the revenue loss for farmers is small. The loss for fertilizer producers increases linearly with the amount of external cost that is internalized, by contrast to the loss for farmers which increases quadratically but is very small for a damage cost of 0.3 €/kg_N. Expressed as a change in the fertilizer-dependent part of the farmers' revenue (crop yield × crop price – fertilizer used × fertilizer price), the decrease is less than 0.5% for most crops; the losses are larger only for crops with low €/ha revenue. Averaged over wheat, barley, potatoes, sugar beet and rapeseed, the loss to farmers is about 0.1% in the UK and 0.4% in Sweden. The revenue loss for fertilizer producers is larger, about 8% in the UK and 14% in Sweden.

This paper has presented first estimates of the external costs of nitrogen fertilizer use and the consequences of the internalization of such costs. It should be noted that for a complete analysis of the impacts, other factors, such as changes in land use and employment, should be taken into account. This study clearly demonstrates that the damage costs are significant and require attention.

When designing an appropriate policy, great care is needed to make it effective and to avoid unexpected perverse effects. A tax equal to the external cost would be so large as to create serious problems for farmers and should be avoided in favour of tradable permits that are issued free. It would be especially perverse to make farmers pay a tax equal to the total external cost, since a large portion is caused by the producers of the fertilizer. Among the points to take into account is the great variability of local conditions, both environmental and economic. Since eutrophication and the emission of N_2O from fields are highly variable with specific local conditions, a simple limitation of kg_N/ha would not make any sense. Rather, any policy should be based on the nutrient load left in the soil after the harvest, something already routinely measured in Germany.

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References

Ambus, P. & Christensen, S. (1995) Spatial and seasonal nitrous-oxide and methane fluxes in Danish forest-ecosystems, grassland-ecosystems, and agroecosystems, *Journal of Environmental Quality*, 24(5), pp. 993–1001.

Cason, T. N. & Gangadharan, L. (2003) Transactions costs in tradable permits markets: an experimental study of pollution market designs, *Journal of Regulatory Economics*, 23(2), pp. 145–165.

Clayton, H., McTaggart, I. P., Parker, J., Swan, L. & Smith, K. A. (1997) Nitrous-oxide emissions from fertilized grassland: a 2-year study of the effects of N fertilizer form and environmental-conditions, *Biology and Fertility of Soils*, 25(3), pp. 252–260.

Davidson, E. A., Matson, P. A. & Brooks, P. D. (1996) Nitrous-oxide emission controls and inorganic nitrogen dynamics in fertilized tropical agricultural soils, *Soil Science Society of America Journal*, 60(4), pp. 1145–1152.

DEFRA (2003) *British Survey of Fertilizer Practice: Fertilizer Use on Farm Crops for Crop Year 2002* (London: Department for Environment, Food and Rural Affairs).

Desaigues, B. & Rabl, A. (2001) Pollution tax and other policy instruments: who pays what? *Pollution Atmosphérique*, Special Issue, December, pp. 27–40.

EC (1998) Council Directive 98/83/EC of 3 November 1998 on the quality of water intended for human consumption. Available at http://europa.eu.int/eur-lex/en/index.html

EFMA (2004) European Fertilizer Manufacturers Association, Brussels. Personal communication with C. Pallière.

England, R. A. (1986) Reducing the nitrogen input on arable farms, *Journal of Agricultural Economics*, 37(1), pp. 13–24.

ExternE (2004) Project NewExt, New Elements for the Assessment of External Costs from Energy Technologies. European Commission DG Research, Contract No. ENG1-CT2000-00129. Coordinated by R. Friedrich, IER, University of Stuttgart. Final report. Available at http://www.externe.info

FAOSTAT (2004) FAO Statistical Databases, Food and Agriculture Organization of the United Nations. Available at http://apps.fao.org/default.jsp (accessed 9 December 2004).

Gruncharov, I. & Bozadjiev, P. (2000) Final report on the environmental impact assessment of NEOCHIM-Plc, Dimitrovgrad. See also http://www.neochim.bg/

Gupta, S. K., Gupta, R. C., Seth, A. K., Gupta, A. B., Bassin, J. K. & Gupta, A. (1999) Adaptation of cytochrome-b5 reductase activity and methaemoglobinaemia in areas with a high nitrate concentration in drinking-water, *Bulletin of the World Health Organization*, 77(9), pp. 749–753.

Harrison, R. M., Yamulki, S., Goulding, K. W. T. & Webster, C. P. (1995) Effect of fertilizer application on NO and N_2O fluxes from agricultural fields, *Journal of Geophysical Research—Atmospheres*, 100(D12), pp. 25923–25931.

Henault, C., Devis, X., Lucas, J. L. & Germon, J. C. (1998) Influence of different agricultural practices (type of crop, form of N-fertilizer) on soil nitrous-oxide emissions, *Biology and Fertility of Soils*, 27(3), pp. 299–306.

IPCC (1995) Economic and social dimensions of climate change, in: J. P. Bruce, H. Lee & E. F. Haites (Eds) *Contribution of Working Group III to the Second Assessment Report of the Intergovernmental Panel on Climate Change* (Cambridge: Cambridge University Press).

Jacinthe, P. A. & Dick, W. A. (1997) Soil-management and nitrous-oxide emissions from cultivated fields in Southern Ohio, *Soil & Tillage Research*, 41(3–4), pp. 221–235.

Markandya, A., Harou, P., Bellu, L. G. & Cistulli, V. (2002) *Environmental Economics for Sustainable Growth: A Handbook for Practitioners* (Cheltenham: Edward Elgar).

Matson, P. A., Naylor, R. & Ortizmonasterio, I. (1998) Integration of environmental, agronomic, and economic-aspects of fertilizer management, *Science*, 280(5360), pp. 112–115.
Mosier, A. R., Duxbury, J. M., Freney, J. R., Heinemeyer, O. & Minami, K. (1996) Nitrous-oxide emissions from agricultural fields—assessment, measurement and mitigation, *Plant and Soil*, 181(1), pp. 95–108.
Mosier, A., Kroeze, C., Nevison, C., Oenema, O., Seitzinger, S. & van Cleemput, O. (1998) Closing the global N_2O budget: nitrous-oxide emissions through the agricultural nitrogen cycle, *Nutrient Cycling in Agroecosystems*, 52, pp. 225–248.
Netusil, N. R. & Braden, J. B. (2001) Transactions costs and sequential bargaining in transferable discharge permit markets, *Journal of Environmental Management*, 61, pp. 253–262.
Nevison, C. D., Esser, G. & Holland, E. A. (1996) A global model of changing N_2O emissions from natural and perturbed soils, *Climatic Change*, 32(3), pp. 327–378.
O'Shea, L. (2004) University of Bath, personal communication.
Pearce, D. & Koundouri, P. (2003) Diffuse pollution and the role of agriculture, in: D. Helm (Ed.) *Water, Sustainability and Regulation* (Oxford: OXERA).
Pretty, J. N., Mason, C. F., Nedwell, D. B., Hine, R., Leaf, S. & Dils, R. (2003) Environmental costs of freshwater eutrophication in England and Wales, *Environmental Science & Technology*, 37(2), pp. 201–208.
Rabl, A. & Spadaro, J. V. (1999) Environmental damages and costs: an analysis of uncertainties, *Environment International*, 25(1), pp. 29–46.
Richards, I. R. (2000) *Energy Balances in the Growth of Oilseed Rape for Biodiesel and of Wheat for Bioethanol* (Ipswich: British Association for Bio Fuels and Oils (BABFO)).
SENSOR (2005). Sensor Project Website. Available at http://www.sensor-ip.org/
Skiba, U., McTaggart, I., Smith, K., Hargreaves, K. & Fowler, D. (1996) Estimates of nitrous oxide emissions from soil in the UK, *Energy Conversion and Management*, 37(6–8), pp. 1303–1308.
Smith, K. A., McTaggart, I. P. & Tsuruta, H. (1997) Emissions of N_2O and NO associated with nitrogen-fertilization in intensive agriculture, and the potential for mitigation, *Soil Use and Management*, 13(4), pp. 296–304.
Swart, R. J., Bouwman, L., Olivier, J. & Van den Born, G. J. (1993) Inventory of greenhouse gas emissions in the Netherlands, *Ambio*, 22(8), pp. 518–523.
Thornton, F. C. & Valente, R. J. (1996) Soil emissions of nitric-oxide and nitrous-oxide from no-till corn, *Soil Science Society of America Journal*, 60(4), pp. 1127–1133.
Wood, S. & Cowie, A. (2004) A review of greenhouse gas emission factors for fertilizer production, *IEA Bioenergy Task* 38.

Appendix A. Emission of N_2O from Cultivated Land

The global emission of N_2O from cultivated land was estimated by Smith *et al.* (1997) at 3.5 Tg_N annually, of which 1.5 Tg_N has been directly attributed to synthetic N fertilizers. This is equivalent to 1.95% of the total nitrogen applied, about 77 Tg_N (1990). In addition, anthropogenic N_2O emissions of 0.7, 0.4 and 0.08 Tg_N/yr are attributed to livestock manure, land clearing and atmospheric deposition of fossil fuel NO_x, respectively (Nevison *et al.*, 1996).

Emissions from fertilized grasslands appear significantly larger than those from fertilized crop production in most European countries. Swart *et al.* (1993) estimated 46% of the N_2O emissions in the Netherlands to be from the former and 12% from the latter. Similarly, Skiba *et al.* (1996) confirm that grazed grasslands are the single largest source in the UK. Where grasslands are fertilized and grazed, emissions are even higher: on the other hand, N_2O fluxes from unfertilized pasture appear to be lower than those from fertilized cropland (Davidson *et al.*, 1996).

Major factors that influence N_2O fluxes are soil moisture, temperature (N_2O fluxes are much higher in the tropics), degree of compaction and tilling practices, and type and amount of applied N-fertilizer. Spatial variability is high (Ambus & Christensen, 1995). In a study done in England, it was found that "loss of N_2O to the atmosphere increased sharply at superoptimal levels of fertilizer application" (Harrison *et al.*, 1995, p. 25923). Another study on fertilized UK grassland reported N-fertilizer losses to N_2O at 2.2, 1.4, 1.2, 1.1 and 0.4% from ammonium nitrate supplemented cattle slurry, urea, ammonium nitrate, calcium nitrate and ammonium sulphate, respectively (Clayton *et al.*, 1997).

The effect of the type of N-fertilizer, NH_4NO_3, $(NH_4)_2SO_4$: $CO(NH_2)_2$ and KNO_3, and two types of crop (rapeseed and winter wheat) on N_2O emissions was studied under French conditions by Henault *et al.* (1998) and the proportion of applied N lost as N_2O varied from 0.42% to 0.55% with the form of nitrogen applied. For no-till maize crops in Tennessee, Thornton & Valente (1996) determined emissions of 4.23 kg/ha N_2O-N for the 140 kg_N/ha treatment, and 6.56 kg/ha N_2O-N for the 252 kg_N/ha treatment, i.e. 3% and 2.6% of applied N respectively.

Appendix B. Equations

The optimal dosage of fertilizer q_{op} (the 'private optimum'), occurs at:

$$\left.\frac{dR}{dq}\right|_{q_{op}} = \left.\frac{dy}{dq}\right|_{q_{op}} p_C - p_N = 0 \tag{B.1}$$

from which

$$y'(q_{op}) = \left.\frac{dy}{dq}\right|_{q_{op}} = \frac{p_N}{p_C}. \tag{B.2}$$

Denote the damage cost (external cost) by e, in €/kg_N. The social optimum q_{os} is found by adding e to the price of the fertilizer

$$y'(q_{os}) = \left.\frac{dy}{dq}\right|_{q_{os}} = \frac{p_N + e}{p_C} = y'(q_{op}) + \frac{e}{p_C}. \tag{B.3}$$

It is instructive to consider the effect of small changes by expanding the yield function y(q) around q_{op} (the result is exact for the quadratic curves of Eq. 6)

$$y(q) = y(q_{op}) + (q - q_{op})y'(q_{op}) + \tfrac{1}{2}(q - q_{op})^2 y''(q_{op}) + \ldots \tag{B.4}$$

and

$$y'(q) = y'(q_{op}) + (q - q_{op})y''(q_{op}) + \ldots. \tag{B.5}$$

Using Eq. B.5 to find y′ at the social optimum, and equating to Eq. B.3, one finds

$$y'(q_{os}) = y'(q_{op}) + (q_{os} - q_{op})y''(q_{op}) + \ldots = y'(q_{op}) + \frac{e}{p_C} \tag{B.6}$$

which implies to the lowest order

$$(q_{os} - q_{op})y''(q_{op}) = \frac{e}{p_C} \tag{B.7}$$

The change in the amount of fertilizer is therefore

$$\Delta q = q_{op} - q_{os} = -\frac{1}{y''(q_{op})}\frac{e}{p_C} \tag{B.8}$$

and this is a positive number since the crop yield curve is increasing but downward-sloping in the region of interest, i.e. the 2nd derivative $y''(q_{op})$ is negative. In the vicinity of the optimum the reduction in the amount of fertilizer is directly proportional to the amount of external costs to be internalized. Further, it is inversely proportional to the price of the crops.

The cost to farmers is the change

$$\Delta R = R(q_{op}) - R(q_{os}) \tag{B.9}$$

in the revenue contribution R due to internalization. In modeling the effects of taxes and tradable permits, the only difference between the tax and permits given out free is the additional cost of the tax. If the external cost is internalized by means of free permits, the change in farmer's revenue contribution is

$$\Delta R = [y(q_{op})p_C - q_{op}p_N] - [y(q_{os})p_C - q_{os}p_N] \tag{B.10}$$

or

$$\Delta R = [y(q_{op}) - y(q_{os})]p_C - \Delta q p_N \tag{B.11}$$

It is instructive to examine the variation with damage cost e in the approximation of expanding the yield curve according to Eq. B.4 (which is actually exact for the quadratic fits of Eq. 6). One obtains

$$-\Delta R = \left[(q_{os} - q_{op})y'(q_{op}) + \tfrac{1}{2}(q_{os} - q_{op})^2 y''(q_{op})\right] p_C + \Delta q\, p_N \tag{B.12}$$

Replacing $(q_{os} - q_{op})$ with $-\Delta q$ and substituting $y'(q_{op})$ from Eq. B.2 gives

$$-\Delta R = -\Delta q\left(\frac{p_N}{p_C}\right)p_C + \tfrac{1}{2}(\Delta q)^2 y''(q_{op})p_C + \Delta q p_N = \tfrac{1}{2}(\Delta q)^2 y''(q_{op})p_C \tag{B.13}$$

and, combining with Eq. B.8, one obtains the result

$$\Delta R = \tfrac{1}{2}\left(\frac{1}{-y''(q_{op})}\right)\left(\frac{e^2}{p_C}\right);\ \text{it is} > 0 \text{ since } y''(q_{op}) < 0. \tag{B.14}$$

If on the other hand the external costs are internalized by a tax, the loss to farmers is that determined in Eq. B.14 plus the tax e on the purchased fertilizer q_{os},

$$\Delta P = \tfrac{1}{2}\left(\frac{1}{-y''(q_{op})}\right)\left(\frac{e^2}{p_C}\right) + q_{os}e. \tag{B.15}$$