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Costs and Benefits of Nitrogen for Europe and Implications for Mitigation

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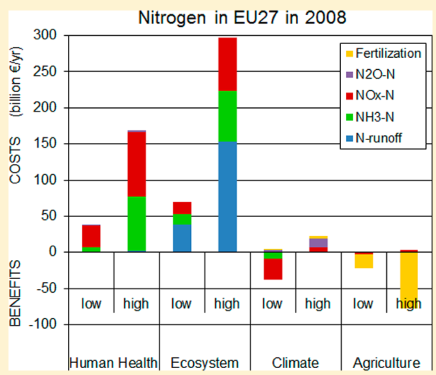
Ⓢ *Supporting Information*

ABSTRACT: Cost-benefit analysis can be used to provide guidance for emerging policy priorities in reducing nitrogen (N) pollution. This paper provides a critical and comprehensive assessment of costs and benefits of the various flows of N on human health, ecosystems and climate stability in order to identify major options for mitigation. The social cost of impacts of N in the EU27 in 2008 was estimated between €75−485 billion per year. A cost share of around 60% is related to emissions to air. The share of total impacts on human health is about 45% and may reflect the higher willingness to pay for human health than for ecosystems or climate stability. Air pollution by nitrogen also generates social benefits for climate by present cooling effects of N containing aerosol and C-sequestration driven by N deposition, amounting to an estimated net benefit of about €5 billion/yr. The economic benefit of N in primary agricultural production ranges between €20−80 billion/yr and is lower than the annual cost of pollution by agricultural N which is in the range of €35−230 billion/yr. Internalizing these environmental costs would lower the optimum annual N-fertilization rate in Northwestern Europe by about 50 kg/ha. Acknowledging the large uncertainties and conceptual issues of our cost-benefit estimates, the results support the priority for further reduction of NH_3 and NO_x emissions from transport and agriculture beyond commitments recently agreed in revision of the Gothenburg Protocol.



■ INTRODUCTION

The global N-cycle is being transformed at a record pace, as global rates of human fixation of atmospheric N_2 to reactive nitrogen (N_r) have increased 20-fold over the last century.[1] Rockström et al.[2] hypothesize that the safe planetary boundary for anthropogenic input of nitrogen is exceeded by around a factor of 3.5. In view of the complexity of the N-cycle, and because of the close relation between production and consumption of N_r via food and energy, Galloway et al.[1] conclude that "optimizing the need for N as a key human resource while minimizing its negative consequences requires an integrated interdisciplinary approach and the development of strategies to decrease N-containing waste". In fact, the challenge is whether the N-cycle can be changed in such a way that a human welfare improvement is achieved, meaning that the sum of social benefits of this change exceed (or at least are in balance with) the sum of all associated social costs.

Wasteful N management results in costs for human health, ecosystems and climate, but improved N management and mitigation of effects is associated with additional costs. These costs should be in balance with benefits from avoided environmental damage and increased agricultural output. This paper describes the development of the nitrogen problem in Europe, provides estimates of costs and benefits of nitrogen for the European Union (current 27 member states, EU27), and explores the potential of cost-benefit assessment to set priorities in an integrated approach to improved management. This critical assessment builds on the European Nitrogen Assessment,[3] which was based on the year 2000, and updates results including additional impacts and emissions for 2008.

The century of N-management in Europe after the Haber-Bosch invention in 1908, has seen the rapid growth in use of industrial fertilizers in agriculture and of energy in industry and transportation resulting in a 3-fold increase of the total N_r production (all N except unreactive N_2) in the EU27, and leading to anthropogenic N_r formation now dominating over natural sources in Europe. This increase was partly offset by a reduction of input of N_r by biological fixation in grassland and semi natural land. By contrast, N-fixation during fossil fuel

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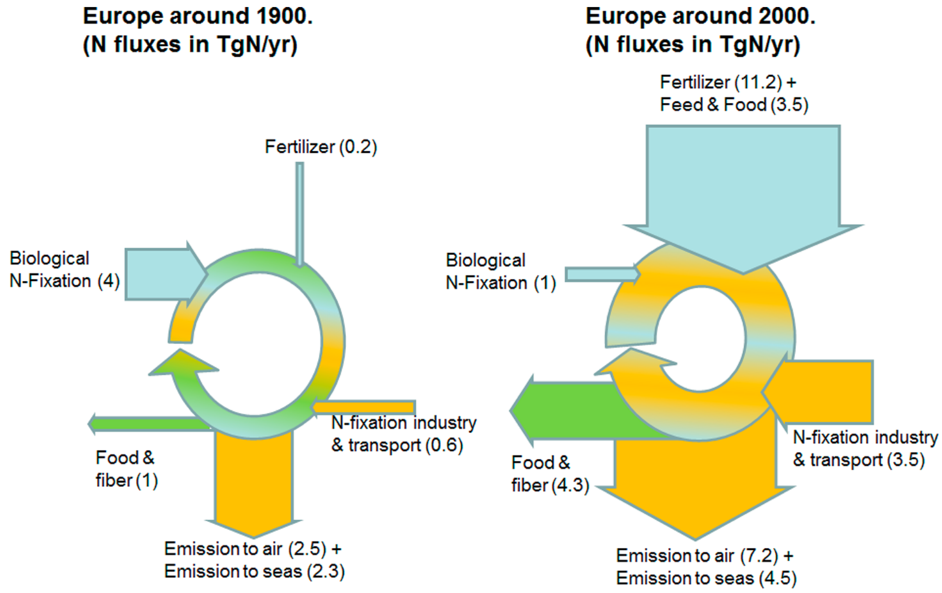

Figure 1. Simplified N cycle for EU-27 in 1900 and 2000; fluxes in Tg per year of reactive N. Fluxes in blue are intentional anthropogenic fluxes, those in orange are unintentional anthropogenic fluxes, fluxes in green refer to beneficial outputs.[4].

combustion has given an additional important source of 3.5 Tg/yr (Figure 1). Almost half of the annual N_r input to the EU27 is lost as N_r to the environment, causing a suite of adverse impacts, the major ones being health damage by ozone and aerosols, where NO_x is the precursor, and eutrophication of terrestrial and aquatic ecosystems. The other half of the annual N input is lost to the atmosphere as unreactive N_2, thus wasting the energy investment in the initial fixation of atmospheric inert N_2, with a much smaller fraction incorporated into food and fiber products. Compared with 1900 total inputs of N_r in 2000 have increased by a factor of 4, and outputs to the environment by a factor 2.5.

After the invention of the Haber-Bosch process, it took nearly half a century before the use of nitrogen fertilizer in the EU27 became common practice in agriculture and increased from 1−2 Tg around 1950 to 11 Tg around the year 2000.[4] As a result of increased cereal yields per hectare,[5,6] the EU could use a large part of its cereal production for animal feed. In combination with substantial imports of protein and energy rich feed-stuff this allowed the strong growth of the pig and poultry sector after 1950. Per capita consumption of animal products in the EU between 1960 and 2007 increased by 50%[7] and doubled relative to 1900[8] (for more detail see Supporting Information (SI)).

■ ENVIRONMENTAL TRENDS AND NITROGEN POLICIES IN THE EUROPEAN UNION

Increased use of chemical fertilizers together with increased numbers of livestock after World War II strongly enhanced emissions of N_r to water and ammonia to air (SI Figure S1). Since the 1980s losses of N_r to the environment in the EU27 have started to decrease as a result of mitigation measures related to EU and national policies addressing air quality, wastewater treatment and application of synthetic N-fertilizer and manures.[9] For example, between 1990 and 2007 the emission of NH_3 (EU27) and the N-surplus on agricultural soils between 1990 and 2004 (EU15) declined by about 20%.[10,11] In line with the decrease in emissions of N_r, nitrate concentrations in major European rivers have decreased since 1990. While this indicates some success of environmental policies, the decrease is small and present nitrate concentrations are as much as 10 times higher than in 1900 (SI Figure S2).

■ ECONOMIC VALUATION OF IMPACTS OF N_r FOR POLICY DEVELOPMENT

Sustainable development includes implementation of environmental policies to protect public health, ecosystems and climate, which can be supported by a broad economic assessment of costs and benefits.[12] For this reason decisions on environmental pollution policies in the EU and US are increasingly based on cost-effectiveness analysis (CEA) and cost benefit assessment (CBA). While CEA supports selection of policy options to achieve environmental targets in the least cost way, CBA helps to identify options where welfare increases most.[13] However, science based CBA and CEA are subject to conceptual difficulties and large uncertainties. In spite of these problems, Constanza[12] concluded that we should choose to make the valuation of policy alternatives explicit. Implicitly, decision making will always involve valuation and weighting, the rationale and implications of which may not be fully understood even by decision makers.

CBA can help to develop consistent policies and provide accountability to decision makers.[14] Birch et al.[15] advocated the use of multiple metrics, both physical and economic, to make proper decisions about abatement of N-management of the Chesapeake Bay area. Taking into account cost-benefit considerations, they concluded that abatement of emission to air is more beneficial than a partial focus on the direct release of riverine and sewage nitrogen into the Bay. Ideally, cost estimates for adverse effects of N_r should be used to internalize these costs, for example to charge the producer or consumer of N_r intensive products and to implement the "Polluter Pays Principle". Similarly, Blotnitz et al.[16] estimated the external cost of nitrogen fertilizer to be 60% of its market price, implying that optimum fertilizer application rates from a social welfare perspective would be far lower than recommended rates from a more narrow agronomic perspective.

Table 1. Marginal Costs and Benefits Between 1995 and 2005 of Different N_r-Threats in EU (See SI Table S1 for Description of N_r-Threats)

effect	emitted nitrogen form	emission/loss to	estimated cost € per kg N_r emitted, used or produced[a]
human health (particulate matter, NO_2 and O_3)	NO_x	air	10−30 (18)
crop damage (ozone)	NO_x	air	1−2
ecosystems (eutrophication, biodiversity)	N_r (nitrate) N_r deposition	surface water	5 to 20 (12)
human health (particulate matter)	NH_3	air	2−20 (12)
climate (greenhouse gas balance)	N_2O	air	4−17 (10)
climate**	NOx	air	−9 to 2 (−3)
climate**	NH_3	air	−3 to 0 (−1)
ecosystems (eutrophication, biodiversity)	NH_3 and NO_x	air	2−10 (2)
human health (drinking water)	N_r (nitrate)	groundwater	0−4 (1)
human health (increased ultraviolet radiation from ozone depletion)	N_2O	air	1−3 (2)
climate (N-fertilizer production)	N_2O, CO_2	air	0.03−0.3
crop yield increase (benefit): 1st year	N-fertilizer	soil	0.5−3 (1.7)
long term			1.5−5 (3.7)

[a]Values in between brackets are the single values that were inferred from studies on individual effects, for details see SI and Brink et al.[14] **Cooling effects include N deposition and ozone effects on forest carbon sequestration, direct and indirect effects of N containing aerosol and other smaller effects.[18]

■ CALCULATION OF COSTS AND BENEFITS

The concepts and data for estimating the cost and benefits of nitrogen in the EU27 were based on the European Nitrogen Assessment[3,14,17,18] (SI Table S1). The economic value of N-damage was based on standard economic concepts and methods for valuation of health impacts (estimating costs of treatment, lost productivity and willingness to pay (WTP) to reduce risk of premature death or pain and suffering), to restore ecosystems or to reduce greenhouse gas emissions. We acknowledge that the mix of methods used here introduces additional uncertainties. For example, the use of "restoration costs" for terrestrial ecosystem damage begs a number of questions. Is society willing to pay for the restoration? Does "restoration" fully capture the disbenefit to society? To account for this additional uncertainty we have set the upper bound of the marginal damage cost of N deposition on terrestrial ecosystems at five times the restoration cost based on scarce data for WTP for N related ecosystem services (see SI Table S1). Acceptance of abatement costs for reducing greenhouse gas emissions is also problematic in the context of this paper, as there is clearly no reason why abatement costs should reflect damage caused. However, for the purpose of this paper the authors consider that it is better to provide the best available indication of overall cost than to omit certain elements altogether, particularly those for which the literature suggests importance.

The environmental damage that can be caused by a unit amount of N_r emission not only depends on the form of N_r but also on local conditions for transport and exposure of humans and ecosystems. As a result differences of unit damage costs between individual EU member states can amount to a factor of 20. WTP is a debatable method for economic valuation of environmental goods and services as preferences of individuals and groups are subject to change and manipulation.[12] The outcome will depend on context and information of the surveys and income, background and education of the people being interviewed. For example, in the assessment of WTP of inhabitants of Baltic states for a clean, uneutrophicated Baltic sea, inferred WTP in 2005 of households ranged between around 100 €/yr in Russia, Poland and the Baltic states to 700−800 €/yr in Sweden and Denmark.[19] For economic evaluation of human health impacts we used a fixed value of a human life year of €40,000 in all EU in accordance with Desaigues et al.,[20] although in reality WTP for a longer and healthier life also depends on income.

In this study the economic value of an environmental impact was linked to nitrogen by dividing the economic loss (€/yr) by the value of the associated N_r flux (kg/yr).[14] For human health impacts we assumed no threshold value for impacts of N_r emissions. We used emission data for the year 2008. Marginal damage costs derived from WTP studies apply to various years between 1995 and 2005 and were not corrected for inflation or changes of income between 1995 and 2008.

The economic value of the direct benefits of N_r in agriculture was based on yield response of cereals, oil seed rape, and milk to N input and current world market prices (Table 1). Distinction was made between yield response to N_r in the first year after fertilizer application in common crop rotations, and the long-term response inferred from continuous wheat trials,[14] [SI]. In common Northwest European rotations, wheat cultivation benefits to a large extent from residual N from more N-demanding preceding crops as potato and temporary grassland. The monetary value of nitrogen benefits for wheat cultivation are based on prices in 2006 for synthetic fertilizer (Calcium-Ammonium-Nitrate, CAN) of 0.8 €/kgN and for wheat of 125 €/tonne. Representative emission factors for NH_3, NO_x and N_2O from fertilizer and manure were taken from the 2011 version of the GAINS model[21,22] (http://gains.iiasa.ac.at) and NO_3 leaching fractions for loamy soils as observed in Rothamsted.[23] Cost of measures to reduce nitrogen emission of NO_x and NH_3 were also derived from the GAINS model.

■ COST OF NITROGEN POLLUTION IN THE EU27

The estimated mean marginal social costs in the EU27 for N_r per unit of emission to the environment (Table 1) show a wide variation between different N_r compounds and contain uncertainties principally related to WTP and to dose−response relationships. The highest unit damage cost values are associated with air pollution effects via NO_x on human health, followed by the effects of N_r loss to water on aquatic

Table 2. N_r Emissions and Associated Damage Costs in the European Union (EU27) in 2008. Cost Values Are Rounded to the Nearest 5 Billion € to Avoid Over Precision

	NO_x emission to air	NH_3 emission to air	N_r loss to rivers and seas	N_2O emission to air	total
emission 2008 (Tg N/yr)	3.2	3.1	4.6	0.8	11.7
agricultural share (%)	4	85	60	42[a]	52
N Cost for All Sources (Billion € Per Year)					
human health	30−90	10−75	<5[b]	<5[c]	40−170
ecosystems	15−75	15−70	40−155		70−300
climate	−30 to 5	−10 to 0		5−15	−35 to 20
total cost	20−170	15−145	40−155	5−15	75−485
N Cost for Agricultural Sources					
human health	0−5	5−65	<5	<5	10−70
ecosystems	<5	15−60	25−95		35−155
climate	0	−10 to 0		0−5	−10 to 5
total cost	0−10	10−120	25−100	0−5	40−230

[a]Agricultural emissions using emission factor of 1% for direct and 0.75% for indirect N_2O emission which are lower than default IPCC values (more detail in SI Table S2). [b]Cost is 0−2 billion €/yr and based on N leaching flux to groundwater under agricultural land. [c]Cost is 1−2 billion €/yr.

ecosystems and the effects via NH_3 on human health. Our unit damage cost value of 18 €/kg N (range 10−30) for human health impacts by NO_x is similar to that by Birch et al.[15] The smallest unit damage cost values were found for the effects of nitrates in drinking water on human health and the effect of N_2O on human health by depleting stratospheric ozone. The damage costs related to the production of chemical N fertilizer are small compared with the cost from losses of N_r compounds at application of fertilizer and manure.

The total social damage cost associated with emission of various N_r compounds (Table 2, SI Figure S3) is obtained by scaling low, high and mid values of marginal damage costs (Table 1) with the actual levels of N_r emissions in 2008 for each member state in the EU27 (SI Table S2). The total environmental cost of N_r was estimated at €75−€485 billion per year. Using the single values of the marginal N cost, 60% of the total cost of N is related to emissions to air and 40% to emission to water. About 60% of the cost is related to impacts on ecosystems, 40% to impacts on human health. There is a small but rather uncertain net climate cooling due to current N_r emissions,[18] with associated benefits (Table 2). A major source of uncertainty is the value of health impacts of NH_4- (and NO_3-) containing secondary particulate matter. The upper bound for the health costs assumes that secondary airborne particles are equally hazardous as primary particles in line with current recommendations for assessment by the WHO.[24] The lower bound reflects minor health impacts of inhalation of nitrate and ammonium containing salt particles.[25] The wide range of total environmental cost of N_r is not only caused by the aforementioned uncertainties but also by the presence of both climate costs and benefits of N_r. The damage cost results for 2008 are higher than our earlier estimates for 2000 in the European Nitrogen Assessment,[14] although emissions of both NO_x and NH_3 have decreased by 18% and 11%, respectively. Major reasons for our higher estimates are consideration of damage to aquatic ecosystems from atmospheric deposition which is consistent with findings for the Baltic,[19] the use of country specific unit damage cost data instead of mean values for the EU27 and using updated model results for N loads to rivers and seas (SI Table S2, Figure S3).

The total damage cost in 2008 equates to €150−€1150 per person. The mean annual per capita cost of nearly €500 was about twice as high as the WTP to prevent global warming by carbon emissions trading, which is 100−300 €/capita (taking a CO_2-eq emission of 11 tonne/capita (2004) and CO_2 emission trading price of 10−30 €/tonne). The cost of N pollution was found to create a welfare loss equivalent to 2% (uncertainty 0.6−4%) of the Gross Domestic Product (in Purchase Power Parity). The relative loss of welfare is somewhat higher in regions and countries with a N intensive agriculture and a low population density. Then perceived costs of coastal water eutrophication per inhabitant will be high due to high N inputs from rivers and by atmospheric deposition and due to a high WTP to prevent eutrophication. This feature is most prominent for Ireland but is also found for Scandinavia (Denmark) and France (Figure 2, SI Figure S3).

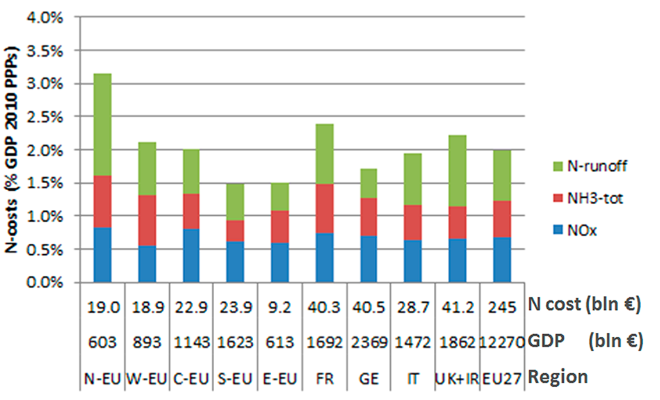

Figure 2. Loss of welfare in 2008 due to impacts of emissions of NH_3, NO_x, to air and of N to water on human health and ecosystems in regions and large member states in the EU27. GDP in Purchase Power Parity. FR: France, GE: Germany, IT: Italy, UK+IR: United Kingdom and Ireland, N: Sweden, Finland and Denmark, W: Netherlands, Belgium and Luxembourg, C: Austria, Czech Republic, Slovakia and Poland S: Spain, Portugal, Greece, Cyprus, and Malta, E: Baltic states, Bulgaria, Romania, Hungary, and Slovenia.

COSTS AND BENEFITS OF N-MITIGATION

The estimates of potential welfare loss due to N_r in the EU27 allow an indication of the maximum level of emission reduction, and associated mitigation costs, up to which there is a net gain of welfare. Combining the marginal mitigation cost curves for individual EU27 member states, as used in the GAINS-model for the revision of the Gothenborg Protocol[26,27] with the marginal damage cost values (Table 1), allows calculation of

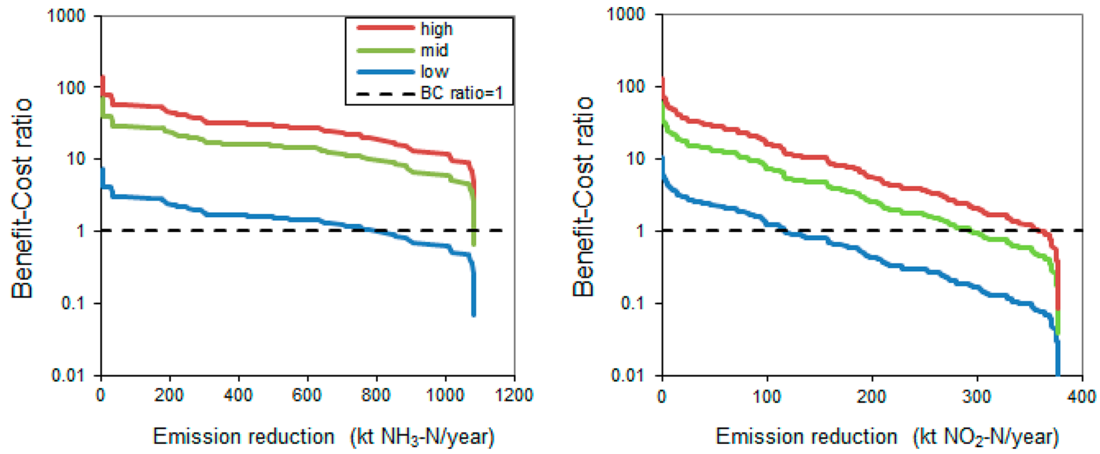

Figure 3. Ratio of marginal benefits of emission reduction over the costs of N-mitigation measures in EU27 for NH_3 and for NO_x from stationary sources, for emission reduction beyond expected levels in 2020 (3030 ktonne/yr for NH_3–N and 1730 ktonne/yr NO_x-N) by effects of current legislation.[26,27].

marginal Benefit-Cost ratios (BCR; Figure 3). We examined the optimum level of emission reduction beyond levels projected in 2020 by effects of current legislation.[26] Considering the uncertainty of marginal damage costs (Table 1), a welfare increase for NH_3 is to be expected up to a reduction of 800–1090 ktonne/yr (25–35% of projected emission for EU27 in 2020), and for NO_x emissions from stationary sources up to 120–360 ktonne/yr (10%-20% of projected emission in 2020[26]).

The modest proportion of NO_x emission reduction with BCR exceeding one (Figure 3), implies that technological innovation is required to combine achievement of lower pollution levels with robust welfare increase. Conversely, for NH_3 there is a large potential for low cost emission reduction measures. The reductions would be achieved through improved N use efficiency that results in lower expenditures on chemical fertilizer. For N_r emissions to water there are no marginal cost curves for EU27 and emission reduction measures are more country and region specific than for NO_x and NH_3.

COSTS AND BENEFITS OF N-FERTILIZATION

In contrast to N_r production from combustion, use of N_r in agriculture is intentional to increase production. Aggregating marginal costs of N_r damage (Table 1) for the EU27 gives an annual total cost between 35 and 230 billion € from agricultural emissions of N_r (Table 2). The relative contributions of the individual pollutants are 46% for NH_3, 48% for N-runoff, 2% for NO_x and 3% for N_2O.

A first estimate of the direct benefit of N-fertilization (synthetic and organic) for farmers was obtained using N response curves and world market prices of winter wheat, which is the major crop in the EU27, milk and oilseed rape. Annual benefits were estimated at 10–50 billion €, and 20–80 billion € when including a high estimate of long-term N-benefits (Table 1). The wide range of the N-benefits reflects the wide variation of N-response of wheat across the EU.[14] This estimate of benefits does not include the added value that is created in the food chain using primary agricultural products. Based on Eurostat data (http://epp.eurostat.ec.europa.eu/portal/page/portal/statistics/), the gross added value created in the primary agricultural sector for the EU27 in 2008 was 170 bln €/y (65% for crops, 35% for livestock). Taking the simple wheat-based estimate of N-benefits of 20–80 billion € for the total agricultural area, N would be responsible for 15–50% of the added value of primary production, which is consistent with estimates by Smil[28] and Erisman et al.[29] The gross added value created in the total agrofood sector of the EU27 in 2008 was 350 bln €/y so twice that of the primary sector. One may argue to include a multiplier of two in the assessment of the N-benefits to account for this difference, but we did not as these multipliers may also be present for N costs considering long-term impacts of loss of ecosystem services or climate stability.

Our estimates of social cost of N-fertilization in EU27 tend to exceed the contribution of N-fertilization to the gross added value of the primary agricultural sector by 70 billion € per year (for the mid values of unit damage costs). Only when taking the lower bound of the cost and the upper bound of the benefits there would be a net benefit of N fertilization for society of 70 billion € per year. The potential absence of net social benefits may be due to counting only the benefits in Table 1 that accrue to farmers as revenue from increased yields. In spite of using a low estimate of the benefits from N fertilization, the wide range indicates there is a large scope to increase the welfare gains from N fertilization. The obvious options are reducing emissions NH_3 and NO_3 (end-of-pipe measures), as these N_r emissions generate most social costs. Increasing nitrogen use efficiency would reduce N-surpluses and hence all N_r emissions (including start-of-pipe measures).

The absence of robust net social benefits of N fertilization is most plausible for N_r in manure and urea fertilizers because of the substantial fraction of N_r lost as NH_3. In the EU27, about 35% of the total N input to the soil is from manure, while 50% is from synthetic fertilizer.[30] Most manure is applied to grassland and fodder crops and about 10–20% to food cereals and tubers. Considering an example of typical wheat cultivation on loamy soils in Northwestern EU, where calcium ammonium nitrate (CAN) fertilizer is used, the ratio of N benefits over N costs would range between 0.5 and 7 (Figure 4), indicating the tendency of CAN to generate net benefits. However, when half of the effective N input is applied as manure the benefit-cost ratio ranges between 0.1 and 2.4 (assuming a fertilizer equivalence of 60% for N in manure). These low ratio's suggest that ample and inefficient use of manure will generate net social costs. Such a high share of manure in total N input on arable crops is typical for regions with high concentrations of pig and poultry farming as present in many parts of Europe like

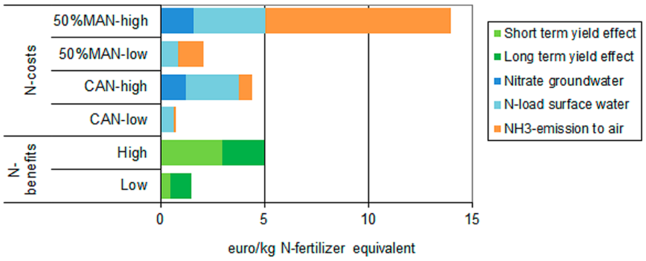

Figure 4. Marginal costs and benefits of nitrogen fertilization to winter wheat on a sandy-loamy soil for a case with Calcium Ammonium Nitrate (CAN) fertilizer and a case with 50% of effective N applied as manure (50%MAN). Only costs of the dominant emissions of Nr to water and of NH_3 were considered.

the south of The Netherlands, the Flemish region, and Brittany (France).[30] Our results underscore the importance of policies stimulating Good Agricultural Practices (Best Management Practices), and especially those that reduce the significant N_r losses from animal manures.

INTERNALIZING EXTERNAL COSTS OF NITROGEN FERTILIZATION

Farmers account for costs of fertilizer purchase by balancing the increase in production against the additional fertilizer price, so as to estimate the agronomically optimum rate of N fertilization. This approach can be extended by incorporating the increased social costs of pollution by N fertilization to health, ecosystems, and climate, so as to calculate the "socially optimum N rate" introduced earlier by Blottnitz et al.[16] Using a fertilizer N price of 0.8 €/kgN (CAN) and a wheat price of 125 €/tonne sets the agronomically optimum N rate for the farmer at 175 kgN/ha (Figure 5). By contrast, including the external N

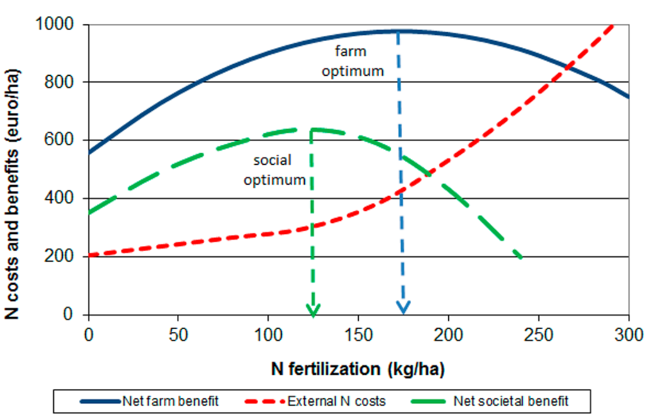

Figure 5. Benefits and costs of nitrogen fertilization (CAN) on winter wheat. (N response based on Henke et al.[31] which is representative for German conditions.).

costs of N losses due to fertilization sets the smaller social optimum N rate at 120 kgN/ha. Using different assumptions about prices of fertilizer and crops, N-response models and emission factors for N_r compounds, the difference between the social optimum and farm optimum N rate was calculated at 30−90 kg/ha (median value 55 kg/ha). This difference is similar to Brentrup et al.[23] who found a difference of 50−100 kg/ha for winter wheat. To maintain high cereal yields at a lower N rate, the N use efficiency may need to be increased to compensate for a possible decrease of the soil N status.

Our estimates of external costs per kg of applied N as CAN, range between 0.4 and 7 € per kg N and are far higher than estimated by Blottnitz et al.[16] who reported 0.3 €/kg of applied N. In Blottnitz et al. 0.25 €/kg N was related to greenhouse gas emission during production and application of N fertilizer, which is comparable with the present estimates. A major reason that external costs in our assessment are much higher is that for N related eutrophication we used WTP to prevent ecosystem damage (0.3−4 €/kg added N for fresh water and marine[19]) while Blottnitz et al.[16] used damage costs of 0.01−0.065 €/kg added N for freshwater eutrophication only, based on Pretty et al.[32] Pretty et al. estimated damage costs for eutrophication by combining costs of a mixture of actual and planned abatement measures and loss of benefits from ecosystem usage. Possible explanations for this order-of-magnitude difference are that costs by Pretty et al. (i) did not cover all water impacts, (ii) included impacts and measures that do not lead to full restoration of the ecosystem and its services, and (iii) did not consider impacts for the marine environment. Furthermore Blottnitz et al.[16] did not take into account the health impacts of NH_3 through particulate matter formation.

SOCIETAL IMPLICATIONS OF OPTIMIZING NITROGEN FERTILIZATION

Implementing a socially optimum N fertilization rate would not be without consequences. Current annual cereal yield in high production areas of Northwestern Europe is between 7 and 9 tonne/ha. A decrease of the rate on winter wheat from 175 to 125 kg/ha would decrease the yield by more than 1 tonne/ha. However, while implementation of a policy targeting optimum N rates would initially reduce the total cereal production, it would also tend to increase market prices of cereal. This could lead to an increase of cereal production in other areas of the EU where N input rates are lower, or a decreasing demand for cereals. Wheat yields in central, southern and eastern parts of the EU are 2−5 tonne/ha below yields in Northwestern member states[6] and could be increased substantially by improvement of inputs and management of , for example, nutrients and water. The EU Common Agricultural Policy (CAP), with an annual budget of 70 bln € and current environmental directives could provide the means, conditions and the instruments for a spatial optimization of agricultural production in the EU. Using our CBA results, a translocation of agricultural production in EU from northwest to east would create net social benefits in both regions. But in case of feed crops such a translocation will have implications for the livestock sector as a whole, in view of economic restrictions for transport of feed to farm, of manure to fields and of livestock products to processing plants and markets. Determining the optimum spatial configuration would require sophisticated models taking into account a.o. land use change, including indirect land use change (outside the EU27), and global commodity markets. Equally important is the need to devise appropriate regulatory or economic instruments that utilize the CAP budget to allow the wider societal costs of N_r pollution to be integrated into profitable farm economics. This will require recognizing the full societal value of N inputs, pointing to the need to improve the nitrogen use efficiency of both mineral fertilizers and manures. Such approaches are surely key to developing an appreciation for these issues among farmers and their advisers, to overcome risk aversion, thereby reducing the tendency to overapply fertilizer, as well as to overcome the common perception that manures are inferior fertilizers.[33] On

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the other hand in EU countries with strict environmental regulations, like Denmark, The Netherlands and the Flemish Region there are already strong incentives to optimize N-application and utilize N in manure, with efficiencies up to 70% achieved as compared with CAN.[30]

IMPLICATIONS OF NITROGEN COSTS AND BENEFITS FOR EUROPEAN AIR POLLUTION POLICY

In a CBA conducted in support of the EU's Thematic Strategy on Air Pollution (TSAP) Pye et al. also concluded that society as a whole benefits from abatement of N_r air pollution based on GAINS results.[34] For the cost-effective (CE) scenario conducted in that study to meet the TSAP targets by 2020, emission reduction in addition to current legislation (CLE) was 9.3% for NO_x and 15.3% for NH_3. The total cost of CLE in 2020 was 80 bln euro/yr, 65% of which was for reducing emissions from transport. The additional cost of the CE scenario, including costs for reduction of SO_2, particulate matter and volatile organic compounds, was estimated at 1.5 bln €/yr. This cost represents just 0.01% of the gross domestic product of the EU27. Pye et al.[34] estimated monetary benefits in the CE scenario to range between 22 and 70 bln €/yr using with a BC ratio for the EU27 of at least 15. This ratio is in the range of the marginal BCRs of 2−32 for NH_3 and 1−8 for NO_x in this study (Figure 3). Important differences with the present study are that Pye et al. did not monetize ecosystem impacts and also considered other pollutants. In Pye et al., health gains from emission reduction of non N containing particulate matter was the dominant benefit of CE. Using our marginal damage cost data for NO_x and NH_3 (Table 1), the emission reductions in the CE scenario in Pye, et al. would generate social benefits of 5−30 bln €/yr and a BCR of 4−20. These results indicate that the benefits of prevention of air pollution will outweigh the costs of mitigation. This is consistent with the conclusion in the Stern report[35] for mitigation of climate change. In the present case, the benefits are even more apparent as emission reduction measures for climate protection will have limited effects in the coming 50 years, while many of the benefits of nitrogen abatement are almost immediate (health effects) or can be anticipated within a few years to decades (water, ecosystem effects). This was clearly demonstrated for the 2008 Olympic Games in Bejing where NO_2 concentration decreased by over 40% within one year and significant effects on biomarkers for cardiovascular health were observed.[36]

The emission ceilings for NH_3 in 2020 under the Gothenburg Protocol agreed by EU member states in May 2012 are only 6% below 2005 levels[37] (equivalent to just 2% below 2010 levels). The optimum ceiling for the EU27 that could be inferred from BCRs in the present study (Figure 3) are 26% to 36% below the expected level in 2020. Lower ceilings for NH_3 also increase the nitrogen use efficiency that contributes to reduction of N_2O emissions.[38] For NO_x, the ceilings negotiated in the revised Gothenburg Protocol are 42% below 2005 levels and approach optimum levels for welfare optimization in this study. Judging from Figure 3, an additional reduction of NO_x emissions from stationary sources beyond 2020 up to 350 ktonne/yr (7−20%) could create net benefits. The BCRs estimated in this study would justify substantial further abatement of both NH_3 and NO_x emissions than was agreed in the recent revisions of the Gothenburg Protocol.

COST BENEFIT ANALYSIS TO GUIDE FUTURE NITROGEN POLICIES

The findings of the present analysis provide a strong support for future initiatives for stricter emission and concentration targets for N_r. We nevertheless recognize considerable uncertainties and conceptual challenges in such a monetized valuation of classically noncommensurable issues. Such uncertainties are inevitable and need to be recognized in supporting the development of future nitrogen policies. In essence, our approach is a method to weigh, compare and add up the multiple effects of autonomous developments and environmental policies on nitrogen pollution, which is almost a prerequisite for evaluating and building integrated policies. In particular for emissions from manure and fertilizer use, our results show that there is large potential to increase nitrogen efficiencies and reduce N_r losses, with limited effects on agricultural production in the EU. For NO_x emissions from energy and transportation there is also scope for improvement, but the emission reduction range with robust net social N_r benefits is modest in view of the steep marginal mitigation cost curves (Figure 3). Our comparison of effects also indicate that N policies should not single out the direct greenhouse effect of nitrous oxide as on the short-term this is a smaller source of damage costs than NO_x, NH_3 and N-runoff (Table 2). In the long run toward 2100 the relative share of N_2O in total N cost is expected to increase because of the modest potential of reducing N_2O emissions from agriculture, long residence time of N_2O in the stratosphere and because anticipated future mitigation of air pollution by NO_x and NH_3 would tend to reduce the short-term cooling effects of Nr. This highlights the need to improve overall nitrogen use efficiency, leading to a simultaneous decreases in N_r losses from all sources over time.

ASSOCIATED CONTENT

Supporting Information

Additional information and data on emissions and costs. This material is available free of charge via the Internet at http://pubs.acs.org.

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Notes

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REFERENCES

(1) Galloway, J. N.; Townsend, A. R.; Erisman, J. W.; Bekunda, M.; Cai, Z.; Freney, J. R.; Martinelli, L. A.; Seitzinger, S. P.; Sutton, M. A. Transformation of the nitrogen cycle: Recent trends, questions and potential solutions. *Science* **2008**, *320*, 889−892.

(2) Rockstrom, J.; Steffen, W.; Noone, K.; Persson, A.; Chapin, F. S.; Lambin, E. F.; Lenton, T. M.; Scheffer, M.; Folke, C.; Schellnhuber, H. J.; Nykvist, B.; De Wit, C. A.; Hughes, T.; van der Leeuw, S.; Rodhe, H.; Sorlin, S.; Snyder, P. K.; Costanza, R.; Svedin, U.; Falkenmark, M.; Karlberg, L.; Corell, R. W.; Fabry, V. J.; Hansen, J.; Walker, B.; Liverman, D.; Richardson, K.; Crutzen, P.; Foley, J. A. A safe operating space for humanity. *Nature* **2009**, *461*, 472−475.

(3) Sutton, M. A., Howard, C. M., Erisman, J. W., Billen, G., Bleeker, A., Grennfelt, P., van Grinsven, J. J. M., Grizzetti, B., Eds. *The European Nitrogen Assessment*; Cambridge University Press: Cambridge, U.K., 2011; http://www.nine-esf.org/ENA-Book.

(4) Bouwman, L.; Klein Goldewijk, K.; van der Hoek, K. W.; Beusen, A. H. W.; van Vuuren, D. P.; Willems, W. J.; Rufino, M. C.; Stehfest, E. Exploring global changes in nitrogen and phosphorus cycles in agriculture induced by livestock production for the period 1900−2050. *Proc. Natl. Acad. Sci. U.S.A.* **2011**, DOI: 10.1073/pnas.1012878108.

(5) FAO. *Yearbook Food and Agricultural Statistics*; FAO: Washington, DC, 1950.

(6) FAOSTAT. http://faostat.fao.org/.

(7) Westhoek, H. J.; Rood, G. A.; van den Berg, M.; Janse, J. H.; Nijdam, D. S.; Reudink, M. A.; Stehfest, E. E. The protein puzzle: The consumption and production of meat, dairy and fish in the European Union. *Eur. J. Food Res.Rev.* **2011**, *1* (3), 123−144.

(8) Smil, V. Eating meat: Evolution, patterns and consequences. *Popul. Dev. Rev.* **2002**, *28* (4), 599−639.

(9) Oenema, O.; et al. Nitrogen in current European policies. In *European Nitrogen Assessment*; Sutton, M. A., Howard, C. M., Erisman, J. W., Billen, G., Bleeker, A., Grennfelt, P., van Grinsven, H. J. M., Grizzetti, B., Eds.; Cambridge University Press: Cambridge, U.K., 2011; pp 612.

(10) Tarrasón, L., Nyíri, Á., Eds. *Transboundary Acidification, Eutrophication and Ground Level Ozone in Europe in 2006*; EMEP Status Report 2008, Norwegian Meteorological Institute, 2008.

(11) *Environmental Performance of Agriculture in OECD Countries Since 1990*; Paris, France, 2008.

(12) Constanza, R. Thinking broadly about costs and benefits in ecological management. *Integr. Environ. Assess. Manage* **2006**, *2* (2), 166−173.

(13) Dame, E.; Holland, M. Cost-benefit analysis and the development of acidification policy in Europe. *Water, Air, Soil Pollut.* **2001**, *130* (1−4 III), 1817−1824.

(14) Brink, C.; et al. Costs and benefits of nitrogen in the environment. In *European Nitrogen Assessment*; Sutton, M. A., Howard, C. M., Erisman, J. W., Billen, G., Bleeker, A., Grennfelt, P., van Grinsven, H. J. M., Grizzetti, B., Eds.; Cambridge University Press: Cambridge, U.K., 2011; pp 612, http://www.nine-esf.org/sites/nine-esf.org/files/ena_doc/ENA_pdfs/ENA_c22.pdf and Supplementary material http://www.nine-esf.org/sites/nine-esf.org/files/ena_doc/ENA_supp/ENA_supp_c22.pdf.

(15) Birch, M. B. L.; Gramig, B. M.; Moomaw, W. R.; Doering, O. C., III; Reeling, C. J. Why metrics matter: Evaluating policy choices for reactive nitrogen in the Chesapeake Bay watershed. *Environ. Sci. Technol.* **2011**, *45* (1), 168−174.

(16) Blottnitz, H.; von; Rabl, A.; Boiadjiev, D.; Taylor, T.; Arnold, S. Damage costs of nitrogen fertilizer in Europe and their internalization. *J. Environ. Plann. Manage* **2006**, *49*, 413−433.

(17) Stouman Jensen, L.; et al. Benefits of nitrogen for food, fibre and industrial production. In *European Nitrogen Assessment*; Sutton, M. A., Howard, C. M., Erisman, J. W., Billen, G., Bleeker, A., Grennfelt, P., van Grinsven, H. J. M., Grizzetti, B., Eds.; Cambridge University Press: Cambridge, U.K., 2011; pp 612.

(18) Butterbach-Bahl, K., et al. Nitrogen as a threat to the European greenhouse balance. In *European Nitrogen Assessment*; Sutton, M. A., Howard, C. M., Erisman, J. W., Billen, G., Bleeker, A., Grennfelt, P., van Grinsven, H. J. M., Grizzetti, B., Eds.; Cambridge University Press: Cambridge, U.K., 2011; pp 612.

(19) Gren, I.-M. *Costs and benefits from nutrient reductions to the Baltic Sea*. The Swedish Environmental Protection Agency; Stockholm, Sweden, 2008.

(20) Desaigues, B.; Ami, D.; Bartczak, A.; Braun-Kohlová, M.; Chilton, S.; Czajkowski, M.; Farreras, V.; Hunt, A.; Hutchison, M.; Jeanrenaud, C.; Kaderjak, P.; MácA, V.; Markiewicz, O.; Markowska, A.; Metcalf, H.; Navrud, S.; Nielsen, J. S.; Ortiz, R.; Pellegrini, S.; Rabl, A.; Riera, R.; Scasny, M.; Stoeckel, M.-E.; Szántó, R.; Urban, J. Economic valuation of air pollution mortality: A 9-country contingent valuation survey of value of a life year (VOLY). *Ecol. Indic.* **2011**, *11* (3), 902−910.

(21) Klimont, Z.; Winiwarter, W. *Integrated Ammonia Abatement—Modelling of Emission Control Potentials and Costs in GAINS*; IIASA IR 11−027: Laxenburg, Austria, 2011; http://webarchive.iiasa.ac.at/Admin/PUB/Documents/IR-11-027.pdf.

(22) Amann, M.; Bertok, I.; Borken-Kleefeld, J.; Cofala, J.; Heyes, C.; Höglund-Isaksson, L.; Klimont, Z.; Nguyen, B.; Posch, M.; Rafaj, P.; Sandler, R.; Schöpp, W.; Wagner, F.; Winiwarter, W. Cost-effective control of air quality and greenhouse gases in Europe: Modeling and policy applications. *Environ. Model. Software* **2011**, *26* (12), 1489−1501.

(23) Brentrup, F.; Küsters, J.; Lammel, J.; Barraclough, P.; Kuhlmann, H. Environmental impact assessment of agricultural production systems using the life cycle assessment (LCA) methodology II. The application to N fertilizer use in winter wheat production systems. *Eur. J. Agron.* **2004**, *20*, 265−279.

(24) Moldanová, J.; et al. Nitrogen as a threat to European air quality. In *European Nitrogen Assessment*; Sutton, M. A., Howard, C. M., Erisman, J. W., Billen, G., Bleeker, A., Grennfelt, P., van Grinsven, H. J. M., Grizzetti, B., Eds.; Cambridge University Press: Cambridge, U.K., 2011; pp 612.

(25) Reiss, R.; Anderson, E. L.; Cross, C. E.; Hidy, G.; Hoel, D.; McClellan, R.; Moolgavkar, S. Evidence of health impacts of sulfate- and nitrate-containing particles in ambient air. *Inhalation Toxicol.* **2007**, *19*, 419−449.

(26) Amann, M.; Bertok, I.; Borken-Kleefeld, J.; Cofala, J.; Heyes, C.; Höglund-Isaksson, L.; Klimont, Z.; Rafaj, P.; Schöpp, W.; Wagner, F. *Cost-Effective Emission Reductions to Improve Air Quality in Europe in 2020, Analysis of Policy Options for the Eu for the Revision of the Gothenburg Protocol*; NEC Scenario Analysis Report Nr. 8: IIASA, Laxenburg, Austria., 2011.

(27) Wagner, F.; Winiwarter, W.; Klimont, Z.; Amann, M.; Sutton, M. A. *Ammonia Reductions and Costs Implied by the Three Ambition Levels Proposed in the Draft Annex IX to the Gothenburg Protocol*; CIAM report 5/2011, IIASA: Laxenburg, Austria., 2012.

(28) Smil, V. Nitrogen and food production: Proteins for human diets. *Ambio* **2002**, *31* (2), 126−131.

(29) Erisman, J. W.; Sutton, M. A.; Galloway, J.; Klimont, Z.; Winiwarter, W. How a century of ammonia synthesis changed the world. *Nat. Geosci.* **2008**, *1*, 636−639.

(30) van Grinsven, H. J. M.; ten Berge, H. F. M.; Dalgaard, T.; Fraters, B.; Durand, P.; Hart, A.; Hofman, G.; Jacobsen, B. H.; Lalor, S. T. J.; Lesschen, J. P.; Osterburg, B.; Richards, K. G.; Techen, A.-K.; Vertès, F.; Webb, J.; Willems, W. J. Management, regulation and environmental impacts of nitrogen fertilization in Northwestern Europe under the Nitrates Directive; A benchmark study. *Biogeosciences* **2012**, *9*, 5143−5160, DOI: 10.5194/bg-9-5143-2012.

(31) Henke, J.; Breusted, G.; Sieling, K.; Kage, H. Impact of uncertainty on the optimum nitrogen fertilization rate and agronomic, ecological and economic factors in an oilseed rape based crop rotation. *J. Agric. Sci.* **2007**, *145*, 455−468.

(32) Pretty, J. N.; Mason, C. F.; Nedwell, D. B.; Hine, R. E.; Leaf, S.; Dils, R. Environmental costs of freshwater eutrophication in England and Wales. *Environ. Sci. Technol.* **2003**, *37*, 201−208.

(33) Sheriff, G. Efficient waste? Why farmers over-apply nutrients and the implications for policy design. *Rev. Agric. Econ.* **2005**, *27*, 542−55.

(34) Pye, S., Holland, M.; van Regemorter, D.; Wagner, A.; Watkiss, P. Analysis of the Costs and Benefits of Proposed Revisions to the National Emission Ceilings Directive. In *National Emission Ceilings for 2020 based on the 2008 Climate & Energy Package*, , 2008.
(35) Stern, N. *The Economics of Climate Change, The Stern Review*; Cambridge University Press, Cambridge, U.K., 2006.
(36) Rich, D. Q.; Kipen, H. M.; Huang, W.; Wang, G.; Wang, Y.; Zhu, P.; Ohman-Strickland, P.; Hu, M.; Philipp, C.; Diehl, S. R.; Lu, S.-E.; Tong, J.; Gong, J.; Thomas, D.; Zhu, T.; Zhang, J. Association between changes in air pollution levels during the Beijing Olympics and biomarkers of inflammation and thrombosis in healthy young adults. *JAMA, J. Am. Med. Assoc.* **2012**, *307*, 2069−2077.
(37) Ågren, C. New Gothenburg protocol adopted. *Acid news* **2012**, *2*, 3−5.
(38) Sutton, M. A.; Oenema, O.; Erisman, J. W.; Leip, A.; van Grinsven, H.; Winiwarter, W. Too much of a good thing. *Nature* **2011**, *472*, 159−161.