Science for Environment Policy

IN–DEPTH REPORT

Nitrogen Pollution and the European Environment

Implications for Air Quality Policy

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Executive summary

Nitrogen Pollution and the European Environment

Nitrogen pollution's wide-ranging impacts include contributions to global warming, acid rain and eutrophication. Scientists have suggested that societal capture of atmospheric nitrogen needs to be reduced to around a quarter of its current volume if we are to avoid crossing a 'deleterious or even catastrophic... planetary boundary' (Rockström et al., 2009b). To achieve this, EU Member States, who collectively emit more than the global average, would need to make an even greater cut in emissions.

Around 10,638 gigagrams of nitrogen are added to soils as synthetic fertilisers each year in the EU-27. Much of this is lost to the atmosphere as gaseous emissions.

This In-depth Report from Science for Environment Policy summarises scientific studies and research results on nitrogen pollution in the European environment. Of the many aspects of nitrogen pollution, five specific topics are addressed:

- Evaluation of the economic impacts of nitrogen pollution
- The relative importance of reduced and oxidised nitrogen pollutants
- Nitrogen and climate change
- Potential co-benefits of improved nitrogen efficiency in agriculture
- The potential for new air quality limits on ammonia to help achieve objectives of the Habitats Directive

Nitrogen and the environment

Disturbance to the global nitrogen cycle receives less attention than disturbance to the carbon cycle, but the nitrogen cycle is at least as, if not more, out of balance. A comprehensive approach to managing nitrogen pollution is a challenge for EU policymaking.

The nitrogen cycle and our influences on it are complex. Seventy eight per cent of the world's atmosphere is composed of di-nitrogen gas (N_2), which is relatively unreactive. Plants, animals and various industrial activities need nitrogen in reactive forms. We have learnt to acquire reactive nitrogen through the industrial Haber-Bosch process, which captures di-nitrogen gas and converts

it into ammonia (NH₃). However, humankind's increasing use of reactive nitrogen in fertilisers, plastics, explosives, among many other products, leads to problems as most of the nitrogen is leaked back into the environment.

Nitrogen can be emitted in various forms to air and to water, and is also deliberately applied to land as fertiliser. Once emitted, its form may change, passing back and forth through various reactions in the atmosphere, waters, soils, and life-forms.

1. Evaluation of the economic impacts of nitrogen pollution

The total annual cost to the EU of nitrogen pollution's environmental impacts has been estimated to be between €70 billion and €320 billion. This raises questions as to the cost-effectiveness of our use of reactive nitrogen. For example, costs to the EU from nitrogen pollution from fertilisers could equal or exceed the economic benefits. The robustness of such economic analyses is questionable, however, as they use some contentious methods, such as 'willingness to pay' techniques. Nonetheless, analysts suggest the economic costs of reducing nitrogen pollution should generally be offset by the economic benefits, although greater clarification of costs and benefits would prove useful in presenting a more accurate picture. Currently, the costs of nitrogen pollution are often 'externalised'; the polluters do not bear the costs, and they are passed on to the rest of society.

2. The relative importance of reduced and oxidised nitrogen pollutants

There is increasing concern that we do not distinguish clearly enough in research and policy between types of nitrogen pollution, in particular between 'reduced' and 'oxidised' nitrogen pollutants. Ammonia is a reduced form of nitrogen pollution and is now seen as particularly problematic, warranting more specific pollution abatement policies. Furthermore, more details are needed to understand the response of different ecosystems to different forms of nitrogen pollution.

3. Nitrogen and climate change

Different forms of nitrogen pollution vary in their impacts on the global climate, with some having a warming effect (e.g. nitrous oxide) released by fossil fuel combustion and after nitrogen fertilisers come into contact with soils), and some having a cooling effect (such as the aerosols formed from ammonia emissions from livestock facilities). A recent attempt to summarise a 'net nitrogen effect' on climate has suggested a slight overall cooling impact from the EU's combined nitrogen pollution. However, this study excluded many aspects of nitrogen pollution and the broader climate implications of our use of

nitrogen, such as the CO_2 emissions during the Haber-Bosch process. Future estimates of the EU's net nitrogen effect could differ markedly. Even with a net cooling effect, much can be done to reduce certain nitrogen pollutants' warming impacts and contribute to efforts to tackle climate change.

4. Potential co-benefits of improved nitrogen efficiency in agriculture

Vast quantities of reactive nitrogen are used in agriculture to generate food. However, most of it is never converted into food, but is instead wasted due to a low nitrogen use efficiency. By improving efficiency in agriculture, various co-benefits may be achieved, including economic savings from reduced input costs (such as synthetic fertilisers) and reduced externalised costs (e.g. de-polluting water costs), environmental improvements from reduced pollution, and social benefits, such as increased food security and more jobs.

The techniques available to increase efficiency of nitrogen use range from minor alterations, to industrialised agriculture to the incorporation of what may be considered more sustainable agriculture, such as agro-ecology and permaculture. The pursuit of co-benefits raises important questions, for example, on the links between policies on nitrogen pollution, agriculture, and broader sustainability for the EU.

5. The potential for new air quality limits on ammonia to help achieve objectives of the Habitats Directive

Nitrogen pollution threatens existing commitments to safeguard biodiversity. For example, 60% of sites protected by the EU Habitats Directive could exceed critical loads for nitrogen pollution. However, the impacts of nitrogen pollution on these protected areas often go unaddressed.

One proposed solution is to establish local air quality limits for Natura 2000 sites based on levels of ammonia that are compatible with favourable status. Such local air quality limits would need to operate in conjunction with appropriate emissions ceilings at national levels.

In conclusion

The EU's use of reactive nitrogen occurs on a massive scale and involves many industries and increasingly threatens ecosystem services. Serious efforts are required to reduce nitrogen pollution levels sufficiently, but there is much potential good practice available, with various possible avenues for further exploration. Current revisions to the EU's air quality framework represent a timely opportunity to initiate this progress.

Introduction

Nitrogen pollution is a significant environmental issue in Europe and around the world. The European Commission is currently reviewing key strands of its approach to air quality including the National Emission Ceilings Directive (Dir.2001/81/EC) and Ambient Air Quality Directive (Dir.2008/50/EC).

Tackling nitrogen pollution also requires trans-national cooperation beyond European borders. To this end, the Gothenburg Protocol of the Convention on Long Range Transboundary Air Pollution (CLRTAP) has also been reviewed, coordinated by the United Nations Economic Commission for Europe (UNECE). The CLRTAP involves Member States of the EU and other nations, including the USA, Canada, and the Russian Federation.

'Over the last decade, you have surely heard many views as to why you should worry about carbon and climate change. But the chances are you're not worrying about nitrogen. In fact, there is a global nitrogen threat out there, yet the world seems not to notice.'

Centre for Ecology and Hydrology (CEH), 2008

Scope of this report

This report focuses on the environmental implications of nitrogen pollution. Whilst the health implications of nitrogen pollution are also a key concern, they are not addressed in this report. The report considers particular topics identified by a panel of experts[1] as being particularly relevant at the current time.

The report considers nitrogen pollution as relevant to the EU. However, the 'transboundary' nature of much nitrogen pollution means the issue is inevitably one of wider geographical scope. It also highlights material which can usefully inform air quality policies, given the air quality focus of the policy framework under review. This involves a broader consideration of the nitrogen cycle, and the movement of nitrogen between air, water, organisms and soils (Galloway *et al.*, 2003). For instance, nitrogen initially released to the air through burning fossil fuels may then be deposited into soils and water bodies, and the nitrogen initially added to soils as fertiliser may then be released to the air.

The nitrogen challenge in a nutshell

The conundrum we face globally, and in the EU, is that a range of human activities increase the flow of reactive nitrogen into global nitrogen cycles in concerning ways. Some of these flows are accidental, such as the nitrous oxide (N_2O) released from burning fossil fuels, and some are deliberate, as when nitrogen is removed from the air in the Haber-Bosch process and used to make chemicals, fertilisers, and many other useful products. Some of our biggest industrial enterprises and activities are currently linked to nitrogen pollution: agriculture depends on nitrogen, largely in synthetic fertilisers, and transport and wider industry continue to depend largely on fossil fuels for energy, emitting reactive nitrogen as a by-product. The question is therefore: how do we continue to source enough nitrogen to meet our needs without dangerously disrupting the nitrogen cycle?

Human alterations of the nitrogen cycle

The carbon cycle usually receives more attention than the nitrogen cycle, due to the pivotal role of CO_2 in climate change. However, the nitrogen cycle is also profoundly disrupted by human activities. According to Aber (as cited by Fields (2004)), *'the nitrogen cycle has been altered more than any other basic element cycle'*. This disruption brings a range of impacts, from influencing global climate to creating oceanic dead zones.

The nitrogen and phosphorus cycles are recognised by Rockström *et al.*, (2009b) as one of nine crucial 'planetary processes' which humankind must keep within certain environmental limits to avoid destabilising conditions for life and causing *'detrimental or even catastrophic [impacts] for large parts of the world'*. They state that the nitrogen cycle is already disrupted beyond safe limits, second only to biodiversity loss as an Earth process out of balance.

Central to 'the Rockström analysis' is the identification of '*human fixation of N_2 from the atmosphere as a giant 'valve' that controls a massive flow of new reactive nitrogen*' (Rockström *et al.*, 2009b). Human conversion of unreactive atmospheric nitrogen (as di-nitrogen (N_2)) into reactive nitrogen occurs primarily through the production of ammonia (NH_3) via the Haber-Bosch process, and also through fixation by leguminous plants and emissions from the burning of fossil fuels and biomass.

Human-caused production of reactive nitrogen through these means now exceeds the entire nitrogen fixation achieved by natural processes (Rockström *et al.*, 2009a), and global reactive nitrogen production has more than doubled during the last century as a result of human activity (Sutton *et al.*, 2009).

Rockström *et al.* (2009b) suggest we need to *'contain the flow of new reactive nitrogen to 25% of its current value'*. This is 'a first guess' at an environmental limit for reactive nitrogen and they acknowledge that more research is needed to establish an authoritative figure. Nonetheless, it is a powerful indication of the scale of the challenge. The importance of meeting this challenge is vital because the nitrogen cycle influences almost all the other 'global processes' identified as

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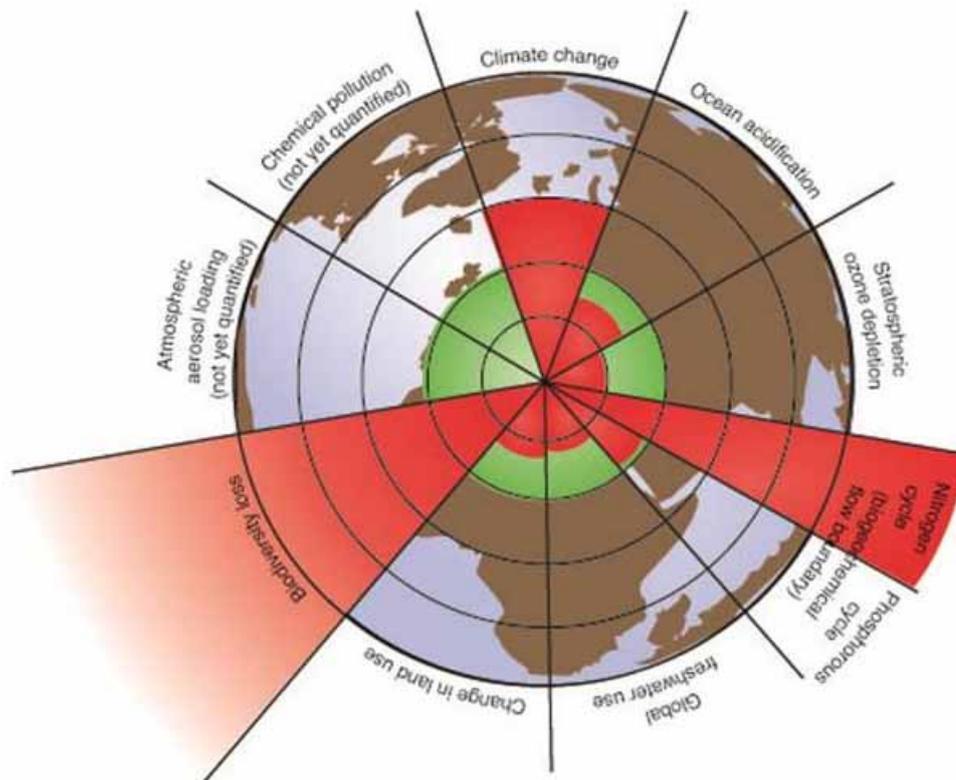1. *Convened during 2012 by the European Commission's Directorate General for the Environment.*

Figure 1. Environmental limits as defined by nine 'planetary boundaries', from Rockström et al (2009).
Note, the inner green circle represents estimated planetary safe boundaries. The nitrogen cycle is already disrupted beyond safe operating limits. Rockström *et al* (2009). consider nitrogen and phosphorus as one 'planetary boundary' owing to similarities in their impacts (such as causing eutrophication), though they discuss each element's cycle separately.

critical by Rockström *et al.*, (2009b), such as climate change and biodiversity loss (see Figure 1).

Nitrogen pollutants affect climate change, ocean acidification, stratospheric ozone depletion, biodiversity loss, water eutrophication, groundwater pollution and toxic, ground-level ozone pollution. The use of reactive nitrogen also plays a part in changes in land use. The global community must therefore address its use of reactive nitrogen if the stable environmental conditions experienced over the last 10,000 years are to continue.

Sources of nitrogen pollution

Figure 2 illustrates the rising global trend of reactive nitrogen generation since the industrial revolution to the year 2000 from the three principal

sources: Haber-Bosch industrial nitrogen fixation, nitrogen fixation by cultivation of leguminous crops and combustion of fossil fuels.

Since 2000, these data have been updated as follows (Galloway *et al.*, 2008):

- **Haber-Bosch industrial nitrogen fixation:** The largest proportion of anthropogenic reactive nitrogen is 'fixed' by the Haber-Bosch process which creates ammonia from di-nitrogen (N_2) gas for use in an array of products including agricultural fertilisers and ammunitions. The Haber-Bosch process fixes around 121 teragrams of nitrogen per year (Tg N/yr). Most of this is used in fertiliser production (70% of the EU's reactive nitrogen use (Sutton and Billen, 2010)).
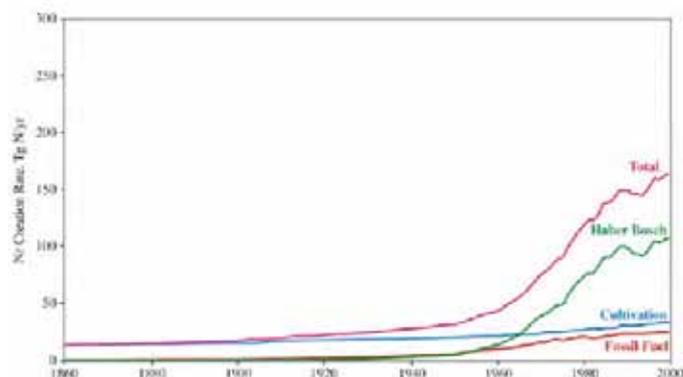
Figure 2. Global reactive nitrogen creation rate (teragrams of nitrogen per year) up to the year 2000. Source: INI, 2010

The scale and pace of change are startling and worrisome: In only two generations, humans have become the dominant influence on global nitrogen cycling, and no slowdown is in sight. New approaches are urgently needed – indeed, they are fundamental to society's struggle to achieve environmental sustainability.'

Townsend and Palm, 2009

- **Nitrogen fixation by cultivation of leguminous crops:** Leguminous crops fix around 40 Tg N/yr of nitrogen through partnering with nitrogen-fixing bacteria.

- **Combustion of fossil fuels:** The burning of fossil fuels releases around 25 Tg N/yr (in transport and energy generation) as nitrogen oxides (NOx).

Usage and emissions of nitrogen

The industrial sector with the highest nitrogen use is agriculture. In addition to the nitrogen fixed by legumes in agricultural systems, most reactive nitrogen from Haber-Bosch process is used in agriculture through the application of synthetic fertilisers to grow crops that feed people and livestock and, increasingly, biofuel crops (Galloway *et al.*, 2008). Livestock, such as pigs and cattle, release reactive nitrogen (e.g. as ammonia and nitrous oxide) through their manure, sometimes resulting in hotspots of air and water pollution. This nitrogen originates from nitrogen previously fixed by the Haber-Bosch process to create synthetic fertiliser for growing livestock feed (see Van der Voet *et al.*, 1996; Leip, 2010).

Europe is a major user and emitter of reactive nitrogen. EU-27 nations are estimated to produce 10% of global anthropogenic reactive nitrogen, even though this region covers less than 3% of the world, in terms of surface area (Sutton and Billen, 2010). Butterbach-Bahl, Nemitz and Zaehle (2011), estimate that the EU emits around 10.6% of the global nitrous oxide.

Forms of nitrogen pollution and their impacts on the environment

This report defines nitrogen pollution as nitrogen containing compounds which contribute to disruption of the nitrogen cycle and the various problems this can lead to, such as eutrophication or radiative forcing. The particular forms of nitrogen regarded as 'pollution' may occur naturally in nature, but become pollution when the quantity and/or location of their flows become problematic.

The most abundant form of nitrogen on Earth is the least reactive form: di-nitrogen gas (Sutton *et al.*, 2009). Di-nitrogen naturally constitutes 78% of our atmosphere. However, reactive nitrogen is the main cause of nitrogen pollution. Reactive nitrogen in short supply in the natural nitrogen cycle usually remains contained within the cycle. It is when reactive nitrogen is supplied in excess that it becomes lost from the cycle to the wider environment as pollution (Sutton *et al.*, 2009).

Reactive nitrogen compounds occurring as gaseous air pollutants are ammonia (NH_3), nitrogen oxides (NOx) and nitrous oxide (N_2O), and ammonium and nitrogen oxides can also contribute to air pollution when part of aerosol particles. Reactive nitrogen also occurs as water pollution in the form of nitrites (NO_2^-), nitrates (NO_3^-), and ammonium (NH_4^+) and dissolved organic nitrogen, affecting soil pH and water quality through surface run-off and leaching into surface and groundwaters. Airborne and waterborne nitrogen pollution are not completely separate, since air pollution can be deposited onto vegetation, soils and water bodies and dissolve to become water
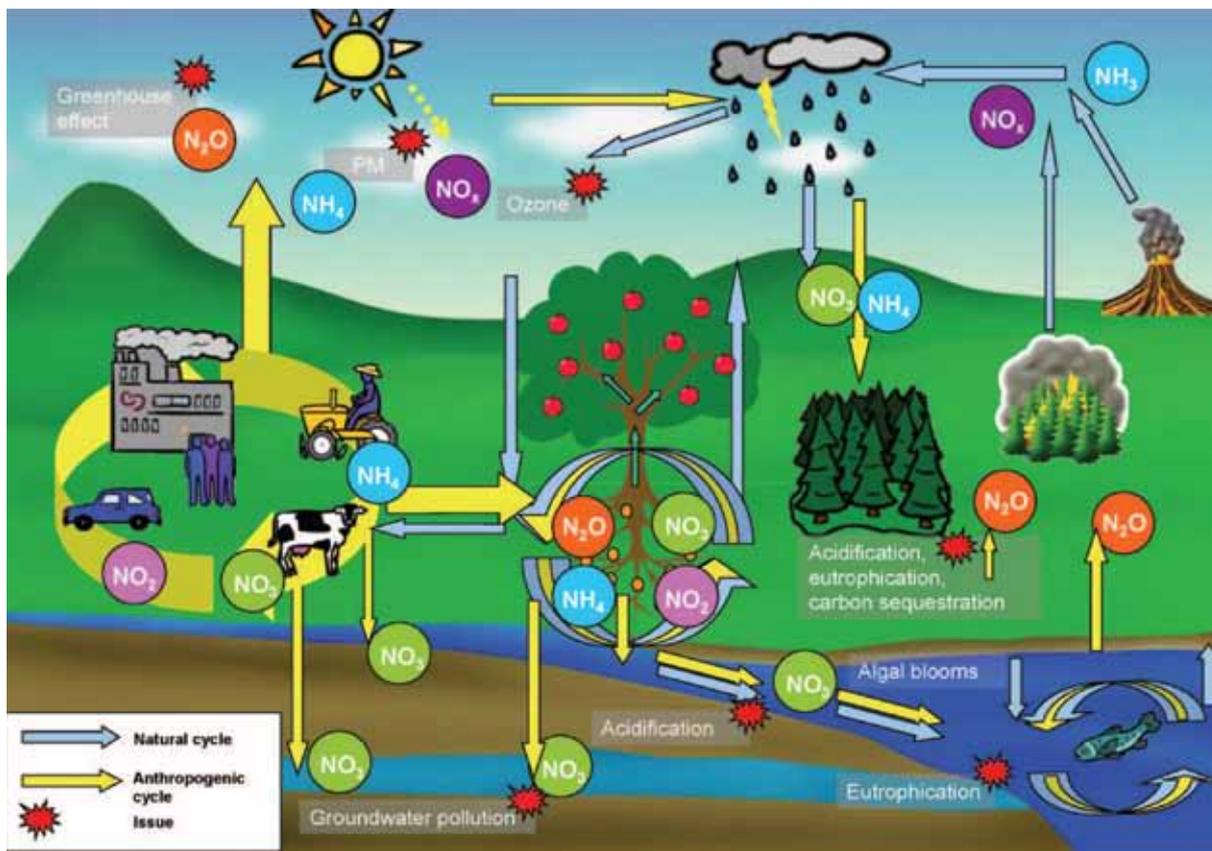

Figure 3. An overview of the nitrogen cycle and anthropogenic impacts upon it. Source: Erisman *et al.*, 2011

pollution. Likewise, waterborne nitrogen pollution may be converted into gases (see Figure 3).

Air pollutants can be deposited on land, water and plants, either through wet deposition, as when dissolved in rain or mist, or dry deposition, when the gas comes into direct contact with a receiving object. Both the reduced and the oxidised forms of nitrogen pollution can be deposited as wet or dry deposition depending on climate. For example, wet deposition dominates in Scandinavia, where rainfall is high, while dry deposition is dominant in the dryer climate of the Mediterranean (Stevens *et al.*, 2011).

According to Galloway *et al.*, (2008), in the absence of human influence, around 0.5 kg of nitrogen is generally deposited to ecosystems per hectare per year. However, for many areas of the world, the average nitrogen deposition rates exceed 10 kg nitrogen per hectare per year.

The majority of nitrogen leached from agricultural soils is in the form of nitrate and originates from synthetic fertilisers, as well as from livestock manure (Leip, 2010). Localised peaks in pollution may also be associated with run-off from urban spaces (Sutton and Billen, 2010), for instance, from poorly treated sewage and industrial sources (WRI, 2013).
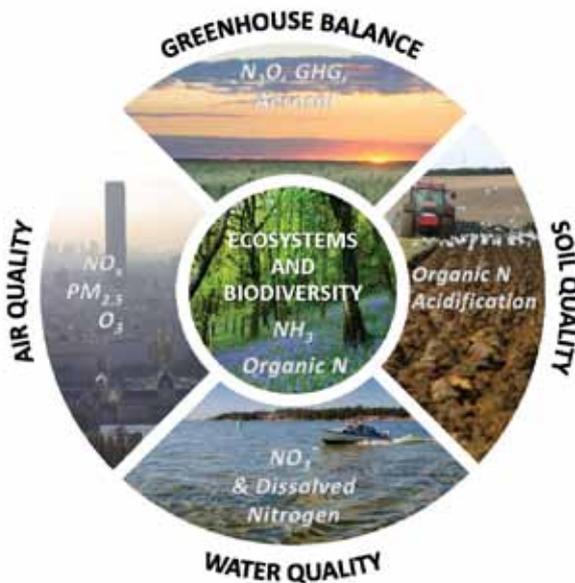
Figure 4. Threats from nitrogen pollution.   Source: Sutton and Billen, 2010

'Excess reactive nitrogen represents a major environmental threat that is only now beginning to be fully appreciated. At a global level, humans have more than doubled the production and cycling of reactive nitrogen, leading to a plethora of impacts that interact across all global spheres: atmosphere, biosphere, hydrosphere and geosphere.'

Sutton *et al.*, 2009

1. The economic cost of the ecological impacts of nitrogen pollution

Environmental impacts

The various forms of reactive nitrogen pollution affect all stages of the nitrogen cycle. A single atom of nitrogen may move around the cycle taking on different forms and creating various environmental impacts, a process referred to as the 'nitrogen cascade'. For example, a single molecule of reactive nitrogen may undergo many forms, including ammonia, nitrogen oxide, nitric acid, nitrate and organic nitrogen, and may therefore lead to a number of environmental, health and social impacts (IGBP, 2009).

Introduction

Determining the economic cost of the impacts of reactive nitrogen pollution on Europe's ecosystems is a complex task, as it is not something that is accounted for on either company or national balance sheets. However, initial estimates provide us with a starting point.

1.1 Total costs of European nitrogen pollution

The first attempt to determine the cost of European nitrogen pollution was presented by Brink *et al.*, (2011) in the European Nitrogen Assessment (ENA). The study considered the impacts on ecosystems, climate and human health of air-borne emissions of nitrogen oxides (NOx), ammonia (NH_3) and nitrous oxide (N_2O), and water-borne emissions of nitrate (NO_3^-) and ammonium (NH_4). The report concluded, based on data from the year 2000, that:

- Combined costs to the European economy of ecosystem, climate and health impacts amount to between €70 billion and €320 billion.

- Per person, this equates to a 'welfare loss' of between €150 and €740 each year, amounting to 0.8-3.9% of the average disposable income per capita. This represents 1-4% of the average citizen's income.

- Costs due to nitrogen pollution represent more than double the extra income gained from the use of synthetic nitrogen fertilisers in European agriculture (one of the main sources of reactive nitrogen pollution).

These costs represent the combined environmental and socio-economic costs, rather than costs linked to the pollution of ecosystems. About 60% of the damage costs calculated by the ENA were estimated to be related to human health, 35% to ecosystem health and 5% to the effects on the greenhouse gas balance. The ENA estimates costs to ecosystems of between €25 billion and €115 billion per year. Figure 5 illustrates the distribution of these costs; the wide range of estimates highlights the difficulty in making this evaluation.

1.2 Costs and benefits of polluting activities

The high costs of nitrogen pollution, as estimated by the ENA, may cancel out the economic gains of nitrogen-releasing activities. For instance, Blottnitz *et al.* (2006) estimated the externalised costs (costs not borne by the polluter, but by society) of nitrogen fertiliser use

Table 1: Estimates of overall social damage costs in the EU-27 as a result of environmental reactive nitrogen emissions. Source: Brink *et al.* 2011

	NO_x emission to air	NH_3 emission to air	N, loss to water	N_2O emission to air	Total
Human health	35–100	5–70	0–20[a]	<5	40–190
Ecosystems	5–35	5–35	15–50[b]	—	25–115
Climate	—	—	—	5–10	5–10
Total	40–135	10–105	15–70	5–15	70–320

[a] The value for health effects is proportionately smaller than the value for ecosystems as not all leaching is associated with health effects (e.g., denitrified during the path from soil to sea).

Table 2: Estimated cost of different reactive nitrogen threats in Europe per unit of reactive nitrogen emitted. Source: Brink *et al.* 2011

| Effect	Emitted nitrogen form	Emission/ loss to	Estimated cost € per kg N, emitted
Human health (particulate matter, NO_2 and O_3)	NO_x	Air	10–30
Ecosystems (eutrophication, biodiversity)	N, (inc. nitrate)	Water	5–20
Human health (particulate matter)	NH_3	Air	2–20
Climate (greenhouse gas)	N_2O	Air	5–15
Ecosystems (eutrophication, biodiversity)	NH_3 and NO_x	Air	2–10
Human health (drinking water)	N, (inc. nitrate)	Water	0–4
Human health (increased ultraviolet radiation from ozone depletion)	N_2O	Air	1–3
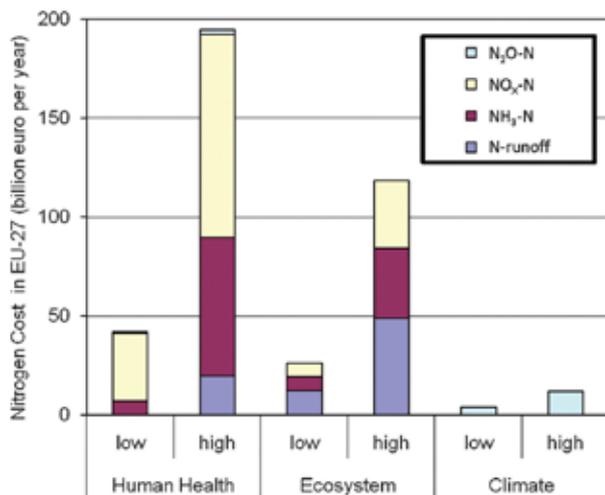
Figure 5. Distribution of costs from nitrogen pollution in Europe. Source: Brink *et al.* (2011)

by farmers (mainly through climate change impacts) as €0.3 per kg compared to the (then) market price of €0.5 per kg of fertiliser. The ENA suggested that overall the benefits of reactive nitrogen fertiliser did not outweigh the costs. The benefit to the farmer was estimated as €1-3 per kg of nitrogen fertiliser, but the total environmental costs were considered to be greater.

Sutton *et al.* (2011b) suggest that if the nitrogen pollution costs of intensive agriculture in north-western Europe were to be internalised, the economically optimal rate of annual nitrogen application would drop by 50kg per hectare: 30% lower than the private economic optimum rate for the farmer who typically externalises these costs. In Blottnitz's view (2006), farmers would face only minor revenue losses if free pollution permits were allocated by the government, but would experience more substantial losses if they had to pay a pollution tax or trade pollution permits on a market.

Considering all sources of European nitrogen pollution (not just fertilisers), and their externalised costs to society as a whole, the ENA concluded that over a diverse range of efforts to reduce air and water pollution across sectors, the financial benefits would outweigh the costs.

However, in some cases (such as winter wheat production) farm incomes and yields may be adversely affected even if overall costs and benefits to society favour measures such as lower application of synthetic fertilisers.

Some caution should be applied when interpreting such results. The ENA study is particularly reliant on 'willingness to pay' (WTP) estimations, which are much debated. WTP can be derived by asking people the extent to which they would be willing to forego money in order to avoid particular costs of pollution, or how much they would pay to fix a problem. WTP studies therefore do not necessarily demonstrate costs to the economy that are actually paid, or the costs which would occur if ecological impacts were effectively internalised. Instead, they serve as a survey of attitudes, using money as a surrogate measure of concern about pollution problems.

1.3 Improving the valuation of nitrogen pollution

One proposal for improving the scientific basis of economic evaluations is to use 'dose-response value chains' which analyse the impact or 'response' of a particular concentration or 'dose' of pollutant. An EU programme, ECLAIRE[2] , has been established to explore this in more depth for airborne nitrogen pollution. While ECLAIRE's findings are not yet available, some of the issues which it may encounter are considered here.

Dose-response relationships currently play a crucial role in European (and wider) approaches to regulating nitrogen pollution. They provide the core science which underpins the setting of critical levels and critical loads – the concentrations (or doses) of pollutants beyond which there are significant adverse impacts on ecosystems.

The dose-response value chain builds on the logic of dose-response relationships. A dose-response relationship plots the dose of a particular input (such as a form of nitrogen pollution) against its impact (i.e. a 'response'). The response of the receptor usually varies depending on the strength of the dose and this response is typically plotted on a graph to create a dose-response curve. This approach is used in fields such as ecotoxicology and epidemiology, often to describe the toxicity of a pollutant to a species.

Critical levels are set to describe the level of air pollution beyond which impacts are observed on ecosystems. Critical loads perform the same role, but for pollution deposited at ground level (e.g. from dry or wet deposition of airborne pollution). To determine critical levels and loads, the vast majority of 'receptors' in such studies have been plants, usually those considered to be sensitive to nitrogen impacts. Species diversity of plants is also commonly used as the indicator (Bobbink and Hettelingh (eds), 2011).

A dose-response 'value chain' differs from a dose-response curve by taking the curve and seeking to quantify the economic costs of the response. In the case of nitrogen pollution, dose-response value chains should be used to discern how European ecosystems are affected by different pollutants, and how these various pollutants act as part of a mixture of pollutants and other pressures on ecosystems. Immediate questions include:

- Should we test pollutants in isolation or in combination?
- Should we test their toxicity against sensitive species from ecosystems? (In which case, bacteria, fungi, plants, animals?)
- Should we look beyond individual responses and try to understand responses at population, community, or ecosystem levels?
- Should we do such tests in controlled laboratory models or out in the field, or both?
- Should we standardise our approaches across EU nations so that every country assesses the impacts of nitrogen pollutants in the same way to enable comparison, or do ecological responses to nitrogen pollution differ sufficiently to warrant different methodologies?

Such questions pose significant scientific challenges. Nonetheless, dose-response relationships have played an important role in EU policy

and there is potential for further benefits by increasing the depth and breadth of dose-response studies.

The science of dose-response relationships can be improved in various ways to provide better economic evaluation of nitrogen pollution. Several significant gaps in knowledge have been identified as follows:

- **Keeping information up-to-date.** For example, recent developments have suggested that critical levels for ammonia have been set too high, an observation resulting in their revision (Hicks *et al.*, 2011a).

- **More detail on different forms of nitrogen pollution.** Currently critical loads and levels are based on total kilograms of nitrogen deposited per hectare (Bobbink and Hettelingh (eds), 2011), which may assume that all forms of nitrogen are equal in their impacts. There are now calls for critical levels and loads to be set more specifically for a range of different nitrogen pollutants (Bobbink, Hettelingh and (eds),(2011), Hicks *et al.*, (2011a), Jarvis *et al.*, (2011)).

- **Use of critical levels and loads.** Currently critical loads are used more than critical levels, neglecting consideration of the airborne phase of nitrogen pollution. Jarvis *et al.*, (2011) argue that the critical level for ammonia may be below the critical load set for total nitrogen deposition, and highlight the importance of using both critical loads and levels.

- **Covering all ecosystems.** Different ecosystems respond to nitrogen pollutants in different ways. There are also gaps in the provision of critical loads and levels data for several types of European habitats (such as marine habitats; coastal habitats; inland surface water habitats; and mire, bog and fen habitats[3]).

- **Basing data on a wider range of receptors.** The current emphasis on plants could be considered an overly simplistic indicator of ecosystem health and could be widened, potentially to include microbial communities, animals and species diversity or abundance of vulnerable species, amongst others.

- **Analysing pollutant mixtures.** More sophisticated dose-response studies are needed to model the interactions between specific nitrogen pollutants and other pollutants or stressors, and the impacts of these, in a more systems-based manner (i.e. towards 'causal networks') (Niemeijer, 2008).

- **Moving beyond 'dose-response' where necessary.** Assessment of ecosystem dose-response is usually averaged out over time (e.g. per year) and used to inform setting of critical loads/levels. This can miss the potential fluxes of concentrations to which receptors may be exposed within this duration of time. Pearce and van der Wal (2006) suggest that unacceptable habitat change may not only be due to critical load exceedance, but also exposure to high nitrogen concentrations.

1.4 Conclusion

Initial estimates suggest the ecosystem impacts of nitrogen pollution cost the EU-27 between €25 billion and €115 billion a year, and total costs (including those from health and climate change impacts) could amount to €320 billion a year. These costs underline the importance of considering the full costs of all forms of pollution and support integrated policy measures.

The balance between costs and benefits of nitrogen polluting activities, such as fertiliser use, is called into question with research suggesting that the societal costs of fertilisers outweigh the benefits. Were farmers to pay the costs of nitrogen pollution, their use of synthetic fertiliser might fall by 30%.

Such economic evaluations could usefully be further developed. This would in turn be assisted by advances in the dose-response science which underpins the monitoring of nitrogen pollution's ecological impacts.

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3. Habitats A, B, C and D of the EU EUNIS classification.

2. The relative importance of reduced and oxidised nitrogen pollutants

Introduction

When assessing humanity's impact on the nitrogen cycle, it is important to account for the many types of nitrogen, as different forms have diverse chemical and physical properties, and disperse differently. However, most ecological studies on the environmental impacts of nitrogen have assessed 'nitrogen' pollution as a collective term (Stevens *et al.*, 2011). Similarly, the critical loads approach to assessing the impact of nitrogen pollution on ecosystems has treated all forms of reactive nitrogen deposition equally (Nordin *et al.*, 2011).

One way of categorising types of nitrogen pollutants is according to whether they are 'reduced' or 'oxidised' (see Box 1 below for definitions). Large changes in the ratio of oxidised and reduced nitrogen inputs are taking place globally and in Europe (Stevens *et al.*, 2011).

The ratio of reduced to oxidised nitrogen in the environment is significant to policy because their main sources are distinct and could therefore be subject to different policy responses. They also have different fates in the environment, and concentrations of oxidised and reduced nitrogen pollutants vary from place to place, as well as over time (Holland *et al.*, 1995, cited by Stevens *et al.*, 2011) – further reasons to consider their respective characteristics.

A higher proportion of nitrogen pollution is believed to come from reduced nitrogen than oxidised nitrogen. An increase in the proportion of reduced nitrogen pollution, referred to as an 'increased NHx:NOy ratio', has been observed in Europe. Fagerli and Aas (2008) found that oxidised nitrogen deposition decreased between 1990 and 2005 as a result of a change from coal burning to gas or nuclear power in Europe, while reduced nitrogen deposition stabilised after 1995, increasing the NHx:NOy ratio.

An increased NHx:NOy ratio is expected irrespective of whether total nitrogen pollution levels rise or fall. Galloway *et al.* (2004) foresaw an increase in both reduced and oxidised forms of nitrogen pollution, suggesting that by 2050, terrestrial oxidised nitrogen deposits will have increased by up to 70% from levels in the 1990s, while reduced nitrogen deposits will increase by 133% over the same period.

It is probably too early to predict confidently the exact effects of a change in NHx:NOy ratio; the scientific community's explorations are in their early stages. What is understood is that reduced and oxidised nitrogen forms enter plants through different means, and this varies between types of plants. The ratio change may also affect plant growth levels and competitiveness. Furthermore, different taxa of species may be affected differently (plants, animals, etc), and the impacts may cascade through entire food webs.

There are also implications of the ratio change for soil health. Soil structure may be affected by interactions between forms of nitrogen and carbon. Effects on the availability of soil toxins are also possible, for instance, where high ammonium (NH_4^+) levels acidify soils and increase levels of aluminium and iron. In addition, the availability of soil nutrients can fall as positively charged ammonium ions can bind with negatively charged soil particles allowing positively charged nutrients, such as calcium and magnesium, to be lost from the soil through leaching.

The leaching of oxidised nitrogen pollutants causes different ecosystem impacts again, such as eutrophication of aquatic ecosystems. Furthermore, forms of nitrogen can change into each other as part of the nitrogen cascade, for instance, ammonium oxidises into nitrate in soils, in a process called 'nitrification'. The acidity of soil may also influence the effects of a change in the NHx:NOy ratio (Stevens *et al.*, 2011).

Although work to date on the effects of reduced and oxidised nitrogen pollutants has mainly been limited to small scale studies of terrestrial plants (Stevens *et al.*, 2011), some key threads are emerging from these enquiries.

2.1 Ammonia as a neglected priority

The environmental impacts of oxidised nitrogen pollutants are better known than the impacts of reduced pollutants, such as ammonia, even though the instances of exceedances of nitrogen critical loads have tended to be caused by high levels of ammonia (NH_3) emissions from livestock facilities, for instance, in livestock-intensive areas of the Netherlands, Italy, Germany and Denmark (Erisman *et al.*, 2007).

Ammonia can be toxic to plants as a gas, entering into leaves irrespective of whether or not the plant uses it as a nutrient, which may be the case at low enough doses. High ammonia levels are considered to 'cause detrimental plant physiological effects, probably on the majority of species' (Nordin *et al.*, 2011). The deposition of ammonia can lead to soil acidification and leaching of soil nutrients, and also contribute

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Box 1
Reduced and Oxidised nitrogen: definitions

In simple terms, the 'reduced' forms of nitrogen pollution contain hydrogen and the oxidised forms contain oxygen.

The **reduced** nitrogen pollutants include ammonia (NH_3), which is a gas, and ammonium (NH_4^+), which is an ion existing in water. Collectively, these reduced nitrogen pollutants are referred to as **NHx**.

The **oxidised** nitrogen pollutants include the nitrogen oxides such as nitrogen dioxide (NO_2) and nitrous oxide (N_2O), nitric acid (HNO_3), and nitrate (NO_3). Collectively, the oxidised nitrogen pollutants are referred to as **NOy**.

to eutrophication and groundwater pollution. Reduced nitrogen can pass through the nitrogen cascade until it becomes oxidised pollutants (Erisman *et al.*, 2007).

These impacts of reduced nitrogen pollution warrant more particular attention than a blanket approach to 'nitrogen pollution' allows, with a critical level for ammonia required at a level 'well below' the general critical load set for total nitrogen deposition (Hicks *et al.*, 2011a).

Research suggests that the current critical load in Europe for ammonia of 8mg of ammonia per m^3 has been set too high, and should be reduced to 1mg per m^3 if impacts on the most sensitive plants (lichens and bryophytes) are to be avoided, and 3mg per m^3 where lichens and bryophytes are not under consideration (Cape *et al.*, 2009).

Recent reports consider dry-deposited reduced nitrogen (ammonia) to have greater negative effects than dry-deposited oxidised or wet-deposited nitrogen on plant communities (Centre for Ecology and Hydrology, 2009) (see also Hicks *et al.*, 2011a). Ammonia is considered more damaging to ecosystems than nitrogen oxides (NOx) and is most damaging when deposited dry rather than wet (Hicks *et al.*, 2011b). Nevertheless, oxidised nitrogen pollutants remain a problem.

2.2 Effects of a changing NHx:NOy ratio

Few studies have looked at the NHx:NOy ratio in depth, and these have mainly focused on terrestrial ecosystems (Stevens *et al.*, 2011). At present, only tentative conclusions can be drawn. The NHx:NOy ratio in soils is often different to the ratio in air, because inputs from the air are modified by soil transformations (Stevens *et al.*, 2011).

Different plants and ecosystems are likely to exhibit different responses to a given balance of reduced and oxidised pollution, and it is possible that local ecologies are specially adapted to certain ratios (Stevens *et al.*, 2011). For example, the microbiology (e.g. bacteria and fungi) of a given soil may be more adapted to a high or low NHx:NOy ratio, with fungal-dominated soils generally more acidic than bacterial soils (Lowenfels and Lewis, 2010).

Although there are exceptions to, and possible inconsistencies in the following findings, studies have suggested that:

- High concentrations of ammonium have been found to be toxic to many herbaceous plants, whereas increased concentrations of nitrate have been found to stimulate growth (van den Berg *et al.*, 2008).

- Fast growing annual plants and pioneer species are more likely to prefer nitrate, whereas slower growing perennial plants are likely to prefer ammonium (Stevens *et al.*, 2011).

Attempts to establish guiding principles like these can inform tentative conclusions about the possible effects of changes in the NHx:NOy ratio on plant communities. Examples, based on soils with distinctly different acidity (largely based on Stevens *et al.*, 2011), are:

- **In ecosystems with acid soils** (pH 4.5 or lower) there is little conversion of ammonium into nitrate (through 'nitrification') and so ammonium levels can build up to potentially toxic levels. Therefore, plants which are best adapted to acid conditions will have a distinct advantage over others. In contrast, species present on these soils which are less well adapted to the acid conditions could be most at risk. Therefore, on acid soils, deposition of ammonium could lead to species loss. Such effects have been observed in acid heathlands (Stevens *et al.*, 2011).

- **In more neutral soils** (pH 6.0 to 7.5), an increase in the NHx:NOy ratio would result in rapid conversion of ammonium into nitrate (i.e. turning the reduced form into an oxidised form). This would not only prevent ammonium toxicity from occurring, but could result in a fertilising effect on plants which prefer their nitrogen as nitrate. The deciding influence on vegetation is likely to be the degree to which species can respond to an increased level of total nitrogen (including as nitrate). Community responses could include the rapid growth of species which cast shade and exclude smaller plants (see Clark *et al.*, 2007 and Stevens *et al.*, 2011). Legume plants on these soils may also be disadvantaged, because their ability to fix their own nitrogen from the air would be less of an advantage given greater levels of soil nitrogen (Suding *et al.*, 2005).

With deposition of ammonium, nitric acid and other acidifying pollution, such as sulphur dioxide (SO_2), the ecosystems which are only 'slightly buffered' are likely to become more acidic. In such cases, *'the acid-intolerant species... might be dramatically affected by a shift from a nitrate dominated system towards an ammonium dominated ecosystem in combination with acidification'* (van den Berg *et al.*, 2008).

2.3 Conclusion

Further research is needed on the NHx:NOy ratio that goes beyond consideration of a limited range of plants and explores a wider sample of plant species, other life forms, habitat types (including aquatic), conservation sites, ecosystem services, and interactions with other pollutants.

The discussion of the relative impacts of reduced and oxidised nitrogen pollution does not seek to define an optimal 'balance' of pollution and should not detract from efforts to reduce total nitrogen pollution, with cuts in both reduced and oxidised pollutants. Nonetheless, policies and procedures should distinguish between the management of nitrogen oxides and ammonia and future research should assess the relative impacts of different nitrogen forms (Hicks *et al.*, 2011a).

Additionally, while critical loads for nitrogen deposition are currently based on total reactive nitrogen deposition, devising habitat specific critical loads for the components of deposition and NHx:NOy ratio could be beneficial in some ecosystems (Stevens *et al.*, 2011).
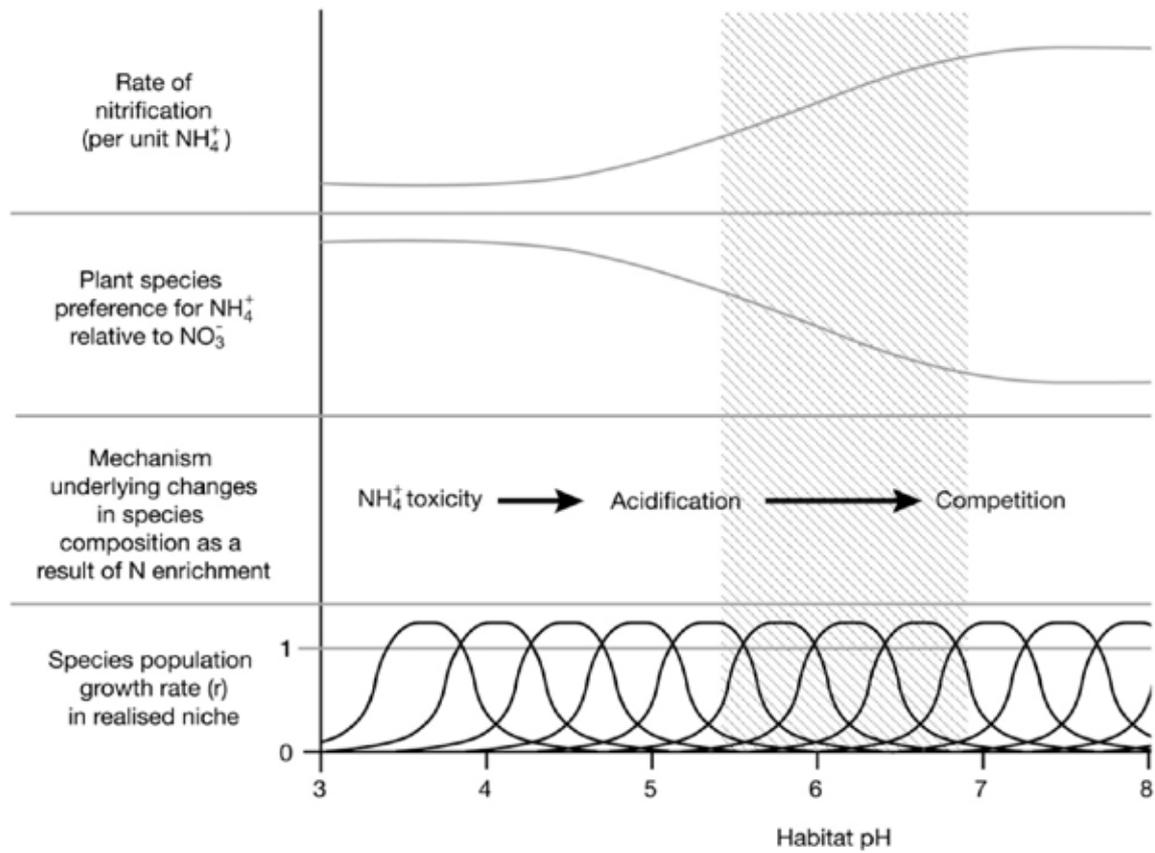
Figure 6. Hypothetical relationships between the background soil pH and the mechanisms regulating plant community composition responses to nitrogen addition. Source: Stevens *et al.*, 2011

3. Nitrogen and climate change

Introduction

The predominant form of nitrogen on our planet, di-nitrogen (N_2), is not a greenhouse gas (GHG). However, forms of reactive nitrogen pollution have various warming and cooling effects on global climate. When nitrogen and climate change are considered together, it is often with reference to the impacts on climate arising from oxidised nitrogen gases, namely nitrous oxide (N_2O) and the nitrogen oxide (NOx) gases. However, these nitrogen pollutants have different impacts on climate.

Nitrous oxide is considered a 'direct' GHG, since it traps solar radiation in the atmosphere and has a warming effect. Nitrogen oxide emissions, however, have indirect effects through their contributions to atmospheric reactions which generate ozone (O_3). Ozone has a warming effect when in the lower atmosphere (the troposphere), but a cooling effect when in the upper atmosphere (the stratosphere) (Galloway *et al.*, 2008).

This section summarises current thinking regarding the warming and cooling impacts of nitrogen pollution, focusing on emissions from the EU. The first attempt to assess Europe's overall impact on warming/cooling (i.e. the 'net nitrogen effect') was prepared by Butterbach-Bahl *et al.* (2011) as Chapter 19 of the European Nitrogen Assessment (ENA). While this assessment is preliminary, it forms the basis of much of our understanding of European emissions' contribution to climate change. Unless otherwise stated, all information in this section is taken from this source.

3.1 Warming effects of nitrogen pollution

Nitrous oxide

Nitrous oxide is released in large amounts each year and has a positive radiative forcing, i.e. its molecules store thermal radiation from the sun, thus contributing directly to the warming of the atmosphere. Per unit of weight, nitrous oxide is a more powerful GHG than CO_2. Galloway *et al.*, (2007) calculate that, over a 100-year period, nitrous oxide has a global warming potential 296 times larger than an equal mass of CO_2.

Estimates vary as to how much nitrous oxide is released by the EU each year[4]. The United Nations Framework Convention on Climate Change estimates 819 gigagrams (Gg) per year, but this figure does not include emissions from EU rivers, continental shelf and estuaries, nor indirect emissions following nitrogen deposition and leaching. These indirect emissions of nitrous oxide have themselves been estimated at 71.1 gigagrams (Gg) and 34.6 Gg per year.

Other summary totals conclude a range varying between 579 and 1438 Gg of nitrous oxide released by the EU-27 per year. If converted into a CO_2-equivalent (CO_2e), this would be 269-669 teragrams (Tg) of CO_2e. The key sources of nitrous oxide are outlined as follows:

Nitrous oxide and agriculture

The major driver for changes in atmospheric nitrous oxide concentrations is the increased use of reactive nitrogen fertiliser in agriculture, mainly through emissions from soils that have been applied with reactive nitrogen – either in the form of synthetic fertiliser

or in the form of manure. These fertilisers increase the reactive nitrogen available to *denitrifying* microbes which release gaseous nitrous oxide into the atmosphere (as well as di-nitrogen).

A total of around 10,638 Gg of nitrogen is added to soils as synthetic fertilisers each year in the EU-27, and 11,302 Gg as manures. The knock-on effects of reactive nitrogen fertilisers include the leaching of nitrates into water bodies, which in turn affects the extent to which downstream ecosystems act as a source for releasing nitrous oxide. The Intergovernmental Panel on Climate Change (IPCC, cited in the ENA) estimate that, globally, 1% of applied nitrogen is released directly as nitrous oxide and another 0.4% indirectly (later in the nitrogen cascade). In Europe, the conversion of nitrogen into nitrous oxide from synthetic fertilisers has been estimated at 3-5%.

European cattle, pigs and poultry are all considerable contributors of atmospheric nitrous oxide derived from nitrogen in their manure and urine. 177 kilotons of nitrous oxide are estimated to have been released in 2000 from animal manure in the EU-27.

To summarise nitrous oxide's impacts from agriculture, three studies cited by the ENA respectively suggest that EU agriculture emits a total of 377 Gg, or 400 Gg, or 493 Gg of nitrous oxide per year. These may be underestimates as they do not include other potential sources of nitrous oxide, such as the conversion of peatland soils to arable land in some EU nations. This is a notable omission as peatland conversion has been estimated to cause 3-10% of some EU Member States' GHG emissions (through both nitrous oxide and CO_2).

Nitrous oxide from sewage treatment and waste management

Nitrous oxide is emitted as a result of bacterial processing of nitrogen in wastewater. The amount released depends on the levels of oxygen in the process, being highest at relatively low oxygen levels but decreasing at either very low or very high oxygen levels. Annual nitrous oxide emissions from wastewater treatment have been estimated at 25.7 Gg of nitrogen, representing about 5% of total European nitrous oxide emissions.

The burning of sewage sludge also releases nitrous oxide, but available data for this sometimes includes emissions from the combustion of various other solid wastes. The combined release of nitrous oxide from waste incineration is estimated at 4.98 Gg (in the year 2000), representing 0.8% of European nitrous oxide emissions.

Nitrous oxide from fossil fuel combustion

Combustion of fossil fuels in energy production, industry and transportation, can result in releases of nitrous oxide - particularly at medium temperatures (500-600°C). Whilst combustion of fossil fuels is notorious as a key emitter of CO_2, it is considered to be only a minor source for nitrous oxide emissions. The European Environment Agency (cited by Butterbach-Bahl *et al.*, 2011) reports power stations emit quantities of nitrous oxide equivalent to 7.6 Tg of CO_2. They calculate transport combustion (of diesel and gasoline)

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4. Summarised in Table 19.1 by Butterbach-Bahl et al. *(2011)*

to release 13.4 Tg CO_2e, a small fraction of the agricultural emissions of nitrous oxide.

Large industrial facilities can also be major sources of nitrous oxide and should not be overlooked for potential mitigation measures. Key industrial emitters of nitrous oxide tend to be processes utilising nitric acid to produce particular chemicals, such as adipic acid and glyoxal. There are a very limited number of such installations in Europe, but they do cause high down-wind concentrations of nitrous oxide. Mitigation measures have become available at very low costs and installation of these has already seen sharp reductions in nitrous oxide emissions from these sources within Europe. However, production of nitric acid in Europe remains significant, resulting in 50 Gg of CO_2e emissions from nitrous oxide, representing as much as a quarter of the direct emissions of nitrous oxide from soils. Abatement is possible but more costly than for the aforementioned processes, although emissions trading increases the desirability of mitigating emissions from nitric acid production.

Overall, the emissions of nitrous oxide from combustion have been estimated at between 261 Gg and 461 Gg of nitrogen per year. Nitrous oxide can also be released during efforts to reduce emissions of nitrogen oxides (NOx). This illustrates the need to take a comprehensive overview of nitrogen pollution.

Nitrous oxide emissions from natural and semi-natural ecosystems

Ecosystems emit nitrous oxide to the atmosphere through natural processes of denitrification. However, the rate at which they do so has been modified by human activity increasing the quantity of reactive nitrogen in those ecosystems, for instance, following leaching of reactive nitrogen from agricultural systems or deposition of reactive nitrogen from air pollution.

Summary of nitrous oxide emissions' direct contribution to climate change

The warming impact of Europe's nitrous oxide emissions has likely been underestimated. Butterbach-Bahl *et al.* consider the IPCC's emission factor for nitrous oxide of 1% for indirect emissions from soils following Nr deposition to be too low and propose it should be at least doubled. The total radiative forcing of European nitrous oxide emissions is considered to be approximately $17mW/m^2$ (miliwatts per square metre).

Nitrous oxide and ozone in the troposphere

In addition to the direct warming impact of nitrous oxide on the Earth's atmosphere, the climate change impacts of nitrous oxide are exacerbated by its chemical reactions with ozone in both the troposphere and stratosphere. In the troposphere, nitrous oxide undergoes 'photolysis' in which ozone is created and acts as a GHG with a modest warming effect of $2.9 \ W/m^2$. However, this warming effect in the lower atmosphere is thought to be partly counteracted by the cooling effect that ozone has when it is in the stratosphere (see Section 3.2).

3.2 Cooling effects of nitrogen pollution

Nitrous oxide and ozone in the stratosphere

Conversely to its effects in the troposphere, the impacts of ozone in the stratosphere may have a cooling effect (Galloway *et al.*, 2008).

Whereas in the troposphere, reactive nitrogen causes increased ozone production and thus trapping of solar heat, in the stratosphere it causes destruction of ozone and reduced trapping of solar heat. In the stratosphere, ultraviolet light breaks up nitrous oxide, producing nitric oxide (NO), which in turn breaks down ozone (Fields, 2004).

Emissions of nitrous oxide are expected to continue to be the main anthropogenic emission of a (stratospheric) ozone-destroying compound for the foreseeable future (Portmann, Daniel and Ravishankara, 2011; Chipperfield, 2009). However, efforts to assess the global warming implications of reactive nitrogen in the stratosphere are said to be *'fraught with uncertainty due to the complexities of stratospheric ozone chemistry'* (Butterbach-Bahl, Nemitz and Zaehle, 2011). The ENA considers the cooling influence of Europe's nitrous oxide emissions on stratospheric ozone to be 'comparatively small'.

Aerosols containing reactive nitrogen

Aerosols are suspensions of tiny particles of solids or liquids, held in the air by gas, and are present in clouds and smog (Beresford Kroeger, 2011).

Aerosols, including those from sulphur pollution, may cool the atmosphere in two ways – reflecting sunlight and interacting with cloud formation in ways that influence temperature. Various types of aerosols formed from nitrous oxide, nitrogen oxides and ammonia are considered to have a cooling effect, particularly through scattering sunlight. The cooling impact of nitrogen aerosols has been estimated at between -27.5 and $-5.5mW/m^2$ (an average estimate being $-16.5mW/m^2$).

3.3 Further climate impacts of nitrogen: carbon sequestration and methane emissions

Reactive nitrogen, tropospheric ozone and carbon sequestration

Nitrogen contributes to the formation of tropospheric ozone. This affects the carbon cycle, as tropospheric ozone impedes photosynthesis and thus reduces carbon sequestration. Galloway (2008) raises this possibility and draws attention to uncertainties in how tropical ecosystems will respond to rising reactive nitrogen inputs, as most research has been undertaken in northern latitudes. This is an important question given that tropical ecosystems are expected to receive 'the most dramatic increases' in reactive nitrogen in the future, and are already under pressure from a warming climate.

Reactive nitrogen and methane

A significant issue for the carbon cycle is how reactive nitrogen pollution affects methane (CH_4) levels. Methane is a GHG over twenty times more powerful than CO_2 (over a 100-year period). Just as reactive nitrogen can influence levels of soil carbon and oxidation processes which release CO_2, so can it influence levels of methane released by soils. Furthermore, the use of reactive nitrogen in agriculture can also affect the methane released by livestock, cattle in particular. These issues are outlined below. The impacts of reactive nitrogen on methane may be substantial, but the evidence is not yet clear. Owing to this uncertainty, impacts on methane were not factored into the ENA's final figure for net nitrogen effect.

Methane from wetlands

It is fairly well known that wetland ecosystems emit methane, as do wetland-based agricultural systems, such as rice-paddy agriculture through 'methonogenic' bacteria, which release methane in anaerobic conditions (as found in wet soils).

Due to the natural variation of nitrogen status in soils, the impacts of adding reactive nitrogen are likely to differ between ecosystems. To illustrate this complexity, where nitrogen fertilisers are heavily applied to intensively farmed rice paddies, additional deposition of reactive nitrogen from atmospheric pollution may actually reduce the methane released by slowing down the methane-producing 'methanogenic' microbes. However, where reactive nitrogen levels are lower, the deposition of atmospheric pollution can increase methane release. Conversely, increased nitrogen can stimulate 'methanotropic' microbes, which have the opposite effect of the methane-producing microbes, by essentially feeding on methane.

Increasing availability of reactive nitrogen to plants can result in increased photosynthesis and more carbon being sent by plants to their root systems, which ultimately becomes available to soil-based methanogenic microbes that convert it to methane. However, this increase in plant productivity may dry out the soil, increasing soil oxygen levels, stimulating methanotropic microbes to oxidise more methane. The ENA concluded that whilst EU wetlands (and water bodies) emit 3.92 Tg of methane carbon each year (citing Sarrnio *et al.*, 2009), the impact of reactive nitrogen deposition on this figure is negligible (adding just 0.01 Tg of methane carbon each year).

Reactive nitrogen and its impact on methane from livestock

Livestock, such as cattle, emit methane from the methanogenic microbes that live in their digestive systems. With globally increasing demands for meat and dairy produce, methane emissions from livestock have been increasing.

Different livestock systems and animals emit different quantities of methane. The rearing of pigs, poultry and dairy cattle emit the most methane, as they require large amounts of reactive nitrogen used as fertiliser to grow feed for the animals. These intensive forms of livestock production emit more methane than the rearing of cattle and sheep for meat production. Cattle and sheep tend to graze grass, and although pastures are often fertilised, this is not as nitrogen intensive as growing food crops, such as maize and soy, and feed does not need to be transported.

The nitrogen used to grow feedstock for dairy cattle has been estimated as responsible for generating 50% of the methane released by EU cattle. By developing feed that is digested with less methane production, methane emissions per unit of food have substantially declined over past decades. However, global demand for livestock products has been resulting in increasingly intensive production methods, such as 'super dairies'. According to the ENA, the increase in animal numbers has more than offset methane savings delivered by improvements in feed quality, the genetic characteristics of livestock and management.

Reactive nitrogen and its impact on soil methane and carbon

Methane can also be released, or indeed absorbed, by methanogenic or methanotropic microbes in agricultural soils. This depends both on the type of soil and on management practices. In 'upland soils' (i.e. dryer soils than in wetlands) methanotropic bacteria have enough oxygen to feed off methane, thus making the soil a methane sink and reducing methane contributions to climate change. There is potential for this 'sink' to operate in a wide range of agricultural soils. The viability of this soil methane sink depends on agricultural practices, which have so far received little investigation. Jones (2010) suggests that agricultural research tends to focus on conventionally managed crop and pasture lands, where the breakdown of healthy soil has reduced its natural ability to act as a methane sink.

In addition to inhibiting methanotropic bacteria's efforts to break up methane, reactive nitrogen fertilisers (and other practices of intensive agriculture) can inhibit the soil's ability to convert reactive nitrogen into benign organic forms (Jones, 2010; Christopher and Lal, 2007).

3.4 Conclusion

The 'net nitrogen effect' is the overall impact of nitrogen pollution on global climate after taking account of both warming and cooling effects. After their review of the available evidence, the ENA concluded:

> *'Overall, European Nr [reactive nitrogen] emissions are estimated to have a net cooling effect, with the uncertainty bounds ranging from substantial cooling to a small net warming (–15.7 (–46.7 to +15.4) mW/m²)'*
>
> (Butterbach-Bahl, Nemitz and Zaehle, 2011)

The ENA's best estimate of a moderate net cooling impact of Europe's reactive nitrogen pollution is due to impacts on carbon sequestration and the effects of aerosols. While the ENA's conclusion points towards a net cooling effect of Europe's nitrogen pollution, this result should be interpreted with some caution and treated as an interim finding, as there are many gaps in the science and various issues were not quantified by the ENA, for example, nitrous oxide emissions from burning of sewage sludge (potentially double-counted by waste management sector data) or the release of nitrous oxide (and other GHGs) from the conversion of peatland soils to agriculture. As the ENA itself suggests, the possibility of a net warming effect should not yet be discounted given the insufficient attention given to the topic so far.

If the 'net cooling' conclusion proves correct, this should not suggest that climate change is of no relevance to nitrogen management. The conclusion of 'net cooling' masks mechanisms of reactive nitrogen pollution which do cause warming, and by reducing emissions of nitrous oxide, global heating can be reduced. It may also be possible to reduce nitrogen pollution's inhibiting effects on Earth's cooling mechanisms, such as the ability of soils to act as carbon sinks.

Butterbach-Bahl, Nemitz and Zaehle (2011) suggest that a holistic overview of the various environmental (and social) impacts of nitrogen pollution is needed for both science and policymaking:

> *'The climate-relevant effects of Nr [reactive nitrogen] compounds… comprise not only the effects of atmospheric N₂O, but also the indirect effects of Nr on atmospheric CH₄, CO₂, O₃ and particles. Effects of any policy mitigating Nr emissions will therefore have legacy effects also for these other climatically relevant compounds. Conversely, because of the multiple interactions between these compounds, future emission limitation for any of these substances also relates to the respective contribution of Nr*
>
> (Butterbach-Bahl, Nemitz and Zaehle, 2011)

4. Potential co-benefits from improved nitrogen efficiency in agriculture

Introduction

The pool of reactive nitrogen in the world's nitrogen cycle is continually increasing (Rockström *et al.*, 2009a). To work towards halting or reversing this increase, we should also address the efficiency with which reactive nitrogen is used.

'Nitrogen use efficiency' (NUE) can be defined and measured in various ways. Two specific definitions of NUE are offered by Jarvis *et al.*, 2011: 1.) the *direct* 'recovery efficiency', which is the proportion of added nitrogen as fertiliser that is successfully utilised and converted into food (either crops or livestock) and 2.) the *indirect* recovery efficiency, which is the increase in total biomass yielded divided by the amount of nitrogen applied. However, in broad terms, the objective of considering NUE in agriculture focuses on minimising damaging emissions of nitrogen from agricultural systems whilst also maximising the benefits gained.

The way NUE is defined and measured will vary according to context and can apply to sectors other than agriculture. For instance, because nitrogen pollution is emitted from fossil fuel combustion, such as in car engines and coal and gas-fired power stations, the ratio of nitrogen emitted to benefit (i.e. energy produced) can be calculated.

The Nanjing declaration (Zhu, Minami and Galloway, 2004) refers to ensuring developing nations have nitrogen abatement technology in the fossil fuel combustion processes, which would indirectly improve the general NUE of non-agricultural, as well as agricultural sectors.

4.1 The importance of nitrogen use efficiency in agriculture

It is important to consider NUE in agriculture for the following reasons:
- Agriculture is a significant source of nitrogen pollution
- There are major inefficiencies in agriculture's use of nitrogen
- Reactive nitrogen is central to agriculture and its efficient use will be of critical importance to ensuring food security in Europe and around the world

Reactive nitrogen is essential to agriculture *'to create amino acids and carbohydrates in plants to feed animals and humans'* (Erisman *et al.*, 2007). However, agriculture's usage of reactive nitrogen now makes it the dominant source of nitrogen pollution in many parts of Europe owing to high rates of artificial fertiliser use and inefficient use of manures (Leip *et al.*, cited by Brink *et al.*, 2011).

Nitrogen use is becoming less efficient in agriculture; the global NUE of cereals decreased from ~80% in 1960 to ~30% in 2000' (Erisman *et al.*, 2007). Furthermore, Europe is less efficient than other regions of the world. Erisman *et al.* (2011) calculate that nitrogen efficiency (which they define as the net output of nitrogen in products, divided by the total inputs of nitrogen) in agriculture in Europe in the year 2000 was 36%. This is lower than the world average of 50%, as Europe's use of fertiliser is higher than the global average.

Not only are large quantities of reactive nitrogen used in Europe, but much of it is wasted. In many regions of Europe, reactive nitrogen is added to soils faster than it is removed in crop and animal products (Jarvis *et al.*, 2011). Agriculture can be visualised as a 'leaky pipe' which loses reactive nitrogen through many holes (Oenema *et al.*, 2009).

If the scope of NUE in agriculture is widened to consider all the nitrogen implications throughout global food supply chains (including reactive nitrogen from fossil fuel combustion in the transportation of food products), the NUE is lower still. According to Sutton *et al.*, (2009):

> *'The global food chain has a mean nitrogen use efficiency of 14% for plant products and 4% for animal products (meat, dairy, egg). The remainder is dissipated into the environment, mainly in the forms of NH_3, N_2O and N_2 to air, and in the forms of NO_3^- and NH_4^+ to groundwater and surface waters.'*

The importance of reactive nitrogen to food security

It is said that reactive nitrogen in synthetic fertilisers feeds 40% of the world's population (Jackson, cited by Sutton and Billen, 2010). With a rising global population, agricultural demand for reactive nitrogen will increase, be it in synthetic or organic forms. A doubling of global fertiliser use has been predicted for the 21st century, partly driven by increasing demand for biofuels (Centre for Ecology and Hydrology, 2008). Such predictions assume the price of nitrogen fertilisers, linked to the price of fossil fuels involved in the Haber-Bosch process, will not be an impediment to this increase in consumption. However, the rising global population increases demand and potentially competition for energy and fertiliser, thus the efficiency of nitrogen fertiliser needs to be greatly increased to avoid compromising global and European food security.

4.2 Potential co-benefits of improved NUE in agriculture

Environmental co-benefits of improving NUE in agriculture

Reducing agriculture's 'leakage pathways' can be expected to have many environmental benefits, a range of which are presented here.

Less nitrogen pollution

Improved NUE can result in reduced environmental problems from nitrogen pollution:

- **Nitrous oxide (N_2O) emissions** are released to the atmosphere following the deposition of manure and urine from livestock onto the soil. More research is needed to better quantify nitrous oxide emissions from livestock production systems and to identify a range of options for reducing emissions (Galloway *et al.*, 2008).

- **Ammonia (NH_3)** is released by various agricultural practices from soil tillage to the application of synthetic and organic fertilisers. Pig farms are particularly associated with ammonia hotspots. Ammonia pollution contributes to nitrification of ecosystems and, at high enough concentrations, can be toxic to plants.

- **Nitrate (NO$_3^-$) and ammonium (NH$_4^+$) leachate** from agricultural areas can cause eutrophication and damage to aquatic life. Improving the efficiency of plant uptake of these nutrients could help reduce total fertiliser use and nitrogen loss to ecosystems.

Water efficiency

- **Improved water-use efficiency** can go hand in hand with improving NUE. For instance, irrigation can flush unutilised nitrates and ammonium out of agricultural soils and into rivers (Jarvis *et al.*, 2011).

Biodiversity

- **Increased NUE could lead to increases in biodiversity** through reductions in nitrogen pollution and eutrophication, increases in on-farm biodiversity both above and below ground (soil biodiversity), and through improved water efficiency (see above) since many farmers extract irrigation waters from rivers and thus affect aquatic ecology.

Climate change

- **Nitrogen dioxide (NO$_2$)** is a greenhouse gas (GHG) produced when livestock manure and urine come into contact with soil surfaces, and can also be a product of applying fertilisers to soils. Increasing the retention of nitrogen into the soil, crops, and livestock, improves NUE and reduces NO$_2$ emissions

- **Methane (CH$_4$) from livestock** is emitted to air as a waste gas from ruminant digestion. Alterations to the feedstock and genetics of cattle can reduce both the amount of reactive nitrogen excreted and the amount of methane (Hindrichsen *et al.*, cited by Butterbach-Bahl, Nemitz and Zaehle, 2011). While methane emissions per unit of food have substantially declined over recent years, increased global consumption of meat means that overall levels of methane emissions from livestock have continued to rise (Butterbach-Bahl, Nemitz and Zaehle, 2011).

- **Methane emissions from soils** can be increased or decreased through different agricultural practices. Improved usage of fertilisers could improve the status of soils as potential methane sinks.

- **Fewer Haber-Bosch emissions.** Improving NUE should reduce demand (per unit of food produced) for synthetic fertilisers fixed by the Haber-Bosch process, which is currently responsible for around 2% of global energy consumption (Sutton *et al.*, 2011a).

- **Production of biogas.** Manures (and other agricultural by-products) not used by agriculture due to improved NUE can be used to create biogas for energy generation, thus converting methane into less harmful CO$_2$ emissions while reducing the need to burn fossil fuels.

- **Improved sequestration of carbon.** Improved nitrogen management can contribute to improved sequestration of carbon in both soils and plants. For instance, 'holistic management' (Mackintosh, 2010) of ruminant livestock is considered to improve the carbon soil sink.

Economic benefits of improving NUE in agriculture

- **Reduced input costs**, particularly from reducing expenditure on bought-in synthetic fertiliser. Whilst synthetic fertiliser is often considered a cheap way to boost yields, and is thus economically attractive to farmers, it is nonetheless a cost and one that is likely to rise as reduced availability of fossil fuels affects the price of fertilisers (Sutton *et al.*, 2011a).

- **Reduced output costs**. Instead of treating organic by-products, such as animal manures and plant residues, as wastes, farmers may look at NUE practices which recycle these nutrients onsite, further minimising input costs, as well as potentially reducing the time and financial costs of waste disposal.

- **Improved yields** may result from improving the efficiency of reactive nitrogen is uptake by crops and livestock.

- **Improved quality** may result from improving NUE (Brink *et al.*, 2011), and thus potentially improve the value of a farmer's produce.

- **Improved sustainability of yields.** If NUE gains are pursued in ways that move agriculture towards increasing sustainability, farmers and society stand to gain from future-proofing their businesses.

- **Improved ecosystem services and reduced externalised costs.** NUE measures could contribute to improved ecosystem services which build natural capital. For example, healthier soils retain water and reduce societal costs of flooding. Externalised costs, such as those for de-polluting water, would also be reduced.

Social benefits of improved NUE in agriculture

- **Reduced health problem**s from nitrogen pollution

- **Improved nutritional content of food**, thanks to better soil nutrient balance (e.g. of micronutrients, as well as nitrogen, phosphorus and potassium). This would have knock-on effects for health and wellbeing.

- **Improved long-term food security,** if NUE measures help to improve the sustainability of agricultural practices.

- **Local economies** could potentially benefit, and not just through farmers reducing costs of N inputs per benefit achieved. If NUE efforts were to include reducing NOx emissions from agriculture's fossil fuel use (and the transportation of food and supplies), more localised food systems might result. Combined with alterations in farming practices, such as the return of more mixed farming and greater use of manures, a more diverse range of agricultural jobs and local economic interactions might be created, though this requires further exploration.

4.3 Methods of improving nitrogen use efficiency

NUE improvements and reductions in nitrogen pollution from European agriculture will clearly require changes in agricultural practices. Only modest reductions in reactive nitrogen pollution have been observed over the past 20 years (Sutton, 2010), but improved

NUE could provide a 'win-win scenario' as many studies have shown that it could be both environmentally and financially beneficial (Sutton *et al.*, 2009). According to Sutton and Billen (2010):

> *'The basis of good nitrogen management in agriculture is to increase the temporal and spatial coincidence between nitrogen availability in the soil and nitrogen uptake by crop, thus increasing the nitrogen use efficiency (ratio of nitrogen produced in final agricultural goods to nitrogen introduced as fertilizer) and minimizing nitrogen flows into water and the atmosphere.'*

This assumes that nitrogen efficiency necessarily relates to reactive nitrogen introduced as fertiliser. The use of reactive nitrogen supplied by legume crops, nitrogen fixing bacteria, organic nitrogen (stored in the tissues of soil organisms), deposition of background nitrogen pollution and natural deposition of reactive nitrogen caused by lightning and delivered by rain should all also be considered.

Technically, improving NUE is not the same endeavour as reducing reactive nitrogen emissions from agriculture, although there are clear overlaps. For instance, it is usually the case that improving NUE can reduce both direct nitrous oxide emissions from soils as well as indirect nitrous oxide releases (Brink *et al.*, 2011). We cannot always, however, assume improved NUE equates to less nitrogen pollution.

In a contribution to the European Nitrogen Assessment (ENA), Jarvis (2010) illustrates that high NUE and low pollution cannot be assumed to be co-benefits. Jarvis reviews research in northwest Europe showing that beef and (organic) dairy farms emit less nitrogen per unit area than do more intensively managed pig and dairy farms.

However, despite their higher nitrogen losses per unit area, and being notorious for ammonia emissions, pig farms were shown to be more efficient in converting nitrogen inputs into the end product (pork) than beef, dairy and organic dairy farms.

One of the reasons cited is the lower efficiency of the cow digestive system as compared to that of pigs. Pig farms generally use feed that is considerably more nitrogen-rich than that used on cattle farms, and pigs are relatively efficient in converting these nitrogen inputs into protein. However, the high total flows still result in nitrogen rich excreta.

Jarvis's chapter in the ENA points out that a range of nitrogen fluxes have been found in different studies on different types of farm. We might therefore look to identify the most nitrogen efficient and least polluting approaches in each type of farming.

Further, the 'nitrogen flows' in a wider variety of farming practices remain to be explored, since Jarvis's review considered studies of the most common specialised types of farms (arable, pig, beef, dairy, and organic dairy) in only the northwest of Europe, omitting a wide variety of practices across Europe, including mixed farming.

The core question of fertilisers

Fertilisers are a key aspect of NUE in agriculture, owing to the volume of their application as well as the inefficiency of uptake by crops, variously estimated at 'less than 50%' (Oenema *et al.*, 2009) to '5-15%' (Erisman *et al.*, 2007).

Synthetic and organic nitrogen fertilisers

It is commonly believed that farmers need to apply reactive nitrogen fertilisers to help make up for the gap in nitrogen supply by the soil and requirements of nitrogen-hungry commercially-viable crops.

Synthetic nitrogen fertilisers, or 'mineral fertilisers', refer to fertilisers containing nitrogen fixed during the industrial Haber-Bosch process. In contrast, organic fertilisers are derived from biological sources, manures, seaweeds, and green manures (such as legumes). Other sources of nitrogen include natural processes of capturing atmospheric nitrogen (e.g. as fixed by lightning and then deposited in rain, or as fixed by soil microbes and held in the soil as 'organic nitrogen'), as well as inputs of reactive nitrogen pollution, such as wet or dry deposition from the atmosphere.

In the EU, each year there is a broadly comparable amount of reactive nitrogen generated in synthetic fertilisers and by livestock. Jarvis *et al.*, (2011) calculate that the total amount of nitrogen excreted by livestock in the EU-27 peaked at about 11 teragrams (Tg) in the late 1980s, which was very similar to the 12 Tg used as fertiliser (Oenema *et al.* 2007). This highlights the importance of considering both mineral fertiliser and manure nitrogen inputs.

However, much manure is never used as fertiliser, and is disposed of as waste. On a global level, livestock excrete about 100 Tg of reactive nitrogen per year, but just 20-40% of this is recovered and applied to crops (Butterbach-Bahl, Nemitz and Zaehle, 2011). It has been estimated that total annual nitrogen input to the world's cropland is 169 Tg. Inorganic nitrogen fertiliser, biological nitrogen fixation from legumes and other nitrogen-fixing organisms, atmospheric deposition, animal manures, and crop residues account for 46%, 20%, 12%, 11%, and 7%, of this total, respectively (Cassman, Dobermann and Walters, 2002).

A far higher proportion of reactive nitrogen is therefore added as synthetic mineral fertiliser (46%) compared to manures (11%), as well as the notable contributions of biologically-fixed nitrogen (20%) and atmospheric deposition (7%).

Problems with synthetic fertilisers

Vast amounts of synthetic fertiliser are applied to crop production each year. Synthetic nitrogen application has increased globally from 12 Tg to 104 Tg (per year) over approximately the last 40 years (Mulvaney, 2009). Far more of this is applied to soils than is taken up by plants. For example, worldwide NUE for cereal production (wheat, corn, rice, barley, millet, oats, rye) is only 33% (Raun and Johnson, 1998). Much of the remainder is lost through gaseous emissions and leaching into water courses, with consequences ranging from eutrophication to global warming.

The application of synthetic fertiliser is generally considered to be economically worthwhile to ensure higher yields (Mulvaney, 2009). However, this notion has been challenged. For example, synthetic fertilisers have been criticised (see, for instance, Jones, (2010) and Tomlinson (2010)) for undermining the on-going viability of crop production through damage to soil structure, biodiversity and nutrient status.

Furthermore, the production of synthetic fertilisers is a major use of fossil fuel energy and contributor to climate change, estimated at around

1.2% of global CO_2 equivalent emissions (Wood and Cowie, 2004). This is relevant to NUE as firstly, the combustion of fossil fuels to fix nitrogen actually results in emissions of reactive nitrogen, and secondly, because a more holistic definition of fertiliser NUE could potentially include non-nitrogen issues such as the carbon footprint per unit benefit of nitrogen fertiliser use.

Problems with organic fertilisers

Manure

Research has identified various issues with the use of organic fertilisers, such as manures. As they are currently produced and used, they do not resolve the problem of low NUE.

Firstly, the ratio of reactive nitrogen derived from manures taken up by plants can be lower than for synthetic fertilisers. Manures are more variable in their contents and impacts on crop yields are less predictable then for synthetic fertilisers (Jarvis *et al.*, 2011).

Secondly, the storage and handling of manures can result in notable emissions of reactive nitrogen in gaseous form (particularly ammonia created from excreted urea, but also nitric oxide, nitrous oxide and di-nitrogen) and also in leachate (ammonium and nitrate) (Jarvis *et al.*, 2011). 30% of the nitrogen excreted in European animal housing systems is lost during storage and 18% immediately after application to land. Thus, in total, 48% of the nitrogen excreted in animal housing is lost during storage and immediately after application.(Oenema *et al.*, 2007, cited by Jarvis, 2011).

Thirdly, whereas once manures were an integral component of the mixed farm's nitrogen economy and were kept on the farm to help maintain soil nutrient levels, over recent decades manures have become increasingly regarded as a waste product and are often exported from the site and even burnt.

The subsequent loss of reactive nitrogen is compensated for with synthetic fertilisers (Sutton and Billen, 2010; Jarvis *et al.*, 2011). This is exacerbated by the increasing separation, often to different geographical regions, of crop and livestock farming, which makes it less easy to apply manure as fertiliser to crops. High livestock densities in many areas can mean that manure has a negative economic value, owing to the costs of treating it as a waste (Brink *et al.*, 2011). Export and disposal of manure also contributes to GHGs from transport and incineration.

These problems have led to calls to prioritise endeavours to increase the NUE of manure and to ensure manures are reused on the land (Jarvis *et al.*, 2011).

Other organic nitrogen inputs

Rockström *et al.* (2009b) called for a reduction in the capture of atmospheric nitrogen (more than half of which comes from reactive nitrogen from the Haber-Bosch process, along with additional capture by legume fixation, fossil fuel consumption and biomass burning) to a quarter of its current volume (amounting to approximately 35 megatonnes (MT) of nitrogen per year[5]). They suggest a reduction of a

quarter would bring more efficient and less polluting food production. Alternatives to synthetic fertiliser include biological products, such as worm leachate (vermiliquid), compost extract, seaweed extract and fish emulsion, applied as a seed dressing or leaf spray. These can improve soil and plant health (Jones, 2010).

Human sewage is another potential form of fertiliser (Rockström *et al.*, 2009b). The reuse of human effluent would recycle many nutrients and prevent much reactive nitrogen in urine and faeces from being lost to rivers and oceans (Jenkins, 2005). However, encouraging farmer and public acceptance of this form of fertiliser presents a challenge due to cultural reasons and perceptions of hygiene risks – which depend on the way in which 'humanure' is processed (Jenkins, 2005).

As yet, the NUE efficiency of these various potential alternative fertilisers is not widely researched. The use of legume plants is perhaps the most widely discussed. Many leguminous plants have relationships with nitrogen fixing bacteria which can take di-nitrogen from the air (e.g. in soil pore spaces) and convert it into organic nitrogen (such as amino acids) making it available to the legume plant and the surrounding soil. This 'cultivation-induced biological nitrogen fixation' (C-BNF) is estimated to fix around 40 Tg of nitrogen per year (in 2005) (Galloway *et al.*, 2008).

Ways to improve NUE in agriculture

A large number of potential co-benefits from improving NUE in agriculture have been recognised, but questions remain on how they are to be achieved, and the relative merits of alterations to mainstream (industrialised) agriculture compared with more radical overhauls of entire agricultural systems.

Less mainstream forms of agriculture, from organic farming to agro-ecology, conservation agriculture, restoration agriculture and permaculture, increasingly emphasise the role of microbes in providing crop nutrition and maintaining healthy soils, and the value of soil amendments to balance carbon and nitrogen for optimal soil composition (including microbial life). Although these topics are gradually being picked up by peer-reviewed research, they have not been extensively studied by science and there have been calls for more investigation into such techniques (Tomlinson, 2010).

The ENA (Jarvis *et al.*, 2011) observes that in comparison to natural ecosystems, agricultural systems perform very poorly in terms of NUE. This points towards possible new designs for agricultural systems that resemble natural systems more closely. This notion of modelling agriculture on natural systems has been called a 'biomimicry' approach (Benyus, 1997) and is at the heart of permaculture (originally called 'permanent agriculture' (Mollison and Holmgren, 1978)), natural farming (Fukuoka, 1985; de Schutter, 2010) and agro-ecology (Francis *et al.*, 2003).

The ENA discusses various practices of more sustainable agriculture which could reduce nitrogen pollution whilst increasing NUE, such as minimal tillage, intercropping, cover crops, catch crops, green manures (including legumes), animal manures, broad crop rotation, effective use of crop residues, and landscape planning Jarvis *et al.*, 2011; Sutton

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5. Rockström et al. *suggest that such a flow of 35 MT of N, essentially captured from the atmosphere as a result of human activities, would be equivalent to a quarter of the amount of atmospheric N currently fixed by terrestrial ecosystems each year.*

and Billen, 2010). Theories and techniques from 'alternative' agriculture (such as permaculture, natural farming and agro-ecology) which may warrant investigation include:

- **Fertility measures**, such as building biological fertility (e.g. storing nitrogen in microbial life) and diverse nutrient density through mulching, compost teas, and composting of human and animal manures. Tomlinson (2010) also suggests that more should be done to assess and experiment with the NUE of legume-based agricultural systems

- **Soil management** approaches such as no-till systems which better preserve fungi, bacteria and other microbes which may assist in retaining soil nitrogen.

- **Crop systems**, such as bi-cropping, polycultures, use of perennial plants (rather than reliance on annuals) and associated techniques of agroforestry, alley-cropping and food-forests

- **Integration of animals**, through various animal-crop systems, 'holistic management' of pastures, etc.

- **Water management** (one of the variables which may influence NUE), such as sustainable capture (e.g. through dams, swales, soil organic matter) and usage (e.g. drip irrigation)

- **Design of farming systems**, through application of agro-ecological, permaculture, and other principles for landscape design

Although an array of such agricultural methods could be explored, most proposals tend to focus on improving agriculture's usage of synthetic and organic manures. Currently, the Nitrates Directive (91/676/EEC) aims to reduce water pollution by nitrates from agricultural sources and to prevent further such pollution. Water bodies suffering from eutrophication or high levels of nitrates are identified and draining land is designated as Nitrate Vulnerable Zones, in which farmers are obliged to implement an Action Programme.

Potential for improving NUE

Galloway *et al.* (2008) identified four measures they estimated would reduce global emissions of reactive nitrogen to the environment by around 53 Tg of reactive nitrogen per year (a reduction of around a third). Of these, they suggested that increasing nitrogen-uptake efficiency of crops would decrease reactive nitrogen creation by about 15 Tg of nitrogen per year, and improved animal management strategies would decrease reactive nitrogen creation by about 15 Tg of nitrogen per year.

In one of the first attempts to quantify potential NUE gains across Europe, Oenema *et al.* (2009) considered the effects of implementing three measures throughout EU agriculture: 'balanced fertilisation', 'low-protein animal feeding' and 'ammonia emissions abatement measures'. They concluded that even without implementing their measures, business-as-usual trends would see NUE in crop production improving from 44% in 2000 to 48% in 2020, with total nitrogen losses decreasing by 10%. The implementation of these three efficiency measures would result, they calculated, in crop NUE increasing to 51-55%, with decreases in emissions of ammonia (by up to 23%), nitrous oxide (by up to 10%) and nitrogen leaching (presumably as nitrate and ammonium) (by up to 35%).

Such projections may require concerted and coordinated action across European agriculture, but they are nonetheless optimistic given the relatively conventional nature of the efficiency proposals and the encouraging estimates of their effectiveness.

4.4 Economic costs and benefits of improved NUE

Many of the potential economic benefits of NUE depend on how it is implemented. For example, Brink *et al.*, (2011) suggest that generally, there are no net costs or costs are low, because they result in a higher yield and/or less use of mineral nitrogen fertiliser.

However, costs can be incurred through implementing NUE measures. Such costs could range from capital costs of new equipment to increased costs of feedstock to negative impacts on yield (at least initially). Costs vary between agricultural systems and how those systems are designed. Examples of identified costs include:

- US$5 (€3.8) per hectare of cropland in general, or US$20 (€15.4) with the added costs of improved agronomy (i.e. agricultural practices to increase yields, such as changes in crop rotations) (Brink *et al.*, 2011)

- €10 billion per year for the EU-27 through implementing low-protein animal feeding which may restrict the use of certain types of inexpensive industrial residues and raw materials in the compound feed industry (Oenema *et al.* 2009).

Economic benefits depend on wider economic factors which may detract from highly nitrogen efficient practices. For example, the relatively low price of nitrogen fertiliser as compared to the value of

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Box 2

Improving nitrogen efficiency of fertilisers

Suggestions for improving the NUE of fertilisers include:

- Matching the dosage of fertilisers to crop requirements more accurately (Mulvaney, 2009)

- Testing soil and/or crop nitrogen levels to better assess the dose of fertiliser

- Timing the application of fertilisers to minimise loss of soluble nitrogen from leaching (from excessive rain) or volatisation of gaseous nitrogen due to excessive heat

- Spreading the fertiliser application over several instances rather than a single event, so dosage can be modified in response to weather, crop development needs, etc

- Selecting particular methods of application to minimise nitrogen leakage

- Improving storage and handling of manures to reduce gaseous and liquid emissions of nitrogen

land, labour and crops, may make application of reactive nitrogen beneficial for the farm economy up to high rates of application (Brink *et al.*, 2011). As a result of the relative cheapness of fertiliser, farmers are often tempted to use more than may be required (Jarvis *et al.*, 2011). Such economic variables change over time and in different places; in many less developed nations it is uneconomical for farmers to use synthetic fertilisers (Zundel and Kilcher, 2007).

The costs and benefits of NUE improvements clearly depend on many factors, including agricultural practices, economic variables, and the wider context of policies, subsidies and incentives. Work to explore options for greater NUE should also consider the value of affected ecosystem services, which are often excluded from accounting studies. This includes services for which a financial value may be attributed (such as for clean water not requiring chemical treatment), other financial value may be less clear for other services (such as the ability of soils to sequester carbon dioxide and methane) or cultural importance (such as recreational enjoyment of unpolluted nature) (see MEA, 2005; TEEB, 2011).

4.5 Conclusion

There is both a recognised need and potential for improving NUE in European agriculture. Large amounts of reactive nitrogen are used in agriculture but are never converted into food. Instead, they are wasted due to a low NUE. Improved efficiency in agriculture's NUE could lead to various co-benefits, including economic savings from reduced input costs (such as synthetic fertilisers), environmental improvements from reduced pollution, and social benefits, such as increased food security and more jobs.

Proposed ways of improving NUE include changing our current conventional agricultural systems or potentially the design of new systems. Many co-benefits may be achieved provided these attempts are pursued with consideration of related environmental, social and economic issues. The pursuit of co-benefits raises important questions, for example, on the links between policies on nitrogen pollution, agriculture, and broader sustainability for the EU.

5. The potential for new air quality limits on ammonia to help achieve objectives of the Habitats Directive

Introduction

Ammonia (NH_3) is widely regarded as a neglected pollutant. Whereas EU levels of sulphur dioxide (SO_2) are expected to have been reduced by 72% by 2020, compared to 2000, and nitrous oxides (N_2O) by 53%, ammonia has only declined by 7% (Sutton *et al.*, 2011a). This is of concern because the ecological impacts of ammonia, for instance through toxic effects on plants (see Section 2), are considered to be more problematic than other types of reactive nitrogen, such as nitrogen oxides (NOx).

The EU Habitats Directive is a cornerstone of EU conservation policy. It seeks to 'protect the wild plants, animals and habitats that make up our diverse natural environment' through species protection and a network of conservation sites referred to as the Natura 2000 network. This network is comprised of 26,000 protected areas covering 18% of the EU's land area[6]. The Habitats Directive aims to ensure these conservation sites are maintained or restored to 'favourable' status depending on the characteristics of a given site.

Many protected areas are vulnerable to increased nitrification, acidification, or toxic impacts of ammonia (Hicks *et al.*, 2011a) and the problem often goes unrecognised. Natura 2000 sites are not routinely assessed for the risk of nitrogen deposition effects.

> 'Nitrogen deposition represents a major threat to European biodiversity, including sensitive habitats and species listed under the Habitats Directive'
>
> Hicks *et al.,* 2011a

It has become apparent that site management alone cannot compensate for increased loads of ammonia pollution affecting protected areas (Nordin *et al.*, 2011; Stevens *et al.*, 2011). Air pollution controls are therefore needed to safeguard the favourable status of protected areas. Currently, however, there is little linkage between air quality policies and the goals of the Habitats Directive, although research indicates the latter's conservation sites cannot be safeguarded without controls on nitrogen pollution. Of the types of nitrogen deposition and their potential to affect Natura 2000 sites, Hicks *et al.*, (2011b) conclude that '*ammonia emissions present the greatest policy challenge in Europe*'. They state that 60% of Natura 2000 sites could exceed the critical load for nitrogen.

Proposals to deal with the threat from ammonia to European protected areas (as well as non-protected areas) cover a wide range of policy areas and European policy instruments. These are presented in more detail in Sutton *et al.*, (2011a) and Hicks *et al.*, (2011b). Air quality policies are the focus of this report.

6. *http://ec.europa.eu/environment/nature/index_en.htm*

5.1 Improving air quality limits to safeguard conservation sites

Protected areas are vulnerable to local as well as far-off sources of pollution and will be affected by air quality policies implemented at different spatial scales.

National limits

Limits on ammonia pollution have been set at international and national levels through the UNECE Gothenburg Protocol (1999, revised in 2012) as implemented in the EU through the National Emissions Ceiling Directive (NECD, 2001/81/EC). However, as these mechanisms were not intended to protect local conservation sites, they do not do so effectively (Sutton *et al.*, 2011a).

The 2012 revision to the Gothenburg Protocol resulted in national emission reduction obligations for ammonia, amounting to about a 6% reduction for the EU as a whole, compared to the 2005 levels, to be met by 2020. These commitments will, to some extent, contribute to improving the situation, but will not be sufficient to protect the 'favourable' status of protected areas from the damaging effects of long range nitrogen pollution, irrespective of local pollution controls. Even when national limits for ammonia are met, local critical levels can still be exceeded (Brink *et al.*, 2011).

Local limits

Ammonia is of particular concern as a localised pollutant. Although it does have transnational implications, strong local sources, such as pig and poultry facilities, can cause high exceedances of critical levels and loads, threatening the status of local protected areas. This has led to calls for air quality limits to be set at a local scale. Sutton *et al.* (2011a) propose a system based on the ambient air standards set for nitrogen oxides, sulphur dioxide, ozone (O_3) and particulate matter (PM) under the Air Quality Directive (AQD) (2008/50/EC).

The prime focus of this system is the protection of human health in urban and industrial environments, but the approach could be used to protect ecosystems from ammonia (and other pollutants). The receptor and the pollutant differ from those covered by the AQD, but the principle is the same. Under this approach, localised air quality limits for ammonia would be established based on the critical levels and loads of local ecosystems (such as Special Protected Areas). These would be monitored and when exceeded, a local air quality management plan

> 'Nitrogen deposition is compromising our ability to deliver current conservation commitments and future risks from nitrogen deposition remain high'
>
> Joint Nature Conservation Committee (JNCC), 2011

would need to be created in order to bring down levels of ammonia to below the critical level or load.

As with any air quality policy, this would need to be supported by appropriate delivery mechanisms for which various options are available. Under the Integrated Pollution Prevention and Control Directive (now encapsulated in the Industrial Emissions Directive (2010/75/EU)), industries are allocated permissible pollution levels based on local air quality conditions. Ammonia is included, only as far as big pig and poultry installations are concerned as a source. There are calls to include cattle farms (Sutton *et al.*, 2011a), which can cause local exceedances of critical levels or loads.

Such a local framework for ammonia levels would not only affect permissible emissions from existing facilities, but also the conditions under which new facilities would be approved. Local air quality limits would provide clear criteria for assessing the local suitability of facilities, and setting conditions for operation. The Integrated Pollution Prevention and Control Directive currently provides an annex of 'best available practices' to limit industrial pollution.

Various mechanisms of implementation have been explored (Hicks *et al.*, 2011b, Sutton *et al.*, 2011a) including agricultural policy areas, such as the Single Farm Payment Scheme. Sutton *et al.* (2011a) point out that under the principle of cross-compliance '*any farmer in receipt of a single farm payment should already have demonstrated that they have no adverse impact on Natura 2000 sites*'. They go on to propose that further guidance needs to be developed on: a.) general rules for avoiding impacts on Natura 2000 sites through nitrogen concentrations and deposition, and b.) specification of suitable impact assessment approaches, including cost-effective methods applicable for small farms.

The implementation of low ammonia practices overlaps strongly with the rationale for increased nitrogen use efficiency in agriculture, with many potential co-benefits as well as some costs (see Section 4).

5.2 Conclusion

It may be difficult to achieve favourable conditions at all Natura 2000 sites unless air quality limits for ammonia are set at EU, national and localised levels. There are numerous existing levers to help the implementation of ammonia limits.

Implementation issues will vary from place to place, as will the costs and benefits. Although 'economic and conservation priorities' may appear to clash in some locations (Hicks *et al.*, 2011a) it is difficult to argue with the logic that such an approach is necessary to meet Habitats Directive requirements, and it is likely that at a European scale, overall benefits (environmental and economic) would outweigh the costs (Brink *et al.*, 2011).

Nitrogen pollution in the European environment

In conclusion

For Europe to reduce its share of global nitrogen pollution to a quarter of present levels (as recommended by Rockström *et al.*, 2009b), significant efforts and a step change in ambition are required.

Current and future updates to European air quality policies provide opportunities to make significant progress. It is essential such policymaking takes a systems-thinking approach due to the complex nature of nitrogen pollution, the many environmental and social issues with which it interacts, the many sectors involved, the transnational nature of nitrogen pollution, and the potential for interventions to range from systems design (such as agricultural systems), to abatement technologies, to changes in attitudes.

Although the challenge may be great, there is already a range of good practice available to reduce nitrogen pollution which could bring a range of co-benefits.

'Single issue policies have been an effective means of reducing reactive nitrogen (N_r) emissions in the EU, but to make further reductions more integrated approaches are required.'

Brink et al., 2011

'This is not to imply that a switch to more sustainable nitrogen use will come easily. There is no silver bullet. But many of the tools, ranging from technological solutions to policy instruments, already exist. In many cases, making those solutions a reality is hindered not by the lack of an option but by a lack of political will and an understanding of the magnitude of the problems.'

Townsend and Palm, 2009

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