**Q5. Can the control of ozone precursors in Europe be regulated to maximise the benefits for health and climate effects?**

Addressing air pollution and climate change together provides a unique opportunity to simultaneously achieve both air quality and climate policy goals in the mid-term (<50 years). Air pollution regulations are an important component of strategies to protect human health and ecosystems, including crop production. However, some air pollution emission controls, such as sulphur reductions, have adverse effects on climate change mitigation efforts. Equally, measures to mitigate climate change can have adverse effects on air quality.

Therefore, simultaneous solutions that offer net benefits for both air quality and climate taking into account the possible trade-offs between human health, food and water security and ecosystems would be advantageous.

An assessment of existing emission projections of greenhouse gases and air pollutant (IPCC RCP, Global Energy Assessment, and European Commission Roadmap for moving to a low carbon economy by 2050) shows that the magnitude of the co-benefits of climate policies for air pollution varies depending on the projections. But none of the scenarios investigated yields a negative trade-off i.e. increased air pollutant emission with GHG mitigation scenarios (Colette et al. 2012). In terms of impacts, average annual $O_3$ increases in some $NO_x$-saturated urban centres in the climate projections, but again the comparison of the reference and mitigation pathways shows that $O_3$ air pollution decreases substantially when the climate policy is enforced. It is worth noting that in general, changes in surface $O_3$ due to climate change are much smaller than those from anthropogenic emission reductions over the same time period (Langner et al. 2012).

**Summary**

> Integrated assessment of air pollution and climate policy synergies are a vital part of the process.

> Ozone air pollution decreases substantially when a climate policy designed to limit global warming to 2°C by the end of the century is enforced.crop yield loss in continents downstream, in addition to effectively mitigating local ozone-induced production losses".(from MEGAPOLI Pandis et al. 2010).



# Chapter 4
# Nitrogen

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**Introduction**

The European nitrogen cycle and the effects of reactive nitrogen on climate and the environment have been examined in detail over the last five years under the lead of the NitroEurope Integrated Project, funded under the 6th Framework Programme (www.nitroeurope.eu; Sutton and Reis 2011). NitroEurope provided the central foundation to conduct the work of the European Nitrogen Assessment (Sutton et al. 2011b). Together with input from the Nitrogen in Europe (NinE) and COST 729 programmes of the European Science Foundation, the findings have provided input to support the UNECE Convention on Long-range Transboundary Air Pollution (CLRTAP), especially through the coordinating activities of its Task Force on Reactive Nitrogen (Sutton et al. 2011a,c; Reis et al. 2012; TFRN 2012). These issues are now being extended in the ÉCLAIRE, examining how climate change may alter the air pollution threat to ecosystems (ECLAIRE 2013), thereby making a contribution to addressing the emerging challenges related to nitrogen (Sutton et al. 2009c).

Many of the major outcomes, as addressed in the following sections, have been discussed in detail in the ENA, with the key aspects collated from the ENA Summary for Policy Makers. Individual chapters of the ENA, which give a more comprehensive picture, have been referred to and are accessible online.

## Q1. How do nitrogen emissions affect air quality?

Nitrogen is an extremely challenging substance for air quality: it is present in the air in many different chemical forms and it has proved hard to achieve substantial reductions in its emissions. The first point to recognize is that 78% of the world's atmosphere is nitrogen, present in the form of di-nitrogen gas ($N_2$). This is the stable state of nitrogen, which because of its unreactive nature provides a necessary stability to our atmosphere that moderates other changes. However, atmospheric $N_2$ can be converted into 'reactive nitrogen' compounds (collectively called $N_r$), forming both oxidized nitrogen compounds ($NO_y$) and reduced nitrogen compounds ($NH_x$). It is the emission of these reactive nitrogen compounds that generates the problems for European air quality.

There are many different forms of reactive nitrogen present in the environment, and it is important to see the atmospheric component as part of the wider nitrogen cycle. In particular, as Figure 4.1 illustrates, $NO_y$ emissions are dominated by high temperature combustion processes, such as in large combustion plants and vehicle engines, while $NH_x$ emissions are dominated by agricultural activities, especially those including livestock. The $NO_y$ and $NH_x$ react with each other and other atmospheric constituents to form nitrogen containing aerosol. These aerosol undergo long-range transport in the atmosphere, with the different $N_r$ forms produced eventually being removed by a combination of direct uptake by the ground ('dry deposition') and scavenging by clouds and precipitation (leading to 'wet deposition'). These $N_r$ compounds contribute to high nitrogen dioxide ($NO_2$) concentrations in urban areas, fine particulate matter (PM2.5 and PM10) and tropospheric ozone ($O_3$) concentrations, each of which pose a threat to human health (Hertel et al. 2011; Moldanova et al. 2011). In addition, with parallel emissions of nitrous oxide ($N_2O$) from soils and industry, $N_r$ poses a risk for climate change and depletion of the stratospheric ozone layer (Butterbach Bahl et al. 2011; Ravishankara et al. 2009). Exposure of ecosystems to high $N_r$ concentrations and the reaction products, such as tropospheric $O_3$, as well as their deposition leads to both phytotoxic effects and long term changes in natural and semi-natural ecosystems. These include crop losses due to enhanced $O_3$ concentrations, increases in forest productivity due to $N_r$ deposition (which may partly offset the warming effects of $N_2O$ by removing more $CO_2$ from the atmosphere), and loss of terrestrial biodiversity, as plants characteristic of natural ecosystems are out-competed by aggressive competitor species (Dise et al. 2011).

Although this represents only a short-list of the effects of $N_r$ in the atmosphere, the complexity of its many effects should already be becoming clear. Figure 1 therefore provides a helpful summary of the main $N_r$ forms and the associated environmental effects. In addition to effects on air quality and climate, it should also be noted that much $N_r$ is lost directly to surface and ground waters, creating problems of eutrophication, especially in coastal zones (Grizzetti et al. 2011). A key message is therefore that better overall management of the nitrogen cycle has the potential to improve nitrogen use efficiency (NUE) of the full system (from $N_r$ formation to ultimate products), while simultaneously reducing losses and pollution of the environment (Sutton et al. 2011c).

In the following sections, effects are considered that address both the $NO_y$ and $NH_x$ components in more detail. The atmospheric cycle of $NO_y$ is highly complex, but has received the most attention from both scientists and policy makers. Under very high temperatures in the presence of oxygen, $N_2$ is converted to nitric oxide (NO), which rapidly reacts with other atmospheric oxidants such as ozone to form nitrogen dioxide ($NO_2$), the combination being terms nitrogen oxides ($NO_x$). Subsequent transformations form both nitric and nitrous acids ($HNO_3$, $HNO_2$), particles (such as ammonium nitrates and ammonium sulphates) and a complex suite involving literally hundreds of different oxidized organic nitrogen compounds. Many technologies have been developed and increasingly adopted to reduce $NO_x$ emission, mainly focusing on catalytic and non-catalytic reduction of $NO_x$ back to $N_2$, as used in large-combustion plants and in catalytic converters on vehicles. Offset against these achievements has been a steady increase in vehicle use, so that net $NO_x$ emissions have not decreased as much as had been hoped over the last 20 years.

If most attention has been given to $NO_x$ emissions, by far the largest driving change to the European nitrogen cycle is the industrial reduction of atmospheric $N_2$ to form ammonia gas ($NH_3$), which is used to make fertilizers and many industrial products. This process fixes around three times as much atmospheric $N_2$ into $N_r$ in Europe than is achieved by oxidation to $NO_y$ compounds, accounting for 11.2 Tg $N_r$ formation per year. The fertilizer $N_r$ entering Europe's agricultural systems, together with a further 1.3 Tg from biological nitrogen fixation is cycled through plant, animal and organic matter, with a substantial fraction being lost to the environment. For air quality, the key threat is atmospheric $NH_3$ emissions from livestock farming systems and from mineral fertilizers such as urea, which amount to around 3.2 Tg $N_r$ per year across the European Union. These ammonia emissions constitute a threat to European biodiversity, dispersing fertilizing $N_r$ onto natural and semi-natural ecosystems, form particulate matter, constituting a threat to human health, and provide a vector for subsequent increased emissions of $N_2O$. In addition, although smaller, at 0.15 Tg $N_r$ per year, European agricultural and others soils are a significant source of $NO_x$ emissions (Leip et al. 2011).

For ammonia ($NH_3$) the first policy challenge has simply been to recognize the problem as an international concern. This recognition was effectively achieved in 1999 with the inclusion of $NH_3$ as a regulated pollutant under the UNECE Gothenburg Protocol, with emissions ceilings for $NH_3$ subsequently being adopted under the National Emissions Ceilings Directive. However, few reductions in $NH_3$ emissions have been achieved, with very limited further commitments under the recently revised Gothenburg Protocol (Reis et al. 2012). Although a few countries like the Netherlands and Denmark have shown the it is possible to reduce $NH_3$ emissions through active mitigation policies, while retaining profitable agriculture, most other countries in the EU have so far not followed with similar policies (Bleeker et al. 2009; Oenema et al. 2011). **As the ENA and TFRN have shown, and is discussed further below, the economic costs of damage due to $NH_3$ greatly outweigh the modest cost of abatement measures, and significant emission $NH_3$ reduction could be achieved from a technical perspective.** The barriers to change appear to include both social and political dynamics and new approaches are therefore needed that recognize and address these constraints.

One of the key constraints for better understanding of the interactions between oxidized and reduced $N_r$ forms for air quality is the linearity of relationships to particulate matter formation. The message of current studies is that to achieve a substantial reduction in PM2.5 concentrations across Europe will require significant reductions in both $NO_x$ and $NH_3$ emissions.

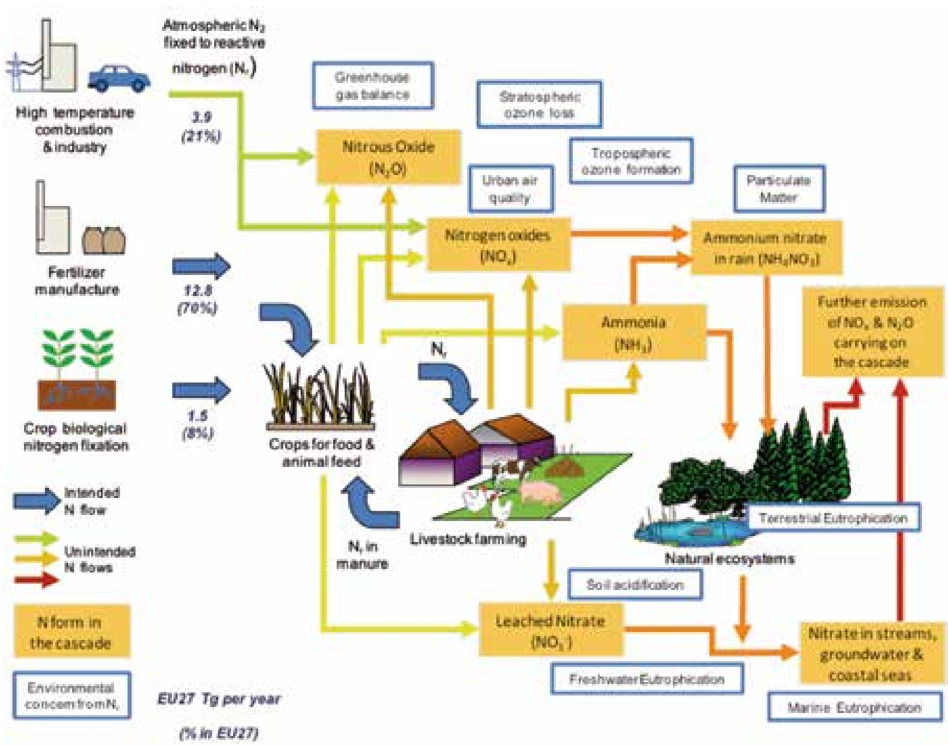


**Figure 4.1** Summary of the main intended and unintended nitrogen flows, forms and their consequences for air quality and other environmental threats. Atmospheric di-nitrogen ($N_2$) is fixed to form a suite of interacting reactive nitrogen ($N_r$) compounds, associated with a wide range of environmental concerns (based on Sutton et al. 2011b).



**Q2. How do the effects of nitrogen emissions on climate change interact with air quality?**

Emissions of $N_r$ have both warming and cooling effects on the global climate. The main warming components are increasing concentrations of nitrous oxide ($N_2O$) and tropospheric ozone ($O_3$), which are both greenhouse gases. The main cooling effects are atmospheric $N_r$, which is deposition presently increasing $CO_2$ removal from the atmosphere by forests, and the formation of $N_r$ containing aerosol, which scatter light and encourage cloud formation see Figure 5.2 (Sutton et al. 2009b; Butterbach-Bahl et al. 2011). In addition, Ravishankara et al. (2009) have identified $N_2O$ as a key contributor to stratospheric ozone depletion, reinforcing the relevance of tackling $N_2O$ not only for its contribution to radiative forcing.

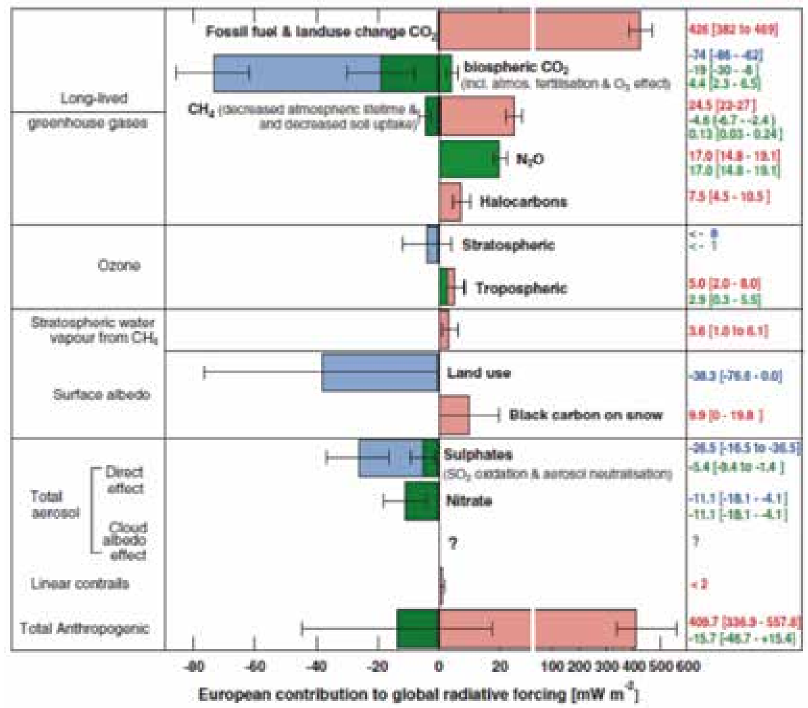


**Figure 4.2** Estimate of the change in global radiative forcing (RF) due to European anthropogenic reactive nitrogen ($N_r$) emissions to the atmosphere.
**Red bars**: positive radiative forcing (warming effects); **light green bars**: positive radiative forcing due to direct/ indirect effects of $N_r$; **blue bars**: negative radiative forcing (cooling effects); **dark green bars**: negative radiative forcing due to direct/indirect effects of $N_r$. For biospheric $CO_2$, the dark green bar represents the additional $CO_2$ sequestered by forests and grasslands due to $N_r$ deposition, while the light green bar represents the decrease in productivity due to effects of enhanced $O_3$ caused by $NO_x$ emissions. For $CH_4$ the positive (not visible) and negative contributions represent the effects of $N_r$ in reducing $CH_4$ uptakes by soil and the decreased atmospheric lifetime, respectively. Other contributions include the positive effect of tropospheric ozone from $NO_x$ and the direct and indirect cooling effects of ammonium nitrate and sulphate containing aerosol (Butterbach-Bahl et al. 2011).

European $N_r$ emissions are estimated to have a present net cooling effect on climate of −16 mW per m$^2$, with the uncertainty bounds ranging from substantial cooling to a small net warming (−47 to +15 mW per m$^2$). The largest uncertainties concern the aerosol and $N_r$ fertilization effects, and the estimation of the European contributions within the global context. It should be noted however, that the $N_r$ effect on biospheric $CO_2$ update may tend to saturate over future decades, while the longest term effect is from $N_2O$, due to its long residence time in the atmospehere. Therefore, while present $N_r$ pollution may have net cooling effects, there is an expectation of long-term commitment to net warming effects.

There are many opportunities for 'smart management', increasing the net cooling effect of $N_r$ by reducing warming effects at the same time as other threats, e.g., by linking N and C cycles to mitigate greenhouse gas emissions through improved nitrogen use efficiency. Measures that reduce emissions of $NH_3$, $NO_x$, $N_2O$ and $N_2$ and reduction of $N_r$ leaching all promote an increase in nitrogen use efficiency, such as in food and for bioenergy production. This means that less new $N_r$ inputs (from fertilizers and biological nitrogen fixation) are needed to produce the same amount of food, thereby providing potential to reduce all forms of $N_r$ pollution.

**Summary**

> Nitrogen pollution effects on climate, include warming from $N_2O$ and the N contribution to tropospheric ozone, and cooling from the N effect on biosphere $CO_2$ exchange and from $N_r$ containing aerosol

> Overall, a net cooling effect is estimated for present emissions, though the warming effect of $N_2O$ is the longest term consequence, while the cooling effect of $CO_2$ uptake may tend to saturate over future decades.

> Strategies to reduce the warming effects of $N_r$ need to reduce both $N_2O$ and $NO_x$ emissions, as well as $NH_3$ emissions, which contribute to indirect $N_2O$ sources.

> Efforts to improve nitrogen use efficiency (NUE), including improved techniques in agriculture and reduction of all forms of $N_r$ loss, allow more food to be produced with less new $N_r$ inputs.

> Improving NUE must be a central element in strategies to reduce emissions of $N_2O$, $NH_3$ and $NO_x$ simultaneously, demonstrating a clear synergy between air quality and climate policies. Better N management for air quality will therefore simultaneously deliver reductions in $N_2O$ emissions.



**Q3. Could ecosystem effects of $NH_3$ and the potential of a new AQ limit value be used in delivering Habitats Directive commitments?**

In the frame of the COST Action 729 on *Assessing and Managing nitrogen fluxes in the atmosphere-biosphere system in Europe* a workshop brought together scientists, environmental managers and policy makers to address the threat of nitrogen deposition to the Natura 2000 network (Hicks et al. 2011). It was concluded that existing legislation controlling $N_r$ emissions to air does not adequately address the impacts of nitrogen on the Natura 2000 network, and that there was need for much closer linkage between future air quality policies and the Habitats Directive.

In particular, the workshop recognized that the Habitats Directive adopts a precautionary approach for the proection of Special Areas of Conservation (SACs), which represent the flagship network of protected sites for the conservation of European natural habitats. This is illustrated by Article 6(3), of the Habitats Directive, which states that:

"Any plan or project … shall be subject to appropriate assessment of its implications for the site in view of the site's conservation objectives." … " the competent national authorities shall agree to the plan or project only after having ascertained that it will not adversely affect the integrity of the site concerned…."

There many plans and projects that may adversely affect Natura 2000 sites through by increasing atmospheric nitrogen pollution levels and deposition. In the case of large combustion plants and other point sources of $NO_x$, it was noted that a strong regulatory framework is already in place, as is the case for activities regulated under the Industrial Emissions Directive (IED). The IED includes large pig and poultry farms above certain thresholds, so to this extent a regulatory framework to protect SACs is also in place for agricultural $NH_3$ sources. However, only around 20% of European $NH_3$ emission arises from IED regulated sources. For the 80% of European $NH_3$ emissions, there is typically insufficient regulatory procedure in place, with the result that relevant plans and projects are typically not assessed in regard to their possible impact on the Natura 2000 network. This includes the development of many cattle and pig farming activities.

The workshop also noted that there was a need to develop a common approach to assessing $N_r$ deposition impacts on individual Natura 2000 sites, and on the conservation status of habitats and species has been proposed, including reliable information on stock at risk, evidence of recovery, and the restoration potential. Future approaches could build on established methods such as the critical loads assessment for the 2013 reporting round under Article 17 of the Habitats Directive and for assessments under Article 6.

The workshop evaluated possible policy options that could provide tools to meet the existing commitment to protect the Natura 2000 network more effectively

(Sutton et al. 2011d). One of the options in particular, was to consider the potential of establishing an air quality limit value for $NH_3$ over the domain of Natura 2000 sites. In this regard, it is noted that while there are AQ limit values for many pollutants (including $NO_2$, $SO_2$, $O_3$, PM , etc.), the question of setting a limit value for $NH_3$ has, until now, been neglected.

'Critical levels' for $NH_3$ have recently been revised as part of the NitroEurope activity (Cape et al. 2009; Sutton et al. 2009a) and these provide a scientific basis describing the $NH_3$ air concentration above which adverse effects on species and habitats are expected to occur according to current knowledge. In the case of higher plants the critical level was set at an annual mean of 3 (2-4) mg $m^{-3}$, while for lichens, bryophytes and habitats where these are important to ecosystem integrity, the critical level was set at an annual mean of 1 mg $m^{-3}$. The workshop concluded that these critical levels could provide the starting point for discussion on setting an air quality limit value. The values themselves have now already been adopted by the UNECE Convention on Long-Range Transboundary Air Pollution (UNECE 2007).

By setting the critical levels as annual means, the UNECE also allowed that assessment avoid the need for expensive continuous monitoring of $NH_3$ concentrations. Rather, time-integrated monitoring (e.g. monthly values), using either active or passive methods provides a low-cost approach to assessing whether the annual $NH_3$ critical level is exceeded.

It is well established that $NH_3$ concentrations are highly spatially variable, decreasing rapidly in the first 1-2 km from pollution sources before being dispersed to the wider atmosphere. This means that there is substantial potential for local measures (such as buffer zones avoiding emissions, use of woodland belts, or requirement for local mitigation techniques) to reduce $NH_3$ concentrations in the vicinity of Natura 2000 sites. Further analysis is required, but initial analyses indicate that such spatial approaches could provide a highly cost effective local complement to the National Emission Ceilings that helps Europe meet its existing commitment under the Habitats Directive to protect that Natura 2000 network.

**Summary**

- Ammonia ($NH_3$) represents a priority pollutant for future European mitigation strategies, given its substantial contribution to the exceedance of critical loads and critical levels, and the limited extent of policies so far implemented.



- Further reductions in emissions of $N_r$ emissions and $NH_3$ in particular are required to reduce effects of $N_r$ on the Nature 2000 network, which is currently estimated to be under significant risk of nitrogen deposition, leading to changes in species composition with loss of key elements of biodiversity.
- The critical levels for $NH_3$ have been revised with the support of FP6 research and are now adopted by the UNECE CLRTAP. These values are set based on annual means, and can therefore be compared with measurements using current low-cost monitoring techniques. The critical level could provide the starting point for considerations to set an AQ limit value for $NH_3$.
- The establishment of an AQ value for NH3 could provide substantial benefits for meeting existing commitments to protect the Natura 2000 network of Special Areas of Conservation, designated under the Habitats Directive. This would foster the adoption of local planning and mitigation measures related to NH3, which would provide a cost-effective complement to the emissions reduction expected from revising the National Emissions Ceilings.
- Given the parallel contribution of $NH_3$ losses to $N_2O$ emissions and particulate matter formation, setting an $NH_3$ limit value in conjunction with National Emissions Ceilings would offer a win-win-win approach with parallel benefits for habitats, climate and health.

### Q4. What is the overall economic cost of nitrogen in the EU environment?

A comprehensive analysis of the costs and benefits of nitrogen in the environment was conducted as part of the European Nitrogen Assessment (Brink et al. 2011; Sutton et al. 2011c). This allowed estimation of the social costs of adverse effects of $N_r$ in the European Environment (see figure 4.3).

The outcomes of the analysis are summarized in Figure 3, which gave a first estimate of the overall cost of European nitrogen pollution at 70 billion to 320 billion Euro per year.

Expressed as € per kg of $N_r$ emission, the highest values are associated with air pollution effects of $NO_x$ on human health (€10–€30 per kg), followed by the effects of $N_r$ loss to water on aquatic ecosystems (€5–€20 per kg) and the effects of $NH_3$ on human health through particulate matter (€2–€20 per kg). The smallest values are estimated for the effects of nitrates in drinking water on human health (€0–€4 per kg) and the effect of $N_2O$ on human health by depleting stratospheric ozone (€1–€3 per kg).

Of the overall costs, 55-60% is related to air pollution effects on human health. The total damage cost equates to €150–€750 per person, or 1–4% of the average European income and is about twice as high as the present 'Willingness to Pay' (WTP) to control global warming by carbon emissions trading.

The environmental damage related to $N_r$ effects from agriculture in the EU-27 was estimated at €20–€150 billion per year. This can be compared with a direct benefit of N-fertilizer for farmers of €10–€100 billion per year, with considerable uncertainty about long-term N-benefits for crop yield.

Apart from the uncertainties inherent in valuing the environment, including the use of WPT approaches for ecosystem services, the main uncertainties in these estimates concern the relative share of $N_r$ in PM to human health effects and of $N_r$ to freshwater eutrophication effects.

These damage costs are substantially larger than the costs of mitigating $NO_x$ and $NH_3$ pollution. In particular, a mid scenario considered in preparation for revision of the Gothenburg protocol indicated a cost of €0.6 billion per year, equivalent to 0.8 Euro/kg N abated, highlighting that agriculture is a cost-effect sector for reducing $N_r$ emissions to the environment (Wagner et al. 2011; Sutton et al. 2011c).

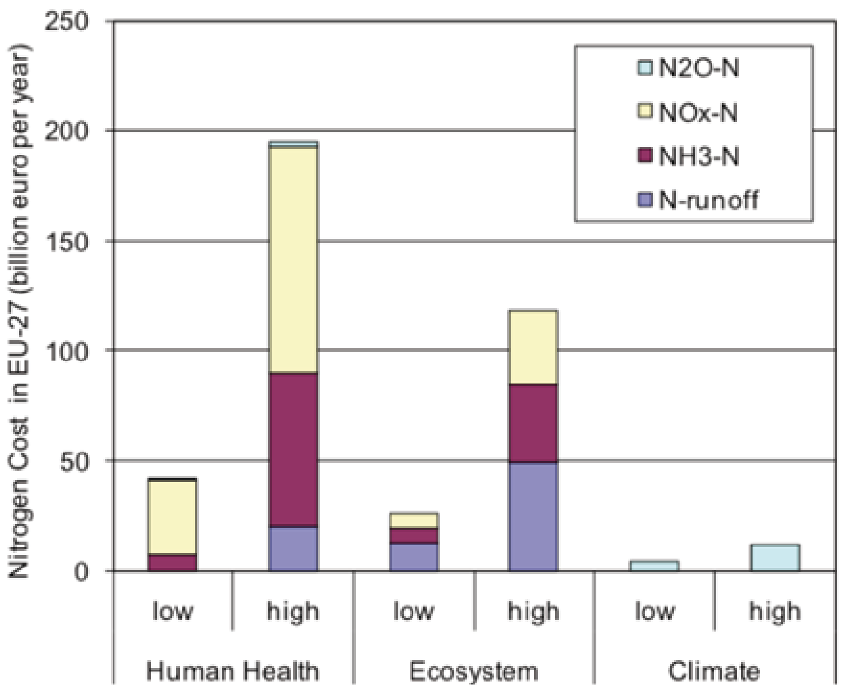


**Figure 4.3** Estimated environmental costs due to reactive nitrogen emissions to air and to water in the EU-27 (Brink et al. 2011; Sutton et al. 2011c)




**Summary**

> The overall costs of nitrogen pollution in the European Union are estimated at €70 billion to €320 billion per year, of which approximately 60% is related to air pollution effects on human health. The total damage of all reactive nitrogen losses to the environment cost equates to €150–€750 per person.

> Environmental damage related to $N_r$ effects from agriculture in the EU-27 was estimated at €20 billion to €150 billion per year.

> Effects of AQ on ecosystems, although harder to value, are estimated to be of comparable order of magnitude to human health effects.

> Revision of $NH_3$ abatement costs across the EU (mid scenario for the Gothenburg Protocol revision) indicates a cost of €0.6 billion per year, equivalent to 0.8 Euro/kg N abated, highlighting that agriculture is a cost-effect sector for reducing $N_r$ emissions to the environment compared with the social costs.


**Q5. How can we improve the valuation of air pollution threats to ecosystems?**

One of the ongoing challenges being addressed in the ÉCLAIRE project is how to improve the valuation of air pollution threats to ecosystems, especially considering dose-response relationships for nitrogen and ozone, and the interaction with future climate change.

This ongoing work recognizes that the concept of Ecosystem Services (ES) and research into the quantification of specific services has the potential to close the existing gap of established valuation approaches for ecosystem effects from air pollution. A conceptual framework for this has, for instance, been proposed by Rounsevell *et al.* (2010) and activities within the UK based *Valuing Nature Network* (http://www.valuing-nature.net/) serve as hubs bringing together experts and projects interested in conducting valuation exercises.
One key requirement for an improvement of valuation is the establishment of robust and well documented ecosystem responses to current and future air pollution levels. The work documented in Sutton et al. (2009a), for instance, reflects the wide range of evidence required for the establishment of new ammonia critical levels and loads. Similar to human health effects, the effects of pollutant mixtures requires substantial further research efforts, as these effects are currently not well documented and understood. A second area for improvement addresses the influence of future climate change on the

susceptibility of ecosystems to air pollution effects, respectively their resilience.

Overall it is recognized at present that the valuation of air pollution effects on human health (e.g. Holland et al. 2011) is more advanced than for valuation of effects on ecosystems. Although a first valuation of ecosystem effects has been made in the European Nitrogen Assessment, it is concluded at present that the coupling of dose-response relationships to damage valuation is not yet sufficiently developed for inclusion within Integrated Assessment Models. Further on-going work is therefore needed before the estimated cost of air pollution on ecosystems can be integrated into the optimization of air pollution control measures.

**Summary**

> The concept of Ecosystem Services (ES) and research into the quantification of specific services has the potential to close the existing gap of established valuation approaches for ecosystem effects from air pollution

> Although a first valuation of ecosystem effects has been made in the European Nitrogen Assessment, the coupling of dose-response relationships to damage valuation is not yet sufficiently developed for inclusion within Integrated Assessment Models, for example to provide a basis to optimise control measures.

### Q6. What are the relative costs and benefits of controlling $NO_x$ and $NH_3$ emissions?

The relative contribution of reduced and oxidised N sources to health and ecosystem effects is a function of both the amounts emitted of these compounds, and the spatial distribution of emissions. In the past 20 years, oxidised nitrogen ($NO_x$) emissions have been substantially reduced by implementing emission control legislation on road transport sources and large combustion plants (Vestreng et al. 2009). By comparison, there has only been a modest decrease in reduced N ($NH_3$) emissions during that period, mainly because the agricultural sources have not been subject to the same stringent air pollution controls as already applied to the main $NO_x$ sources. The outcome is that the relative contribution of reduced vs. oxidised N is gradually changing towards a reduced N, as ammonia sources are becoming the dominant emission sources while further $NO_2$ reductions have been implemented.
The consequence of these changes is that many of the most cost-effective measures for $NO_x$ have now already been adopted, while for $NH_3$ there are still many cost-effective methods to reduce emissions which have not yet been adopted. In addition, a major review conducted by the Task Force on Reactive Nitrogen under the CLRTAP with the support of the NitroEurope project showed that $NH_3$ abatement costs were in many cases much less than previously estimated (UNECE 2011).
These interactions may be visualized in Figure 4.4, which compares the benefit-cost ratio of further controls on $NH_3$ and $NO_x$ beyond current commitments as of 2011. These graphs incorporate the $NH_3$ and $NO_x$ mitigation costs as estimated within the GAINS model (e.g. Wagner et al. 2011) and the estimated damage costs of these forms of pollution, based on the European Nitrogen Assessment, as illustrated in Figure 3. While Figure 3 shows that the damage costs of $NO_x$ pollution in Europe are larger than those due to $NH_3$ (mainly because of the contribution of $NO_x$ to tropospheric ozone formation), Figure 4 shows that it is still substantially more cost effective to mitigate $NH_3$ emissions. This illustrates how there are still many $NH_3$ abatement options, which are several times cheaper per kg N abated than for $NO_x$.

The consequence of these differences is that the cost-effective 'headspace' for further $NH_3$ emission reduction is estimated to be much larger than for $NO_x$. Considering the range of uncertainty estimates in Figure 4, there is potential for around 800-1100 kt N further abatement as $NH_3$ using available technical measures, while only around 100-400 t N using available technical measures for $NO_x$. Further reduction beyond these amounts would require other non-technical measures, such as would lead to behavioural change in consumption patterns (e.g., vehicle miles and agricultural production).

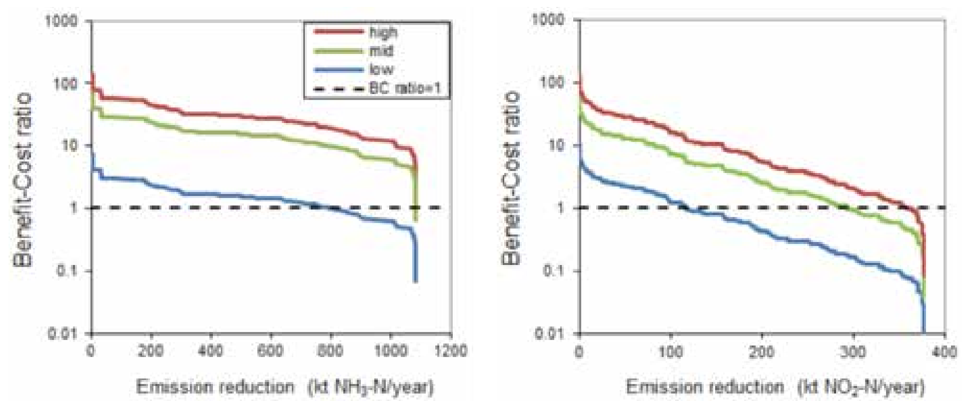


**Figure 4.4** Ratio of marginal benefits of emission reduction over the costs of N-mitigation measures in EU27 for $NH_3$ and for $NO_x$ from stationary sources, for emission reduction from 2010 beyond expected levels in 2020 by effects of current legislation. The comparison shows that there is a much larger potential for cost-effective mitigation of $NH_3$ emissions (800-1080 kt N) than for $NO_x$ (110-370 kt N) (van Grinsven et al. 2013).

**Summary**

- > With reducing $NO_x$ emissions over the last decades, $NH_3$ emissions are now a more powerful driver of effects on ecosystems than oxidized nitrogen, making future control of $NH_3$ emissions a priority.
- > As many measures for control on $NO_x$ emissions have already been implemented, further technical measures become increasingly expensive. By contrast, at the European scale, only a few of the available technical measures for $NH_3$ have so far been implemented, with many low-cost measures still availble. Ammonia therefore offers substantial 'low-hanging fruit' for future air pollution controls.
- > Estimated benefit-cost ratios for technical measures (as included in the GAINS model) justify further European reductions in emissions of around 100-400 kt N for $NO_x$ and around 800-1100 kt N for $NH_3$.



# Chapter 5
# Air Quality and Climate

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**Introduction**

The interactions between air quality and climate are complex and include non-linear feedbacks that are not fully understood.

Firstly, human impacts on the atmosphere influence climate forcing either *directly* through the emission of radiatively active greenhouse gases and aerosols, or *indirectly* through the emission of short-lived reactive compounds that influence the atmospheric lifetime and abundance of radiatively active compounds (greenhouse gases and aerosol particles). Figure 5.1 shows an estimate by IPCC (2007) of the radiative forcing by emissions of compounds resulting from human activities during the industrial era. There are large uncertainties not shown here, particularly from the contributions of aerosols.

Secondly, climate change has a significant impact on the chemical composition of the atmosphere and hence on the level of air pollution. Statistical data analyses performed in North America suggest that higher atmospheric temperatures exacerbate photochemical smog and specifically the concentration of surface ozone with a sensitivity factor (called climate penalty factor) of 2.2-3.2 ppbv/K, depending on implemented pollution controls measures (Bloomer et al. 2009). On the basis of such correlation, Lin et al. (2007) suggest that the number of high-ozone episodes in the industrial areas of the north-eastern US could increase by 10-30% in 2030 and double by 2050. Several and sometimes poorly quantified processes contribute to the influence of climate conditions on air quality. Influencing factors include changes in (1) meteorological conditions including the frequency and duration of stagnant air episodes, (2) atmospheric water vapour content, (3) temperature-dependent natural (i.e., biogenic) emissions of ozone and aerosol precursors including volatile organic compounds (VOCs) by trees and nitrogen oxides by soils, (4) emissions of gases and particles by more frequent wild fires, (5) exchange intensity between the stratosphere and troposphere, (6) production of nitrogen oxides by lightning, (7) mobilisation of dust and sea salt particles by wind, (8) intensity and frequency of precipitation and hence of removal processes for soluble species, (9) temperature-dependent formation and destruction rates of reactive com-