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The ecological effects of increased aerial deposition of nitrogen

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Preface

This booklet represents the outcome of a workshop, held at Charlwood, Surrey, aimed at assessing current knowledge of the impacts on natural and semi-natural ecosystems of inputs of a range of nitrogenous pollutants from the atmosphere. The effects of increased nitrogen on agricultural systems are well known and viewed as highly beneficial in terms of crop production, and it could be anticipated that further inputs of nitrogenous air pollutants in general may have little impact. For many other terrestrial and aquatic ecosystems, however, it is now apparent that a wide range of adverse effects can arise as a consequence of nitrogen input from industrial, transport and agricultural sources. Consequently the protection of ecosystems by emission controls has complex policy implications. The workshop addressed the issue from a multi-disciplinary perspective, covering emissions, atmospheric transport, deposition, effects on aquatic and terrestrial ecosystems, and current proposals for appropriate environmental protection, including the scientifically based "critical loads" approach. We hope our description of this complex issue is concise and readily comprehensible and draws attention to widespread and hitherto unsuspected damage caused by a class of pollutants which, until recently, has been largely ignored.

Nigel Bell
Silwood Park, November 1993

THE ECOLOGICAL EFFECTS OF INCREASED AERIAL DEPOSITION OF NITROGEN.

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1. INTRODUCTION

Nitrogen is a major component of amino acids, proteins and many other biologically active compounds and is, therefore, essential to all forms of life. Thus, the biogeochemical cycling of nitrogen through the aquatic, terrestrial and atmospheric environments has been studied for many years and its pathways and transformations, many of which are controlled by microbiological processes, are well known (Fig. 1.). However, only relatively recently has awareness developed regarding anthropogenic inputs of nitrogen gases to the atmosphere, in the form of air pollutants.

Traditionally, concerns over emissions of gaseous pollutants have been restricted to sulphur dioxide (SO_2) — a highly corrosive and environmentally damaging gas. In Europe, emissions of SO_2 are currently falling sharply as a result of decreased coal combustion and a decline in heavy industry. In contrast, emissions of both reduced and oxidised nitrogen gases have shown a steady increase over the last 40 years or so (Fig. 2).

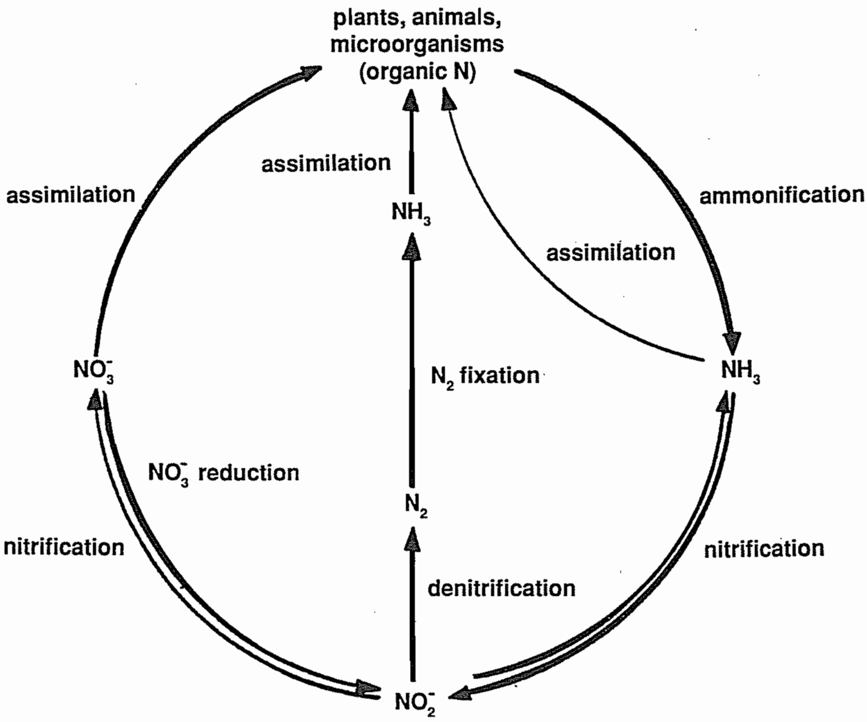

Fig. 1
A generalised N cycle, showing the major processes involved. From: *The Nitrogen Cycle of the United Kingdom* (1983). The Royal Society of London, p.34. (With permission).

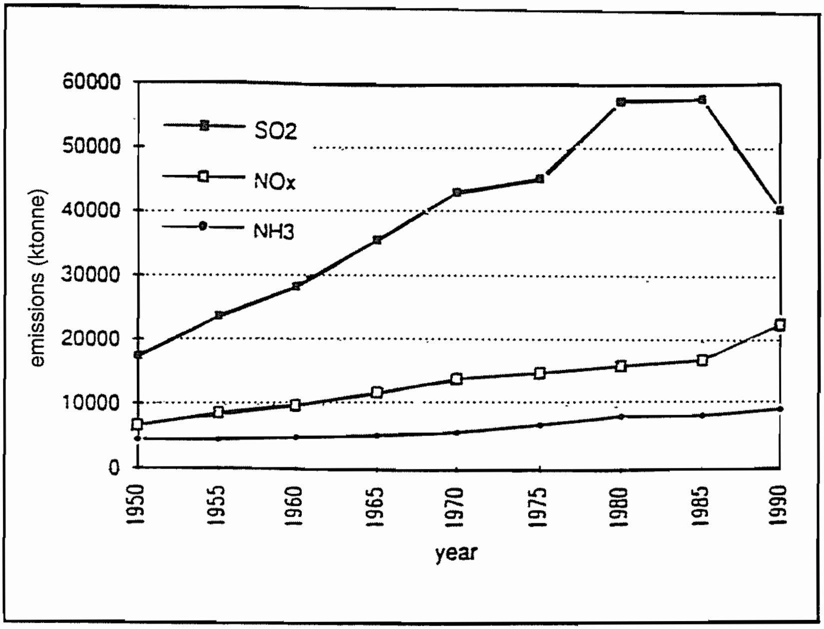

Fig.2.

Emissions of SO_2, NO_x, NH_3 (ktonne yr^{-1}) in Europe 1950-1990. From: Thomas, R. *et al.* (1988). *Emissions of SO_2, VOC and NH_3 in The Netherlands and Europe in the period 1950-2030.* Report No. 758472002. National Institute of Public Health and Environmental Protection, Bilthoven. (With permission).

The background atmosphere over non-industrial countries contains small concentrations of the nitrogen oxides, NO_2 and HNO_3, formed in the atmosphere by the oxidation of nitric oxide (NO) emitted from soils and from biomass burning. The concentrations of NO_2 lie in the range 20 pptv (parts in 10^{12} by volume) to 2 ppbv (parts in 10^9 by volume), while those of HNO_3 are generally smaller by an order of magnitude. In industrial countries such as the UK, these background 'natural' concentrations are supplemented by large quantities of both oxidized and reduced nitrogen. Vehicles and industry contribute the oxidized nitrogen, annual emissions in the UK being about 2 million tonnes. Agriculture provides most of the reduced nitrogen as ammonia (NH_3), largely from animal urine, with UK emissions currently totalling about 350,000 tonnes, although there is a very high level of uncertainty associated with this value. The characteristics and distribution of these sources have an important influence on air concentrations and deposition patterns. The oxidized nitrogen is emitted mainly as nitric oxide, NO, which is oxidized by atmospheric ozone to nitrogen dioxide (NO_2). About half of these emissions are from vehicles and are, therefore, emitted at ground level. The remaining emissions are largely from power stations and are emitted at heights of about 200m. The NH_3 emissions, by contrast, are almost all from ground level sources. The lifetimes of these gases are rather different, with NH_3 and NO having a very short atmospheric lifetime, while that of NO_2 is much longer and results in a substantially greater average transport distance. As well as becoming deposited in gaseous form to the Earth's surface, NH_3 and NO_2 are transformed in the atmosphere to ammonium (NH_4) and to nitrate (NO_3,) respectively.

2. INPUTS OF ATMOSPHERIC NITROGEN TO THE EARTH'S SURFACE

The quantities of fixed nitrogen in the snowpack in Greenland provide a guide to global scale nitrogen deposition over many centuries. The techniques available to investigate historical nitrogen deposition within Europe, however, limit our knowledge to about the last century. In the late 19th century, chemical analytical techniques were rudimentary, but provided estimates of the amounts of nitrogen deposited in rain in England and parts of Germany. The work was stimulated by arguments over the source of nitrogen supply for plants between Lawes and Gilbert, at Rothamsted Experimental Station in England, and Liebig in Germany. The amount of fixed nitrogen in rain deposited at Rothamsted was estimated to be approximately 5 kg N ha^{-1} yr^{-1}. Many other early European measurements suggested that amounts which were wet deposited ranged from 0.5 to 10 kg N ha^{-1} yr^{-1}, with a median value of 2.5 kg N ha^{-1} yr^{-1}. By contrast, the current amounts wet deposited in Europe range from 5 to about 30 kg N ha^{-1} yr^{-1}. More recent work has also shown that, in addition to ammonium and nitrate ions (NH_4^+ and NO_3^-) in rain, the direct uptake of NO_2 and NH_3 contributes an additional 2-50 kg N ha^{-1} yr^{-1} of fixed nitrogen.

The concentrations of major ions in precipitation have been extensively monitored in the UK for the last 8 years. A network of collectors now provides regional estimates of concentrations for NH_4^+ and NO_3^-. These concentrations are combined with precipitation amount to provide the wet deposition input. Hill tops and mountains receive large annual inputs of NH_4^+ and NO_3^- in rain and snow but, for practical reasons, they are difficult areas in which to obtain direct measurements. The few experimental studies to date show that upland Britain not only receives more precipitation but is exposed to much larger concentrations of the major ions, including NH_4^+ and NO_3^-, in hill clouds. The washout of the polluted hill cloud by rain drops from higher altitudes, by a process known as seeder-feeder scavenging, results in greatly enhanced wet deposition in the uplands. Plate 1 shows the wet deposition of total N (NH_4^+ and NO_3^-) over Great Britain, taking into account enhanced values for the uplands. The ratios of concentrations for NO_3^- (or NH_4^+)

cloud/rain range from 2 to 10, and foliar surfaces at upland sites above 600m are exposed to concentrations of NH_4^+ and NO_3^- in the range of 50-3,000 $\mu eq\ l^{-1}$. The importance of clouds is evident from the observation that the Moorhouse National Nature Reserve in the northern Pennines is exposed for around 2000 hours per year at 800m altitude. At these windy, hill-top sites, the cloud droplets are readily captured by foliage. Regional estimates of cloud water deposition throughout the UK have now been made. The amounts of nitrogen deposited are small as a contribution to the national nitrogen input budget but, for sensitive upland ecosystems, this deposition process is very important. The capture of cloud droplets is very sensitive to windspeed and the aerodynamic roughness of vegetation. Isolated trees and upland afforestation, therefore, make an important contribution to local inputs through this pathway. Radiation fog, caused by nocturnal cooling of the ground, is another route of nitrogen deposition. As this requires low windspeeds to form, turbulent deposition of the fog droplets onto vegetation is greatly restricted. The droplets are relatively long-lived, however, and often form in polluted air close to vegetation, so that the concentrations of major ions in the fog water are often very large.

The absorption of NO_2 by vegetation has recently been shown in field experiments to be mainly under stomatal control, and there is very little NO_2 adsorption on cuticular surfaces. Uptake at NO_2 concentrations in excess of 5 ppb shows no evidence of internal resistance to this stomatal route. The relatively simple properties of NO_2 deposition to foliar surfaces make straightforward the extrapolation from experiments to a countrywide scale. A model which uses climatological, land use and air chemistry data computes the deposition rate of NO_2 for each of five land-use categories (arable, grassland, forest, hill vegetation and urban areas) and has been used to provide NO_2 inputs to the country on a 20 x 20 km grid. The amount of NO_2 deposited varies between 2 and 20 kg N $ha^{-1} yr^{-1}$ with the largest amounts in the polluted areas of the Midlands and south-east of England.

The exchange of NH_3 at foliar surfaces is much less straightforward than that of NO_2. Ammonia may be emitted by or deposited on foliar surfaces. A compensation point (the ambient concentration of NH_3 at which there is no net exchange between vegetation and the atmosphere), analogous to that for CO_2, has been shown to be present in agricultural crops. Further, the compensation point probably varies with plant nutrition, physiological activity and environmental conditions (solar radiation, temperature, etc). By contrast, the leaf surface may be a very efficient sink for NH_3, particularly if the leaves are wet and in the presence of an acidifying gas such as SO_2. The current understanding of NH_3 exchange between vegetation and the atmosphere provides a good description of the processes and average rates for non-fertilized vegetation (heathland,

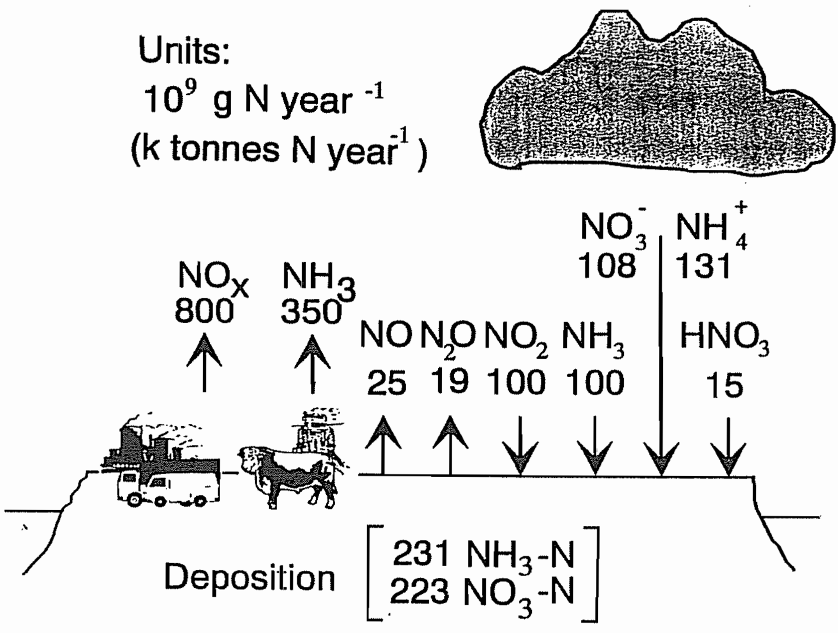

Fig. 3.
Annual exchange between atmosphere and the land surface of reduced and oxidised nitrogen compounds in Great Britain. N.B. There are considerable uncertainties associated with data for NH_3 and NH_4.

moorland and forests), but a rather poor estimate of the net long-term NH_3 exchange over agricultural crops. For the latter, the periods of NH_3 emission following fertilizer application and during senescence appear to be of a similar magnitude to the deposition fluxes at other times. Until a more detailed description of the processes has been obtained, and a suitable model developed, the annual net exchange of ammonia over crops will be unknown. For the remaining vegetation types (unfertilized grassland, woodland, moorland and heathland), the rates of NH_3 deposition are very large. The rates of deposition of NH_3 to different types of vegetation may be calculated from climatological, land-use and air concentration data for NH_3.

The estimates of nitrogen deposited as NH_4^+ and NO_3^- in precipitation and cloudwater may be combined with the net gaseous exchange of NO_2 and NH_3 to estimate the total input to terrestrial surfaces. If the emissions and deposition of reduced nitrogen compounds are compared with oxidized nitrogen compounds, it is clear that reduced nitrogen is removed from the atmosphere much more rapidly. Almost two thirds of UK. emissions of reduced nitrogen are deposited within the UK, whereas only a quarter of the oxidized nitrogen emitted in the UK is deposited before it is transported across the coastline. Further, this leads to similar amounts of reduced and oxidized nitrogen being deposited within the country, despite total emissions of reduced nitrogen (as NH_3) being only half those of oxidized nitrogen. The atmospheric budget for nitrogen also shows that agricultural sources of atmospheric nitrogen contribute as much to the national problems created by nitrogen deposition as the total from heavy industry and vehicles. Figure 3 shows the two-way fluxes of reduced and oxidized nitrogen compounds exchanged between the atmosphere and the Earth's surface annually in the UK.

- The deposition of nitrogen to the Earth's surface has probably increased by up to five times over the last 100 years or so.
- In upland areas nitrogen deposition is enhanced considerably by the deposition of cloud water.
- Large amounts of nitrogen are deposited as either gaseous NO_2, which is well quantified, or NH_3, which is more poorly understood due to complex exchange processes with vegetation.
- There are large uncertainties in both NH_3 emission and NH_3/ NH_4 deposition, partly because of the nature of NH_3 as a reactive gas emitted at low level but also due to dependence on soil type, climate and agricultural practices.
- Reduced and oxidized forms of nitrogen contribute approximately equally to deposition in the UK, indicating similar importance of industrial/transport and agricultural sources.

3. EFFECTS ON SOILS

Nitrogen is essential for all plant growth and in natural ecosystems the supply of plant available nitrogen often limits plant productivity; that is, nitrogen is commonly the limiting nutrient. Initially, any increased atmospheric inputs of nitrogen will have a fertilizer effect in such nitrogen limited systems, with a resulting increase in plant growth and production. If the increased inputs of nitrogen continue, however, one or more of the other essential nutrients may become limiting. Such deficiencies, resulting from an increased input of nutrient from a source external to the soil-plant system, are referred to as induced deficiencies. In sensitive soils, the biologically mediated transformations of nitrogen compounds in soil, and the leaching of nitrate present in excess of plant and microbial requirements, can lead also to soil acidification.

Nitrogen in soils

The mineral component of soils contains very little nitrogen. Rather, most soil nitrogen is in organic compounds and has, ultimately, almost all been derived from atmospheric sources, being input naturally from the rainfall and by fixation. Unpolluted rainfall contains small amounts of nitrate formed as a result of electrical discharges and of ammonium volatilised from animal excreta; together these inputs rarely exceed 1-2 kg N $ha^{-1} yr^{-1}$. In contrast, atmospheric inputs of >100 kg N $ha^{-1} yr^{-1}$ have been reported from parts of The Netherlands and inputs of >30 kg N $ha^{-1} yr^{-1}$ are common over much of Britain.

Atmospheric nitrogen can be fixed by some lichens and mosses but the amounts are small and much larger quantities can be fixed by symbiotic soil bacteria associated with leguminous plants, such as peas, beans and clover; for example, a healthy clover crop can fix more than 100 kg N $ha^{-1} yr^{-1}$. The nitrogen accumulated in soils from atmospheric sources over time is circulated very efficiently in natural soil-plant systems, with only small losses as nitrate in solution in leachates and in the gaseous forms N_2 or N_2O (nitrous oxide) as a result of denitrification. In cropped systems, nitrogen is removed in the harvest and in intensively cropped systems this net removal has to be balanced by inputs of fertilizer nitrogen. Fertilizer inputs of nitrogen to agricultural systems are generally much greater than the inputs in unpolluted rainfall plus those from fixation; fertilizer inputs to intensive arable and grassland systems are often in excess of 200 kg N $ha^{-1} yr^{-1}$.

Although soils can contain considerable amounts of nitrogen, most of it is in the form of complex organic molecules and unavailable to plants. Nitrogen is taken up from soils by plants in solution as the nitrate or ammonium ion. The conversion of the complex organic nitrogen compounds into ammonium and nitrate, often referred to as mineralisation, is carried out by populations of soil bacteria or fungi. Ammonifiers convert the organic nitrogen into ammonium and autotrophic nitrifiers convert the ammonium into nitrate. The rate of breakdown of the organic compounds to forms of nitrogen available to plants depends partly on the size of the populations of these bacteria and fungi in a given soil and on the chemistry of the organic compounds, in particular the carbon to nitrogen ratio. Populations of soil bacteria and fungi in turn vary with soil chemical and physical properties. Bacterial populations are much smaller in acid as opposed to circum-neutral soils and are absent from anaerobic, waterlogged soils. In acidic soils, fungal populations dominate the conversion of the organic nitrogen compounds to ammonium. The breakdown of organic compounds with low carbon to nitrogen ratios (< c.30) proceeds much faster than those with high carbon to nitrogen ratios.

Impacts of nitrogen mineralization and NO_3 leaching on soil acidity
The conversion of organic nitrogen compounds to ammonium involves the consumption of protons whereas the conversion of ammonium to nitrate involves the release of protons and is, therefore, an acidifying reaction. In most undisturbed natural ecosystems the production and consumption of protons during the cycling of nitrogen between plants and soil and between organic and inorganic compounds is balanced, with no net soil acidification. Nitrification of atmospherically derived ammonium, will, however, lead to a net soil acidification.

The nitrate ion is very mobile and nitrate present in soil solution in excess of plant and microbial requirements is readily leached in drainage waters. Leaching of the nitrate anion is accompanied by a cation; where the cation is calcium or magnesium, a net acidification of the soil will take place unless the leached base cation is replaced from mineral weathering. In acid mineral soils, with an exchange complex dominated by aluminium, the leaching of nitrate can be accompanied by ionic aluminium, which is potentially toxic to plant roots and to freshwater organisms. In most natural systems, there is little or no nitrate in excess of plant demand during the growing season and concentrations of nitrate in drainage waters are consequently small. An important exception is stands of alder (*Alnus glutinosa*) in which large amounts of nitrate can be produced, with a resulting rapid acidification of soils and relatively large outputs to drainage waters. Atmospheric inputs of nitrate at levels in excess of plant demand will also lead to leaching of cations, with resulting soil acidification and/or mobilization of aluminium.

Impacts of increased nitrogen inputs on soil acidification and cation leaching

Increased atmospheric deposition of nitrogen, in either the oxidised or reduced form, can thus lead to the development of nutrient imbalances and, in sensitive soils, to soil acidification and to the mobilization of potentially toxic aluminium. In studies on the impact of acidic deposition and acidifying pollutants, much greater emphasis has been placed on the acidifying effects of atmospheric sulphur compounds. Indeed, in most situations pollutant inputs of sulphur compounds do exceed those of nitrogen. Additionally, the inputs of nitrogen to most natural ecosystems and to production forests have hitherto been retained in the system and utilised in plant growth or by microbes; they have had a fertilizer effect. In this situation, there could still be soil acidification due to nitrification of ammonia but there would be little acidification due to nitrate leaching. In recent years, however, the emissions of sulphur dioxide have declined while those of nitrogen compounds have increased. The relative importance of nitrogen inputs has therefore increased. The accumulation of nitrogen in some ecosystems over the past 20 or so years has resulted in more nitrogen being available than can be utilised in the plant-soil system; there has been a gradual increase in nitrate leaching from these systems and such ecosystems have been referred to as nitrogen-saturated. In a few situations, nitrogen has dominated acidic inputs over a significant period; this is usually in areas with large inputs of ammonium, derived from animal wastes.

Perhaps the most dramatic examples of soil acidification resulting from atmospheric inputs of nitrogen come from The Netherlands. Soil pH values of less than 3.5 have been reported from forest soils receiving annual inputs of more than 100 kg N ha^{-1}, mainly as ammonium, the soil acidification being driven mainly by the production of protons in the nitrification process. However, nitrate is also being produced in excess of plant and microbial requirements and further acidification results from nitrate leaching. Aluminium has become the dominant cation in soil solutions in these extremely acid soils and damage to fine roots is resulting with consequent forest damage.

In Scandinavia, the uplands of Wales, north-west England and Scotland, the main impact of acidic deposition has been on freshwater ecosystems, with the reduction or elimination of fish populations at many locations. The damage has resulted from a long term acidification of soils, initially as a result of natural processes but, since the nineteenth century, increased by acidic deposition to the point where further inputs of acidity have resulted in the mobilisation of toxic forms of aluminium from soils into surface waters. As noted above, in most locations the sulphate ion has been more important than nitrogen compounds in this acidification. However, as sulphur inputs decline and as systems become nitrogen saturated the importance of nitrate driven leaching increases. For example, surveys of lakewater chemistry in Norway in 1990 showed that nitrate concentrations have increased over the last 10 years while sulphate concentrations have decreased. In the UK significant leaching of nitrate has been shown from older plantation forests in the Welsh uplands. It would appear that in the later stages of forest rotations in these areas, the forest-soil systems become effectively nitrogen saturated. The system is unable to utilise all the available nitrogen and nitrate leaching results. The limitation on the utilisation of the available nitrogen in these older forests seems to be the availability of phosphorus and potassium. The 'excess' production of nitrogen in plant available forms probably results from a combination of factors, including increased atmospheric deposition and rates of mineralisation due to changed soil conditions. In these areas there can also be a large increase in nitrate flux from soils into drainage waters following felling, with consequent impacts on aluminium mobilisation in acidic soils.

Impacts on soil biota

Soil acidification can also result in a consequent change in soil fauna and flora which can influence nutrient cycling and, ultimately, plant growth. Thus, for example, different species of earthworms are found over different pH ranges, the casting earthworms being most common in circum-neutral or slightly acidic soils. A change in mycorrhizal assemblages has also been reported to result from soil acidification.

- Increased deposition of nitrogen from the atmosphere can lead to soil acidification, with effects on both terrestrial and aquatic ecosystems.
- As sulphur emissions fall, nitrogen deposition is becoming increasingly important in acidification of soils.
- High deposition of nitrogen in The Netherlands has resulted in widespread changes in forest soils.
- Acidification of soils can result in leaching of basic cations, mobilisation of potentially toxic aluminium and adverse impacts on soil biota.

4. EFFECTS ON TERRESTRIAL VEGETATION

Differences between ecosystems in their response to nitrogen

Nitrogenous fertilizer is added to many crops at very high rates to increase the yield, so any nitrogen that is added to agricultural ecosystems from the air might be interpreted as being beneficial. In practice, however, it is extremely doubtful whether aerial deposition is ever sufficient, or occurs at the right time of year, to be of any real benefit to crops. In forests, if the sole aim is to maximise timber production, then aerially deposited nitrogen may also be seen as potentially beneficial. Other species in forest ecosystems, however, may have conservation value, and there is abundant evidence that some herbaceous species, bryophytes, lichens and mycorrhizal fungi are adversely affected by nitrogen concentrations that have no short-term deleterious effect on the trees. Similarly, some *Sphagnum* species that are a vital component of wetland ecosystems have such a low capacity to utilise nitrogen that even a modest addition may be toxic. Judging the effects of aerial nitrogen then depends on the ecosystem and the aims of management. In contrast to crop and timber production, most ecosystems that are not under intensive agriculture, such as mires, heaths, dunes and calcareous grasslands, have relatively low supplies of nitrogen and in many cases this plays a key role in maintaining species diversity. Such habitats are of great conservation value, both nationally and internationally. For example, it has been calculated that 65-80% of the central European Red List of threatened plant species are restricted to ecosystems with the lowest nitrogen status. Many of these ecosystems that have an intrinsically low nitrogen supply are situated in parts of Europe where there are high rates of nitrogen deposition, so important habitats may be damaged.

Ways in which nitrogen may affect ecosystems

Although all plants need nitrogen, they differ greatly in their requirements, not only in terms of the concentration that promotes optimum growth, but also in their capacity to utilise the gases NH_3 and NO_2, and the ammonium and nitrate ions that are in the soil solution. These differences are the main bases for concern about aerial nitrogen. Some species, such as perennial ryegrass (*Lolium perenne*), require large amounts of nitrogen in order to thrive. When the supply is high they grow rapidly and are aggressive competitors. Other species, such as mat-grass (*Nardus stricta*) have a lower optimum and lower capacity to respond in terms of growth or spread. In general, species of acidic, waterlogged soils have a greater capacity to use ammonium nitrogen than those of neutral or calcareous soils, so there may be differences in response to deposited nitrogen, depending on the species and the balance between nitrate and ammonium.

Many fertilizer experiments have shown that if nitrogen is added to low-nitrogen ecosystems highly competitive, aggressive species, that may originally be only a minor component, expand and out-compete those with a more limited potential to respond. For example, in a study in which 53 kg ha^{-1} yr^{-1} of nitrogen were applied annually to mown chalk grassland for ten years, dry matter production was increased from 730 to 1967 kg ha^{-1}, almost entirely because of stimulation of the grass, red fescue (*Festuca rubra*). The number of species dropped from 28 to 12 m^{-2}, those lost including species that are typically found in short, open turf such as stemless thistle (*Cirsium acaule*) and scabious (*Scabiosa columbaria*). In addition, this experiment demonstrated that the effects of nitrogen are modified by management, because there was a difference between mown and unmown plots. Species density dropped with nitrogen addition in the unmown plots, just as in those that were mown, but a different suite of species increased, notably the tall-growing nitrophiles, creeping thistle (*Cirsium arvense*), goosegrass (*Galium aparine*) and stinging nettle (*Urtica dioica*). Removal of biomass by cutting, grazing or burning alters the canopy density and architecture and thus the availability of light and nutrients, resulting in shifts in species composition. It may also remove sufficient nitrogen to counteract aerial deposition. The aggressive species that increase with nitrogen application tend to have a high dominance index and a high capacity to absorb nitrogen. Conversely, species that are ousted tend to be small, short lived, slower growing and with a limited capacity for lateral or vertical spread. Some are known to have a limited capacity to reduce nitrate chemically.

Nitrogen may also alter ecosystems by creating nutrient imbalance. This may occur in two ways: first, ammonium production from organic matter and secondly nitrate leaching can lead to acidification and a reduction in supply of magnesium: and second, the large inputs of plant available nitrogen can also increase the demand for other nutrients and can induce a deficiency of magnesium or other nutrients. A deficiency in magnesium is an important factor in European forest dieback. Large inputs of ammonium can also result in soil solutions with high concentrations of the ammonium ion; this can 'block' the uptake of other essential cations, such as potassium, from soil solution. A molar ratio of $NH_4^+:K^+$ of 5:1 in soil solution has been suggested as critical. The acidification of soils by nitrogen deposition affects plant growth both directly and indirectly. The mobilisation of aluminium in acid mineral soils can result in root damage. The tolerance of plant roots to aluminium in soil solution, however, varies considerably between plant species. Thus, death of fine roots has been recorded in Norway spruce (*Picea abies*) once the calcium to aluminium ratio in soil solution falls below 1:1 but fine root death is found in beech (*Fagus sylvatica*) at base cation to aluminium ratios of c.1:2. Root damage due to low base cation to aluminium ratios in soil solution is considered an important factor in forest dieback.

In agriculture and horticulture, the addition of nitrogen may alter the allocation of assimilates between shoot and root. Plants given high nitrogen may produce much greater amounts of leaf and less flower and root. Changing seed output may have obvious effects on long-term survival of the population, while differences in root/shoot balance and leaf chemistry may alter drought or frost resistance. There is evidence that effects on frost resistance may be of some significance in the decline of heather (*Calluna vulgaris*) in The Netherlands.

The evidence that nitrogen deposition affects ecosystems

Whilst the potential effects of an increase in nitrogen supply to an ecosystem are easily demonstrated, it is much more difficult to determine whether aerial deposition is actually having any effect on natural ecosystems. The problem lies largely in the difficulty of interpreting published fertilizer experiments and in establishing cause-effect relationships from long-term floristic records.

Almost all of the published fertilizer experiments involved the application of nitrogen on only one or a few occasions, usually in spring when plant demand from plants was highest. Also, the rates of application were often considerably higher than the rates of aerial deposition and, in many cases, the nitrogen was not supplied alone but in combination with other nutrients, such as phosphate. Few published experiments used nitrate and ammonium in the proportions that are deposited; as aerial nitrogen input in most areas of the UK is dominated by NH_3 and NH_4, this is a serious limitation. It is only recently that experiments have begun in which the nitrogen is added continuously or very frequently throughout the year, and with the appropriate NH_x/NO_3 ratio. Until data from these experiments are available, it is not possible to determine whether there is a difference in impact between traditional fertilizer experiments and long-term aerial deposition. Furthermore, not all communities may respond to deposited nitrogen, because there may be other factors that limit response. For example, in Derbyshire, no effect of nitrate was found on the yield of oat-grass (*Arrhenatherum elatius*) or sheep's fescue (*Festuca ovina*) seedlings growing in plots on limestone, but there were positive effects of watering and phosphate. Further work is needed on the interplay between nitrogen and other potentially limiting factors in order to determine which components of which ecosystems respond to additional nitrogen.

Turning to field observations, in The Netherlands, Scandinavia and Britain many floristic changes in the last few decades have coincided with temporal changes and spatial patterns of nitrogen deposition. Many other environmental factors have changed over the same period, however, notably management and grazing of semi-natural ecosystems. Associations and correlations do not establish a cause-effect relationship but need to be supported by experimental manipulation. This has been done in The Netherlands, in particular, where there have been dramatic changes in the flora of woodlands, heathlands and calcareous grasslands. Dutch workers present a persuasive case that changes such as the decline of heather (*Calluna vulgaris*) in dry heaths, the increase of purple moor grass (*Molinea caerulea*) in wet heaths, and the increase in slender false brome (*Brachypodium pinnatum*) in calcareous grasslands, are linked to aerially deposited nitrogen.

The data available for Britain have recently been collated and allow a cautious interpretation of observed changes in ecosystems that might indicate effects of nitrogen deposition. There are two kinds of evidence: chemical analysis of plant tissues and comparison of floristic records. Chemical analysis of heather and bryophytes showed that patterns in the nitrogen content are compatible with temporal and spatial differences in aerial deposition of nitrogen. Comparison of heather from different locations suggested that the spatial pattern of tissue nitrogen is determined by the pollution climate. In Cumbria, at the site showing the greatest increase in bryophyte tissue nitrogen over 15-35 years, there has also been a decline in the bryophyte flora of permanent plots. It has been suggested that the cause underlying the spread of *Brachypodium*, with its parallel decrease in species richness, is that it is largely due to the decline in grazing but that "it is not unreasonable to suppose that the abundance of *Brachypodium* in Surrey and Kent, and recent increase in Sussex and Hampshire, are partly due to large inputs of atmospheric nitrogen".

Evidence that current atmospheric nitrogen deposition directly reduces the performance of certain plants is particularly strong for ombrotrophic (*Sphagnum*) species. The widespread decline of these plants in the southern Pennines since the early 19th century was associated with high concentrations of SO_2, but their poor recovery in recent decades appears more closely related to the increased deposition of nitrogen compounds. Although ombrotrophic *Sphagnum* relies for growth on moderate nutrient input in precipitation, the importance of the present atmospheric nitrogen supply in the decline of southern Pennines *Sphagnum* was demonstrated by transplanting *S. cuspidatum* from 'clean' Welsh sites, where it grows in abundance, into the southern Pennines. When the plants were grown in artificial bog pools isolated from the peat and receiving only atmospheric deposition, they rapidly accumulated nitrogen. High nitrogen concentrations were also found in relict southern Pennine populations of *S. cuspidatum*, and laboratory experiments indicate that a high nitrogen content inhibits growth. The reduction in the growth of *S. cuspidatum* was observed at concentrations, of either NH_4 or NO_3, within the range currently observed in precipitation in the southern Pennines.

Table 1
Change in nitrogen content of *Racomitrium lanuginosum* from the summit area of Ingleborough, North Yorkshire, 1879-1989.
(J.A. Baddeley, 1991, Ph.D. thesis, University of Manchester.)

Collection Year	N content mg g^{-1} $\pm$SE
1879	4.6 $\pm$ 0.3
1956	8.0 $\pm$ 0.2
1989	12.3 $\pm$ 0.5

The mechanism by which ammonium and nitrate can reduce the growth of *Sphagnum* is not known, but presumably results from their (or their assimilatory products) accumulation in the tissues to toxic concentrations. Physiological signs of disturbance following excess nitrate application to *Sphagnum* are observed quite rapidly in changes in the activity of nitrate reductase. The normal capacity for induction of this enzyme in response to nitrate was lost when *Sphagnum* was collected from a 'clean' site and transferred to the southern Pennines. The uncoupling of nitrate deposition and nitrate reductase activity is probably a response by the plant to prevent assimilation of nitrogen to toxic levels.

Vegetation surveys in the past 30-40 years in upland Cumbria reveal a disappearance of numerous species over a period when nitrogen deposition rose significantly but rainfall acidity and SO_2 concentrations fell. Furthermore, measurements of the nitrogen content of the moss, *Racomitrium lanuginosum*, in upland Britain have shown a three-fold increase since 1879, with half of this taking place since 1956 (Table 1). The decline in certain bryophytes and lichens is consistent with a direct adverse effect of nitrogen on some species, but it remains possible that these species may have suffered simply from competitive exclusion by nitrophilous organisms. Field evidence also exists for the replacement in upland Cumbria over recent decades of 'sensitive' species of moss by those with a broader ecological tolerance. One of the more sensitive species, *Racomitrium lanuginosum*, which has declined in northern

England, has a nitrogen content which is, broadly, inversely related to its abundance.

Overall, it is clear that aerially deposited nitrogen may be favouring 'aggressive' plant species, decreasing plant diversity and altering the flora and fauna of some of the most ecologically important ecosystems in Europe. The severity of the effects, their location, the relative importance of NO_3 and NH_x, and the interplay with other environmental factors, urgently requires much more research.

- Species differ in their capacity to utilise extra nitrogen which may lead in some cases to adverse physiological effects, and which may be manifested as increased sensitivity to natural stresses.
- Ecosystems of low nitrogen status are often species rich, of conservation importance and contain a large proportion of threatened and rare species.
- Experiments in which extra nitrogen has been added show that there is often an increase in aggressive competitors and a decrease in species diversity.
- Records show that the flora of many ecosystems has changed over the years in parallel with increased nitrogen input.
- A major decline of *Sphagnum* and other bryophytes is associated with increased nitrogen deposition in upland Britain.
- Nitrogen deposition can lead to major nutrient imbalance and release of toxic aluminium in soils, which may play a major role in forest die-back.

5. EFFECTS ON TERRESTRIAL HERBIVOROUS INSECTS

In recent years interest has been developing into the impacts of increased nitrogen deposition on herbivorous invertebrates, particularly with respect to their changed performance in relation to plant nutritional status. Such changes have the potential to influence other trophic levels, yet the impacts of increased nitrogen at the community level are poorly understood.

Most work has concentrated on the effects of gaseous nitrogen deposition on herbivorous insects, largely because of their short generation times and ease of use as experimental animals. Evidence has accumulated that insect herbivore populations are often increased by a number of pollutants, including NO_x and NH_3 at typical ambient concentrations, but with decreased performance at higher concentrations. It is well known that the levels of available nitrogen in the foodplant affect the physiological efficiencies and feeding rates of many herbivores which, in turn, may influence their interactions with climate and natural enemies. Increased available nitrogen in herbaceous plants will decrease the relative rate of food consumption by chewing insects. This is compensated for by the insect spending a shorter time feeding, which in turn may reduce its exposure to parasitism and predation. On woody species, in contrast, the generally lower amounts of water in the plant tissues do not permit substantial changes in insect feeding rates. In this case the increased nitrogen content of the plant may, instead, affect the growth rate and generation times of the herbivores, again with the potential to modify exposure to adverse weather and natural enemies. Changes in the nitrogen status of plants may also affect their levels of secondary chemicals, such as toxins which are part of the plant defences against herbivores and pathogens. However, this aspect has largely been ignored in studies on the impacts of atmospheric nitrogen deposition.

Several of the earliest studies investigated the causes of increased defoliation of vegetation by both chewing and sucking insects, which had been observed on motorway verges where there are prevailing high levels of NO_x. These indicated a probable role of nitrogen gases in stimulating insect performance, but precise causality was difficult to demonstrate in view of other possible confounding factors in the motorway environment, such as dusts and the influence of wind gusts on natural enemy efficiency. In contrast, laboratory studies have demonstrated unequivocally that NO_2 can produce marked changes in insect performance at concentrations within the range recorded in the field. Such changes have been shown in insects feeding on both herbaceous and woody plants. These usually take the form of an increased growth rate of the herbivores after several days fumigation of the host plants. This has the potential to increase population growth rates, and hence density, via reductions in generation times and increased body size and fecundity. These effects of NO_2 on herbivores are induced via changes in the plant, as in most experiments the insects have been placed on the host after the end of fumigation. Furthermore, experiments in which the insects are fumigated while feeding on artificial diets show no effects on growth rate within the same concentration range employed in fumigation of the host plants.

These effects of NO_2 have the potential to give rise to local outbreaks of the herbivores concerned or may even change the status of a species from a minor local problem to a widespread major pest, as has happened to a number of forest insects in Europe over the last 20 or 30 years. In the field NO_2 is often accompanied by elevated concentrations of other pollutants, such as SO_2 and ozone, which have also been shown to affect insect performance via changes in the host plant. Thus there is difficulty in determining the precise role of NO_2 in chamber filtration experiments, where ambient air has been shown to stimulate insect growth rates compared with charcoal-filtered clean air. This also applies to studies along gradients of NO_2 in the field, where increased growth rates have been shown on both cereal and conifer aphids in relation to increasing pollutant concentration, where there are similar gradients for SO_2. While it remains extremely difficult to assess the ecological importance of NO_2 in terms of effects on insects, in the case of NH_3 there is excellent evidence of major ecosystem deterioration in which this pollutant plays a major role. In The Netherlands widespread replacement of heather by grass-dominated communities has taken place in recent years. This is linked to outbreaks of the heather beetle (*Lochmaea suturalis*) causing extensive defoliation, which appears to be induced by increased plant nitrogen levels, largely arising as a result of NH_3/NH_4 deposition.

- Little research has been performed on the effects of nitrogen deposition on invertebrates at concentrations applicable to natural ecosystems.
- Current knowledge suggests that:
  - no direct effects occur on the insects themselves;
  - most insect herbivores show increased growth rates of individuals and raised population densities, arising from indirect effects via the host plants.
- Evidence for major impacts of increased nitrogen deposition on invertebrates in the field comes from The Netherlands, where it is implicated in increased severity of heather beetle attack, leading to a decline of heather on heathlands.

6. EFFECTS ON AQUATIC ECOSYSTEMS

In natural ecosystems subject to low input (<5kg N ha^{-1} yr^{-1}), the cycling of nitrogen compounds is extremely efficient, resulting in negligible leaching of inorganic forms of nitrogen (NH_4^+ and NO_3^-) into freshwaters. Consequently, nitrogen gradually accumulates in soil and vegetation or is transported into lakes and streams as organic nitrogen. Any minor perturbation of this large pool of nitrogen can, therefore, significantly increase the runoff of inorganic and organic forms of nitrogen into freshwater. Most natural ecosystems have an intrinsic capacity to utilise increased nitrogen up to a certain threshold before leaching occurs, and generally have a much higher threshold for ammonium nitrogen than for nitrate nitrogen. The relative importance of any individual source of nitrogen, as a proportion of the total nitrogen present, shows considerable temporal and geographical variation, especially if its origin is from a point rather than a diffuse source. While most human activities give rise to point sources or discharges of nitrogen (*e.g.*, sewage and industrial waste), aerial deposition can be considered as the most diffuse source of N from non-natural sources. The effects of increased deposition depend not only on the capacity of soils and vegetation to utilise the extra N, but also on the net result of uptake processes. For example, uptake of NH_4^+ is a soil acidifying process while nitrate run-off is an essential component of freshwater acidification. Nitrate and ammonium leaching may alter the trophic status of lakes and streams although, for most upland aquatic systems, the limiting nutrient is likely to be phosphorus rather than nitrogen. Quantifying impacts in lowland waters is particularly difficult because of the problems in separating aerial input from agricultural and urban contributions.

Not only do nitrate levels in British rivers increase from higher to lower elevations but also from the north to south, and west to east of the U.K. This reflects the geographical location of upland and lowland regions and the extent to which agricultural and urban sources contribute to nitrate concentrations. Seasonal cycles of nitrate concentrations are evident in both upland and lowland systems, giving winter maxima and summer minima.

Current total nitrogen inputs to the land surface frequently exceed 30 kg ha^{-1} yr^{-1} in the upland regions of central and southern Scotland. Despite these relatively large inputs, over 90% is utilised in most catchments with heathland and other natural vegetation. Nevertheless, elevated nitrate levels have been recorded in some lakes in upland Wales and the Pennines which closely match the areas of maximum nitrogen deposition. Changes in weather patterns may also create considerable noise in "normal" season patterns. After droughts nitrate peaks may increase by 2-20 fold during the first phase of increased run-off in summer and autumn. Winter nitrate peaks may also increase during the melt of polluted snow, when major soil pathways are bypassed and biological activity is minimal.

The highest nitrate leaching rates are generally from arable land, declining in the sequence arable, pasture, woodland, moorland. The impact of increased deposition is likely to be small on drainage water from arable and pastureland. Streams draining woodland and heathland are more likely to reflect increased nitrogen input. At many heathland sites, nitrate in streams is generally close to the expected background concentration of 0-10 μeq l^{-1}, even where nitrogen inputs are relatively high. Improvement of heathland areas for agriculture can also result in short-term increases in nitrate and even pulses of ammonium-nitrogen have been detected in drainage channels, especially where peatlands are ploughed and fertilised. Probably the most significant land-use change in the U.K. during the past 50 years has been the conversion of rough moorland pasture to plantation forests, each stage of the development of which has the potential to modify streamwater quality and quantity. In some, but not all, instances the ploughing/planting phase results in elevated nitrate concentrations. Where fertilisers are added containing nitrogen in a reduced form (*e.g.* urea) leaching losses are generally low (< 10%), but where mixed fertilisers are applied (*e.g.* NH_4NO_3) leaching rates can virtually double. Interestingly, the application of nitrogen fertilisers to forests has virtually ceased, possibly reflecting the benefit from increased aerial deposition. A key question to be resolved is the extent to which growing forests can increase nitrogen input to catchments and consequently cause significant nitrate leaching. Recent studies in Wales have shown a significant correlation between forest age and nitrate leaching into streams. In old forest stands, throughfall and soil water nitrate levels have been shown to have increased to such an extent that a large proportion is surplus to biological requirements, resulting in significant nitrate leaching. This response to a virtual doubling of nitrogen input via forest interception processes also may be occurring in other upland regions of the UK. The greatest disruption of the nutrient cycle, and potentially the greatest acidification/eutrophication threat, is during the felling phase when nitrate levels in stream water increase dramatically and remain high for 2-3 years. In sensitive catchments this extra supply of mobile anions is likely to induce pH depression and increase the concentrations of toxic forms of acid soluble metals, such as aluminium. The amplitude of the nitrate peak depends on a number of factors but as a general rule, the larger the area of forest felled and the smaller the area of non-forest ground vegetation, then the greater the nitrate peak.

Because of the key role of nitrogen as a structural component of proteins, any change in the cycling of nitrogen in freshwaters may have serious biological implications, especially for primary production. The extent to which nitrogen leaching (from whatever source) into lakes and streams produces a biological response depends on a variety of physical and chemical interactions. The rate of supply and fate of nitrogen compounds depends on rainfall pattern and intensity, streamflow, catchment-to-lake ratio (flushing rate) and sediment characteristics. In addition, mixing depth and light penetration are important in lakes. These mediating factors also determine the relative abundance of a whole suite of key nutrients (*e.g.* P, Mo, Si) which ultimately determine primary production, but it should be reiterated that for most upland catchments in the UK, phosphorus rather than nitrogen is the limiting nutrient. There is still a degree of uncertainty as to the likely impact of increasing nitrate levels on primary production. This stems from the ability of certain algal species to fix nitrogen from the atmosphere, and so, if nitrate levels increase, there may be a shift in the algal composition away from nitrogen-fixing species. Any significant compositional changes may have a knock-on effect on grazing invertebrates and, ultimately, fish. In general, the more diverse invertebrate communities are found in waters with higher nitrate concentrations. This may not necessarily be a direct effect and may simply reflect the fact that streams rich in nitrate are usually also of high base status. The biological effects of surface water acidification are well known and include:

(a) reduced rates of decomposition,
(b) changes in species composition of primary producers (this has enabled the use of diatom succession to construct the pH history of lochs),
(c) reduced diversity of invertebrates, notably mayflies, caddis, molluscs and crustacea,
(d) decline and loss of fish stocks,
(e) reduced densities of some birds and mammals using streams and rivers as feeding areas *e.g.* dippers (*Cinclus cinclus*).

There is no reason to expect that the impact of nitrate-induced acidification will differ from that caused by sulphate. In the present circumstances, however, there may be occasions when small increases in nitrate may push some freshwater systems over the acidification threshold (*e.g.* following forest clearcutting and during snowmelt or rain induced episodes). The role of nitrate during episodes requires further attention especially in the context of flow/concentration patterns and the extent to which changes in nitrate concentrations have a direct effect on pH, aluminium and calcium levels. To date there are few studies which have measured the direct impacts of atmospherically derived nitrogen on aquatic biota. Consequently there is little or no evidence in the UK from which to draw any valid conclusions.

Unlike freshwater systems, which are predominantly phosphorus limited, the balance between nitrogen and phosphorus limitation in the marine environment is relatively even, with a tendency for phosphorus limitation in low salinity waters and nitrogen limitation in the high salinity waters of the open sea. Recent concern over increased nitrogen input to coastal waters has been generated by the occurrence of blooms of specific planktonic algal species that, in some areas, have resulted in increased oxygen consumption with associated deleterious effects on benthic communities. Nitrogen deposition to the North Sea averages about

10 kg ha^{-1} yr^{-1}, which is some 2-5 fold lower than the total land based input via large river basins. While input of nitrogen from land based and atmospheric sources can be calculated relatively easily and with some degree of accuracy, any predictions concerning possible effects on nitrogen status in the sea are far more difficult because of the complex mixing, uptake and removal process in the sea. Even quantification of background nitrogen concentrations is uncertain because of wide variability associated with upwelling of nutrient rich water from deeper zones.

- Large concentrations of nitrogen in drainage water indicate catchment inputs in excess of soil/plant demand.
- Elevated nitrate concentrations are present in streams and lakes on acid-sensitive, non-fertilised catchments in areas of Wales, Cumbria and the Pennines, as well as Central Europe and Scandinavia.
- Changes in trophic status may occur, but in most cases these are small, because the system is limited by phosphorus availability rather than nitrogen.
- The acidifying potential of increased nitrate may be most important during short term episodes or after serious disruption of nitrogen cycles *e.g.* clear-felling.
- The effects of nitrogen deposition to the sea are very uncertain, but may be important in relation to algal blooms.

7. CONTROL MEASURES TO REDUCE NITROGEN DEPOSITION

Controls on emissions of air pollutants aimed at protecting sensitive ecosystems initially concentrated on SO_2. With increasing evidence of adverse impacts of atmospheric nitrogen pollutants, however, both directly and indirectly via their role in photo-oxidant formation, pressure has grown for stringent controls on emissions of these compounds. This was focussed in the first place on motor vehicles and, to a lesser extent, on large combustion plant emissions but currently concerns are also growing over agricultural sources.

Under the European Commission's (EC) Large Combustion Plant Directive, the UK is committed to reduction in the total emissions of nitrogen oxides from existing installations of 15% by 1993 and of 30% by 1998, on a 1980 baseline. In order to fulfil this commitment low NO_x burners (where the combustion conditions are adjusted to minimise NO_x emissions) are being fitted to 12 of the largest coal-fired power stations. Increasingly stringent legislation has also been enacted by the EC in recent years to control NO_x emissions from motor vehicles. At first the UK supported the employment of "lean-burn" engines, with a reduced air to fuel ratio, for this purpose. Under the terms of the EC Directive 91/441/EEC, however, more rigorous standards were required for all new cars from 1992 onwards, which entails the use of catalytic convertors. This will ultimately reduce NO_x emissions by about 80%, given present traffic levels, but there are concerns that predicted increases in motor vehicle usage will to some extent counteract this. Controls on NH_3 have scarcely been addressed by most European countries, but represent a very different problem in view of the diffuse and agricultural nature of its source. Only The Netherlands has made any substantive progress so far, with proposals to incorporate animal wastes into soils (rather than spreading them on the surface), modifications of animal housing and manure storage and even limitations on numbers of farm animals; recently, however, both Sweden and Denmark have made similar moves to control ammonia.

The current reductions in emissions required by the EC Directives are effectively uniform across regions and countries. It can be argued that,

Table 2

Critical loads for nitrogen (kg ha^{-1} yr^{-1}) to (semi-) natural terrestrial and wetland ecosystems, indicating levels of uncertainty.

Habitat	Critical load	Level of uncertainty
Acid (managed) coniferous forests	15 - 20	*
Acid (managed) deciduous forests	<15 - 20	*
Calcareous forests	unknown	
Acidic (unmanaged) forests	unknown	
Lowland dry heathlands	15 - 20	**
Lowland wet heathlands	17 - 22	**
Species-rich lowland heaths/acid grassland	7 - 20	*
Arctic and alpine heaths	5 - 15	(*)
Calcareous species-rich grassland	14 - 25	**
Neutral-acid species-rich grassland	20 - 30	*
Montane-subalpine grassland	10 - 15	(*)
Shallow soft-water bodies	5 - 10	**
Mesotrophic fens	20 - 35	*
Ombrotrophic bogs	5 - 10	*

**reliable, * quite reliable, (*) best guess. From: Grennfelt, P. & Thörnelöf, E. (Eds.), 1992, *Critical loads for nitrogen.* Nordic Council of Ministers, Copenhagen. (With permission).

Table 3

Critical levels for nitrogen gases

NO_x (expressed as NO_2)	30	μgm^{-3}	1 year*
	95	μgm^{-3}	4 h*
NH_3	8	μgm^{-3}	1 year
	23	μgm^{-3}	1 month
	270	μgm^{-3}	1 day
	3,300	μgm^{-3}	1 h

For all vegetation categories. * in combination with concentrations of SO_2 (annual mean) and exposures of ozone below their critical levels.

while undeniably having a beneficial effect in terms of reducing ecosystem damage, they are arbitrary and not based quantitatively on scientific evidence. An alternative approach is currently being developed and adopted as a European-wide initiative to minimise harmful effects of transboundary air pollution. This is known as "Critical Loads" and "Critical Levels", aimed at maximising the environmental benefits of emission reductions, and is the product of the United Nations Economic Commission for Europe's (UNECE) Convention on Long-Range Transboundary Air Pollution. It involves all European countries, not just members of the European Union. The critical load is defined as the maximum rate of deposition per unit area of a given type of ecosystem that can be endured indefinitely without 'adverse' effects taking place. It can be defined in various terms, which include total nitrogen. As described earlier, different categories of ecosystem can tolerate different extra inputs of nitrogen. On the basis of current knowledge in this respect, critical loads for nitrogen have been defined for broad classes of ecosystem of different sensitivities (Table 2). Emission reductions are then aimed at lowering deposition rates towards critical loads. This requires a massive mapping exercise, involving the distribution across Europe of ecosystems with different critical loads, deposition rates for nitrogen and emission sources. Then, on the basis of detailed knowledge of the intensity of emission sources on a geographical basis and modelling of long range pollutant distribution, the appropriate controls can be applied. The critical level is defined as the maximum atmospheric concentration of a pollutant that can be tolerated by different broad categories of vegetation over defined averaging times. Current UNECE critical levels for nitrogen gases are shown in Table 3. While critical loads have some theoretical basis, critical levels are based totally on empirical experimental data and are far less developed, but the basic principles involved in their use as a tool for emission control are essentially the same. Plate 2 shows the parts of Great Britain over which the critical load for nitrogen is currently exceeded. Clearly this will require substantial reductions in emissions from industrial, transport and agricultural sources over wide areas in order to achieve the target of protecting sensitive ecosystems and, indeed, permitting recovery of those which have been damaged. In order to give the maximum protection to all ecosystems, much further research is needed on their relative sensitivity because there remain enormous gaps in knowledge.

- In the European Union large reductions in emissions must take place from both large combustion and mobile sources of NO_x.
- Reductions in NH_3 emissions are planned in The Netherlands, by changes in agricultural practice.
- Under the UNECE Convention on Long-Range Transboundary Air Pollution, the Critical Loads and Critical Levels approach is being employed to maximise environmental benefits of emission reductions.

8. FURTHER READING

de Vries, W. & Heij, G.J. (1991). Critical loads and critical levels for the environmental effects of air pollution. In: *Acidification Research in The Netherlands*. (Eds. Heij, G.J. & Schneider, T.). Studies in Environmental Science 46. Elsevier, Amsterdam, pp. 205-214.

Duyzer, J.H., Verhagen, H.L.M., Westrate, J.H. & Bosveld, F.C. (1992). Measurement of the dry deposition flux of NH_3 on to coniferous forest. *Environmental Pollution*, **75**, 3-13.

Ellenberg, H., (1988). *Vegetation Ecology of Central Europe*. Cambridge University Press, Cambridge. (Details on decline of nitrophobic plant species).

Emmet, B.A., Reynolds, B., Stevens, P.A., Norris, D.A., Hughes, S., Gorres, J. & Lubrecht, I. (1993). Nitrate leaching from afforested Welsh catchments - interactions between stand age and nitrogen deposition. *Ambio*, **22**, 386-394.

Falkengren-Gerup, U. & Eriksson, H. (1990). Changes in soil, vegetation and forest yield between 1947 and 1988 on beech and oak sites in southern Sweden. *Forest Ecology Management*, **38**, 37-53.

Fowler, D., Cape, J.N. & Unsworth, M.H. (1989). Deposition of atmospheric pollutants on forests. *Philosophical Transactions of the Royal Society of London. Series B*, **324**, 247-265.

Grennfelt, P. & Thörnelöf, E. (1992). *Critical Loads for Nitrogen*. Nord 41. Nordic Council of Ministers, Copenhagen.

Lewis, O.A.M. (1986). *Plants and Nitrogen*. Edward Arnold, London.

Mattson, W.J. (1980). Herbivory in relation to plant nitrogen content. *Annual Review of Ecology and Systematics*, **11**, 119-61.

Nihlgard, B. (1985). The ammonium hypothesis. An additional explanation to the forest dieback in Europe. *Ambio*, **14**, 2-8.

Pearson, J. & Stewart, G.R. (1993). The deposition of atmospheric ammonia and its effects on plants. *New Phytologist*, **125**, 283-305.

PORG (1990). *Oxides of Nitrogen in the United Kingdom*. Second Report of the U.K. Photochemical Oxidants Review Group, Department of the Environment, London.

Press, M.C., Woodin, J.J. & Lee, J.A. (1986). The potential importance of an increased atmospheric nitrogen supply to the growth of ombrotrophic *Sphagnum* species. *New Phytologist*, **103**, 45-55.

Royal Society (1983). *The Nitrogen Cycle of the United Kingdom.* A Study Group Report. The Royal Society, London.

Schulze, E.D., Lange, O.L. & Oren, R. (Eds.). (1989). *Forest Decline and Air Pollution.* Ecological studies No. 77. Springer Verlag, Berlin.

Sutton, M.A., Fowler, D. & Moncrieff, J.B. (1993). Exchange of atmospheric ammonia with vegetation surfaces: unfertilized vegetation. *Quarterly Journal of the Royal Meteorological Society*, **119**, 1023-45.

TERG (1988). *The Effects of Acid Deposition on the Terrestrial Environment in the United Kingdom.* First Report of the U.K. Terrestrial Effects Review Group, H.M.S.O., London.

van Breeman, N. & Verstraten, J.M. (1991). Soil acidification/N cycling. In: *Acidification Research in the Netherlands.* (Eds. Heij, G.J. & Schneider, T.). Elsevier, Amsterdam, pp. 289-352.

van der Eerden, L.J., Dueck, T.A., Berdowski, J.J.M., Greven, H. & van Dobben, H.F. (1991). Influence of NH_3 and $(NH_4)_2\ SO_4$ on heathland. *Acta Botanica Neerlandica*, **40**, 281-96.

Wellburn, A.R. (1990). Why are atmospheric oxides of nitrogen usually phytotoxic and not alternative fertilizers? *New Phytologist*, **115**, 395-429.

9.APPENDIX

Common and Scientific Names of Species Mentioned in the Text.
Alnus glutinosa - Alder
Arrhenatherum elatius - Oat-grass
Brachypodium pinnatum - Slender false brome
Calluna vulgaris - Heather
Cinclus cinclus - Dipper
Cirsium acaule - Stemless thistle
Cirsium arvense - Creeping thistle
Fagus sylvatica - Beech
Festuca ovina - Sheep's fescue
Festuca rubra - Red fescue
Galium aparine - Goosegrass
Lochmaea suturalis - Heather beetle
Lolium perenne - Perennial ryegrass
Molinia caerulea - Purple moor grass
Nardus stricta - Mat-grass
Picea abies - Norway spruce
Racomitrium lanuginosum - Woolly fringe moss
Scabiosa columbaria - Scabious
Sphagnum spp. - Bog moss
Urtica dioica - Stinging nettle