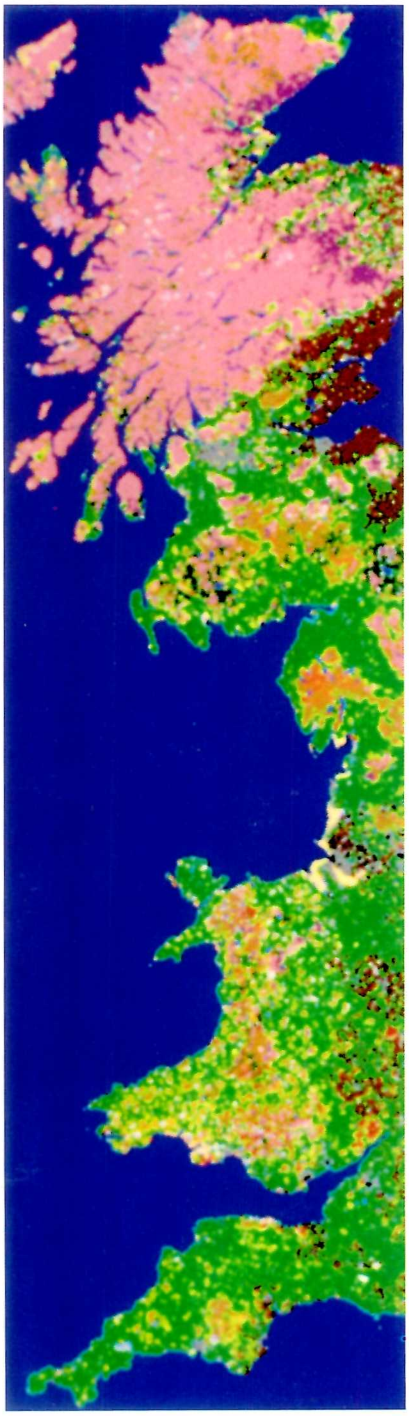
1992 — 1993
REPORT



Institute of Terrestrial Ecology

Natural Environment Research Council

The ITE mission

The Institute of Terrestrial Ecology will develop long-term, multidisciplinary research and exploit new technology (molecular ecology, information technology, and modelling) to understand the science of the natural environment, with particular emphasis on terrestrial ecosystems

Priority is placed on developing and applying knowledge in the following areas

- the factors which determine the *composition, structure,* and *processes* of terrestrial ecosystems, and the *characteristics* of individual plant and animal species
- the dynamics of *interactions* between atmospheric processes, terrestrial ecosystems, soil properties and surface water quality
- the development of a sound scientific basis for *modelling* and *predicting* environmental trends arising from natural and man-made change
- the *dissemination* of this research to decision-makers, particularly those responsible for environmental protection, conservation, and the sustainable use of natural resources

The Institute will provide training of the highest quality, attract commissioned projects, and contribute to international programmes

By these means, ITE will seek to increase scientific knowledge and skills in terrestrial ecology, and contribute to national prosperity and prestige

Front cover illustration
An overview of the ITE land cover map of Great Britain

This Report has been printed on environmentally responsible paper, manufactured from between 60% and 75% bagasse fibre, with the balance from softwood/pine planted, grown and harvested specifically for paper production Bagasse is the vegetable waste remaining after the extraction of sugar from sugar cane, and traditionally has been burnt

The Reminder Printing Company, Ulverston, Cumbria LA12 7EE

Report of the Institute of Terrestrial Ecology 1992–93

Natural Environment Research Council

CONTENTS - Science Reports 3

Environmental pollution **50**

Ecological impacts of the *Braer* incident 52

Ground-level ozone concentrations in the UK 53

Aerial transport of ammonia from agriculture to forest 57

Interactions of atmospheric NH_3 and SO_2 exchange with plant communities 59

Nitrogen saturation in Welsh Sitka spruce forests 62

Effects of excess nitrogen on calcareous grasslands 65

The ecological effects of liming 67

Pesticide interactions 69

Environmental pollution

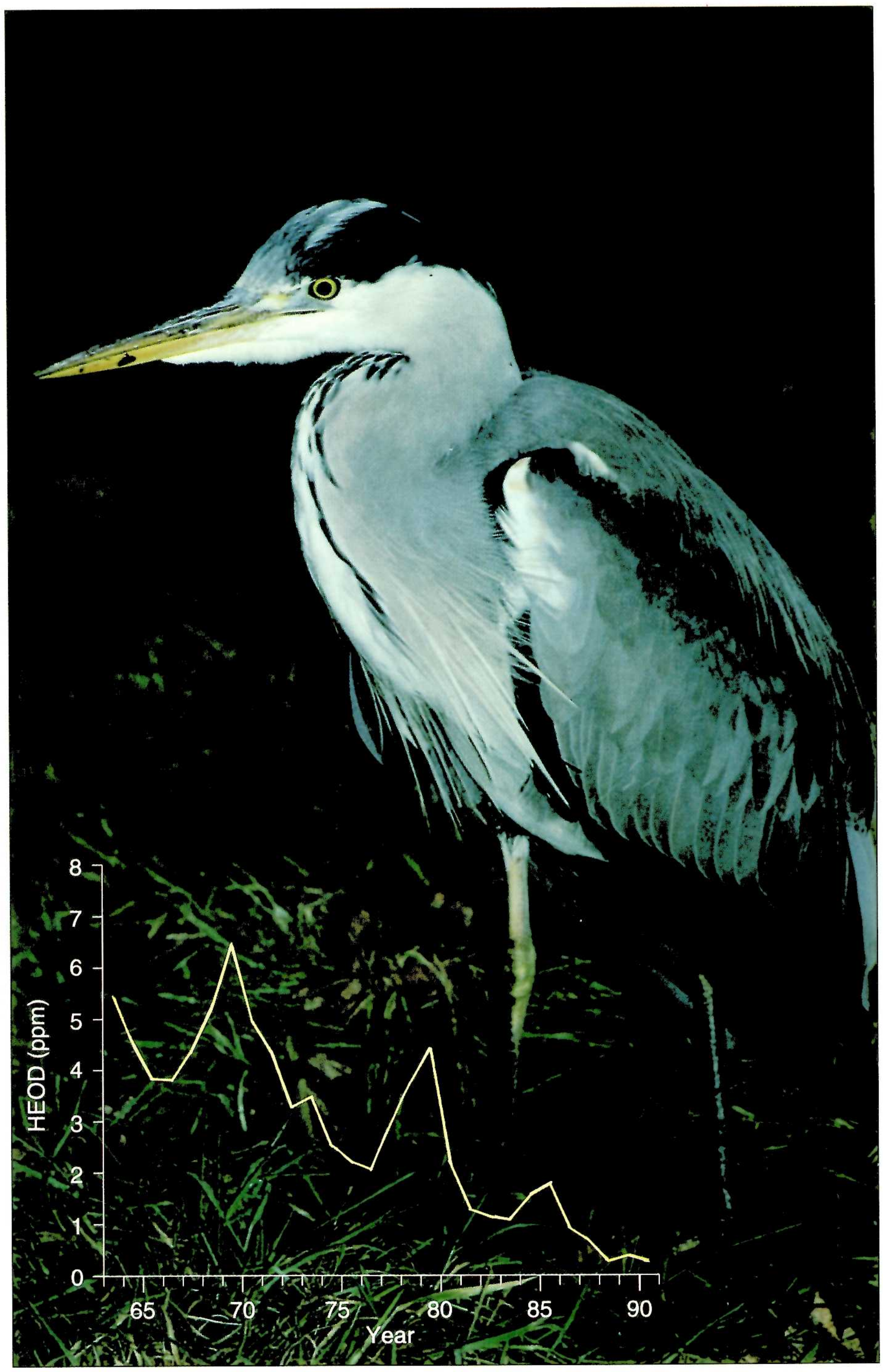


Plate 23. The heron (*Ardea cinerea*), one of the species of bird for which we have long-term data on levels of organic pollutants

Environmental pollution and the impacts of pollutants on our natural fauna and flora, our food and our health are major concerns of individuals, local and national government, and international organisations. The legislation covering the release of potential pollutants and the permitted levels of pollutants in food, soil and water continues to increase. The launch of new chemicals requires a hazard assessment before they can be marketted. To date, such assessments have been based on acute toxicity tests, but there is now a move towards evaluations based on the impacts of the chemicals on populations of organisms. There is also an increasing awareness that there may be synergistic effects of pollutants and that future assessments may need to consider possible interactions. An environmental risk assessment is required as part of the evaluation of plans for new facilities which may emit pollutants or cause a risk to the environment; the bodies responsible for the assessment of such proposals require methods for evaluating the environmental quality of the area affected by the proposal.

Research within the Institute, and centred at Monks Wood, is at the forefront of such hazard and risk assessment; the current work programme includes both basic science and projects linked to specific policy needs. It ranges from the bringing together and evaluation, for international organisations, of existing information on the biological impact of chemical compounds, to the development of new methods of hazard and risk assessment and evaluations of the impacts of specific pollutants on animal and insect populations. Some examples of the current work and its impact are described below.

Reviews and assessments produced for the International Programme on Chemical Safety, and published by the World Health Organisation, are now used as the one of the sources for regulation of chemicals in more than 80% of countries in the world. Evaluation of the ecological impact of the current generation of rodenticides will lead to guidelines which will enable decisions to be made on the most appropriate compound to be used in a given situation. The past year has seen the publication of the analysis of a long-term data set derived from 30 years of measurements of organo-chlorine compounds in birds. The results indicate an increase in the breeding success of species of birds of prey as use of organochlorine pesticides has declined. The synergistic effects of two pesticides have been shown in a study with partridges, and aspects of this study are reported.

Control of transboundary pollution is the subject of international agreements and protocols. A new protocol on the control of emissions of sulphur is currently being negotiated under the auspices of the United Nations Economic Commission for Europe and, for the first time, the critical loads approach will form an integral part of the negotiations. Once agreement has been reached on the sulphur protocol, negotiations will begin on a protocol covering emissions of NO_x; this will involve consideration of NO_x compounds as precursors of ozone Such negotiations require

- information on the current levels of emissions, atmospheric concentrations and deposition of the respective pollutants,
- models for use in assessing the impacts of given emission control scenarios,
- the quantification and mapping of critical loads, or other methods of impact assessment

These inputs to the negotiations require considerable fundamental research to provide the basic understanding of the processes and mechanisms involved Groups at four of ITE's research stations – Bangor, Edinburgh, Merlewood and Monks Wood – are involved in this underpinning research, as well as in determining the pollutant concentrations and fluxes, developing models of pollutant deposition, assessing the likely impacts, developing the critical load concept, and producing critical load maps A large proportion of the work is carried out under contract to government departments, but much of the research is also carried out as part of multi-national groups, often with support from the CEC

During the past year, ITE staff have produced an atmospheric budget for fixed nitrogen over the UK, showing that inputs to the UK are dominated by reduced levels of nitrogen Models have been developed of the enhanced deposition of pollutants which occurs in the remote areas when rainfall passes through lower-level clouds (feeder/seeder models), the results from the models agree well with measured deposition The first national map of surface ozone concentrations for any country has been produced for the UK, and is described on pages 53–57 Other reports on impacts of excess nitrogen on calcareous grasslands and on critical loads for Sitka spruce plantations illustrate work which is contributing to the definition and quantification of critical loads In contrast to the situation with sulphur, agriculture is a major source of pollutant nitrogen, and the reports on pages 57–62 reflect some of our current work on this topic

In some instances, reductions in pollutant deposition alone will not result in the recovery of terrestrial or aquatic ecosystems In these cases, it may be necessary to use ameliorative measures, such as liming However, such measures can themselves result in damage to the treated systems A further aspect of the Institute's work on the effects of acidic deposition is examining the ecological impacts of liming as an ameliorative treatment, and is described on pages 67–69

One major area of the Institute's research on environmental pollution, that on radioecology, is not represented in this year's Annual Report Continuing concerns in this area of science are the likely duration of the effects of the Chernobyl deposition, in terms of restrictions on the sale of sheep in upland Britain, the factors controlling the availability to plants of the deposited caesium, its mobility in different plant/soil systems and its transfers to natural and managed populations of animals, and the variation in activity levels within the sheep flocks of the restricted areas The short-term aim of such research is to provide methods of ameliorating the effects of the deposition, while the medium-term aim is to develop improved models of caesium transfers in soil/plant/animal systems for use in predicting the impact of any future nuclear accident Results during the past year have shown that the area in which individual sheep graze is a major factor in determining why some sheep in restricted flocks have consistently higher radiocaesium levels Studies on the behaviour of radiocaesium in heather-dominated soil/plant systems have shown that 30% of the original Chernobyl deposition is still retained in the top 5 cm of a heather/peat system, compared with 70% in a heather/podzol system A model of caesium behaviour in soil/plant/sheep systems has been developed in collaboration with the University of Nottingham, and has been used to assess the impact of changes in pasture management on the caesium activity of grazing sheep The work on caesium behaviour in semi-natural ecosystems has highlighted the weaknesses in our understanding of the behaviour of other radionuclides which can be released following nuclear accidents Studies to overcome these weaknesses are now being started

The research on hazard and risk assessment, on atmospheric pollution, and on radioecology forms part of ongoing programmes of work which are addressing medium- to long-term problems The report which follows provides an example of the Institute's ability to respond to a specific pollution incident, the oil spill on the Shetlands The Institute was able quickly to bring together a multidisciplinary group of scientists to assess the short- and potential long-term impacts of the spill on bird and animal life, vegetation and soils The data bases built up from earlier and ongoing work on the Shetland Islands by Institute staff provided a baseline for the work

M Hornung

Ecological impacts of the *Braer* incident

Introduction

The grounding of the tanker *Braer* on Garths Ness, Shetland (Plate 24), on 5 January 1993 was greeted with concern, not only by the islands' population, but also by the conservation community in general. With 84 000 tonnes of crude on board, and memories of the *Exxon Valdez* still fresh in many people's minds, the scenario of a major ecological disaster developed. With each day, the sea pounded the vessel and more of its cargo escaped into the sea. Within hours of the stranding, the first oiled birds were being collected from the shores of Quendale Bay, and reports of both otters (*Lutra lutra*) and seals (predominantly *Halichoerus grypus*) being seen in oiled waters were being logged with the reception centre.

Shetland is rightly recognised for its rich and varied wildlife, in particular its seabirds, otters and seals, the populations of some being of international and national importance. As the oil moved northwards along the west coast of the mainland, more and more animals were exposed to the threat of pollution.

Staff of ITE, particularly those based at Banchory, have for many years been carrying out research in Shetland (on otters, seabirds and vegetation) and their expertise was quickly in demand for both practical advice and to answer the many questions posed by media.

The nature of the spill

The *Braer* incident was reckoned to be the world's fifth largest oil spill, double that of the *Exxon Valdez*. The potential for a major environmental disaster was therefore immense. Because of a combination of factors, in particular the Gulfaks crude being very light and therefore extremely volatile (and toxic), and the prolonged period of extremely rough weather, relatively little oil actually impacted on the beaches; most of it was dispersed into the water column instead, and several thousand tonnes apparently penetrated the seabed sediments. One unexpected event was the amount of oil which carried inland from the coast, driven by the gale-force winds, contaminating both farmland and semi-natural vegetation – sand dunes, maritime grasslands and moors.

Plate 24. View of the *Braer* wreck

Immediate impact on animals

From the point of view of wildlife, the immediate impact of the spill was perceived as not being great. A total of 235 birds were taken into care for cleaning and rehabilitation by members of the Scottish Society for the Protection of Animals (SSPCA). Less than 1600 birds of 23 species were recovered dead from the beaches. More than 50% of the dead birds were shags (*Phalacrocorax aristotelis*), and 13% black guillemots (*Cepphus grylle*). Of birds wintering on Shetland, the most severely affected were long-tailed ducks (*Clangula hyemalis*) (96 dead) and great northern divers (*Gavia immer*) (13 dead).

Extensive searches of beaches for oiled animals only recovered six otters in the first two weeks following the *Braer* hitting the rocks. Two, both young cubs deserted by their parents, were taken into care and have responded well to treatment. The remaining four were adults in poor condition, a feature common in otters at this time of the year, when food is normally difficult to catch, and exacerbated by the rough weather. One was recovered alive, but died in care, the remainder were found dead, two being hit by cars. None of the deaths could be attributed directly to the current spill.

Impacts on terrestrial communities

Following the wreck, ITE undertook a programme of soil and vegetation sampling in the Garths Ness to Scatness area along a series of transects inland from the coast. The transects covered four main semi-natural vegetation types:

- maritime grassland;
- heather (*Calluna vulgaris*) grassland;
- red fescue (*Festuca rubra*) dune grassland;
- mat-grass (*Nardus stricta*) grassland.

The aims of the work were to:

i. assess the pollution burden of sample soil and vegetation, and relate it to distance from source;

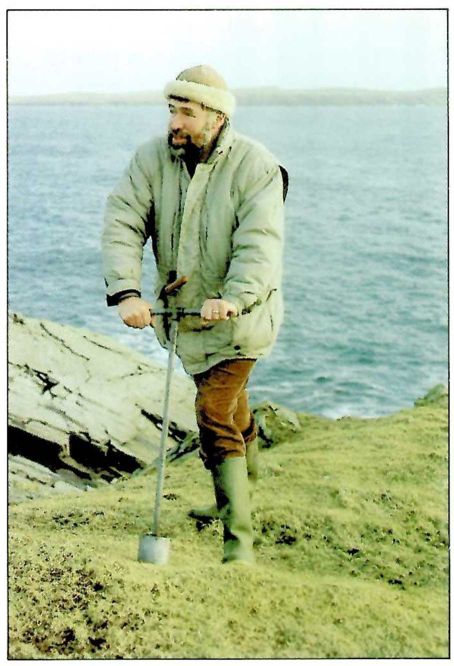


Plate 25. ITE scientist cutting soil cores

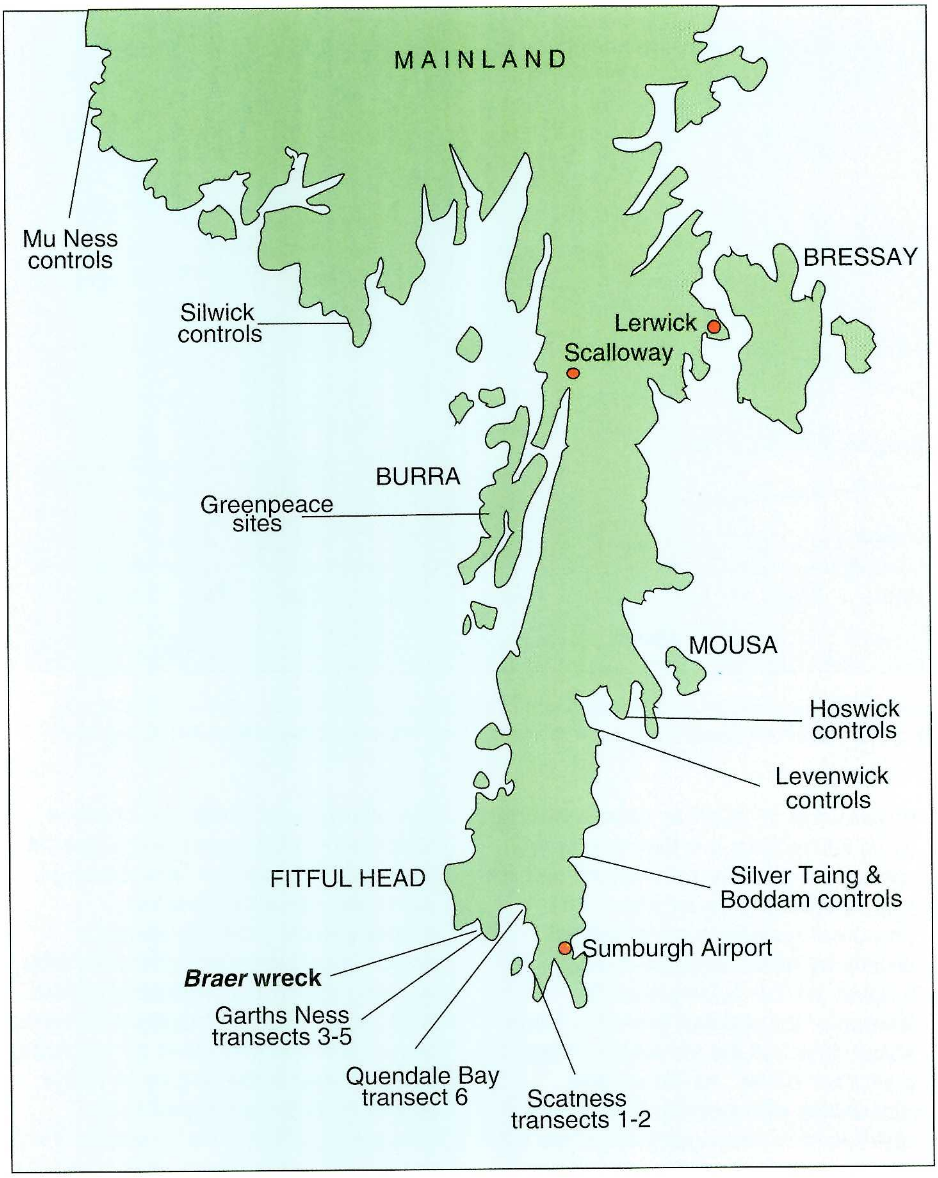


Figure 32. Sites used for sampling vegetation and soils following the *Braer* incident. The wreck occurred in the south-west corner of the mainland, and was sampled by transects at Garths Ness, Quendale and Scatness. The remaining sites were controls

ii. record the extent and type of damage to vegetation;

iii. examine recovery growth;

iv. compare numbers and species of invertebrates hatching from contaminated and uncontaminated cores;

v. assess the effects of the pollution on soil enchytraeid and earthworm populations.

The sampling comprised the cutting of soil and vegetation cores at six polluted sites, and at control areas on both the east and west coasts (Plate 25). Figure 32 indicates the sample locations. Cores of 10 cm in diameter were cut and used for:

- chemical analysis of petrogenic hybrocarbons;
- bioassay of vegetation growth;
- bioassay of emerging invertebrates;
- estimation of enchytraeid and earthworm numbers.

Preliminary results indicate that there has been scorching of vegetation by oil near the coast, with some evergreen species, such as heather, particularly affected. Recovery growth has generally been vigorous, although some species (notably plantain (*Plantago*) and dock (*Rumex*) species) appear to be making more growth than others.

Studies of soil invertebrates so far indicate no significant impact of the pollution on numbers or depth distribution within sample cores.

Longer-term effects

The short-term impacts of the wreck suggest that the severity of the disaster was much less than might have been feared. It will be important, however, to examine the longer-term effects. In particular, ITE has initiated research to determine effects on otter populations, and on inshore fish which form their food. There are indications of substantial falls in fish populations along polluted shores. Scientists at ITE Banchory have begun a detailed study of fish populations at selected locations, polluted and unpolluted, and including Lunna Ness where extensive research on otters, their diet and populations, was conducted in the 1980s.

In 1989, a census of otters on the islands indicated a population of about 700–900 adults. The area of Shetland affected by oil from the *Braer* contained about 10% of the total otter population. The 1989 census will be repeated later this year, and population and reproductive success will be compared for affected and unaffected coasts.

There will also be further study of impacts on vegetation and soil invertebrate communities. All these investigations will complement studies by other organisations, such as the Plymouth Marine Laboratory on offshore fish, the Macaulay Land Use Research Institute and Scottish Agricultural Colleges on impacts on agriculture, and the Royal Botanic Garden, Edinburgh, on lichens.

N G Bayfield and J W H Conroy

Ground-level ozone concentrations in the UK

(This work was funded by the Department of the Environment)

Interest in atmospheric ozone during the last decade has been concentrated on the stratospheric zone at altitudes above 10 km and its depletion as a consequence of chlorofluorocarbon emissions. Ozone is also present in the troposphere (approx 0–10 km) and in the planetary boundary layer (the lowest 1 km) at concentrations in the range 10–200 parts per billion (10^9) by volume (ppbV). The presence of ozone in surface air is not new, and terrestrial

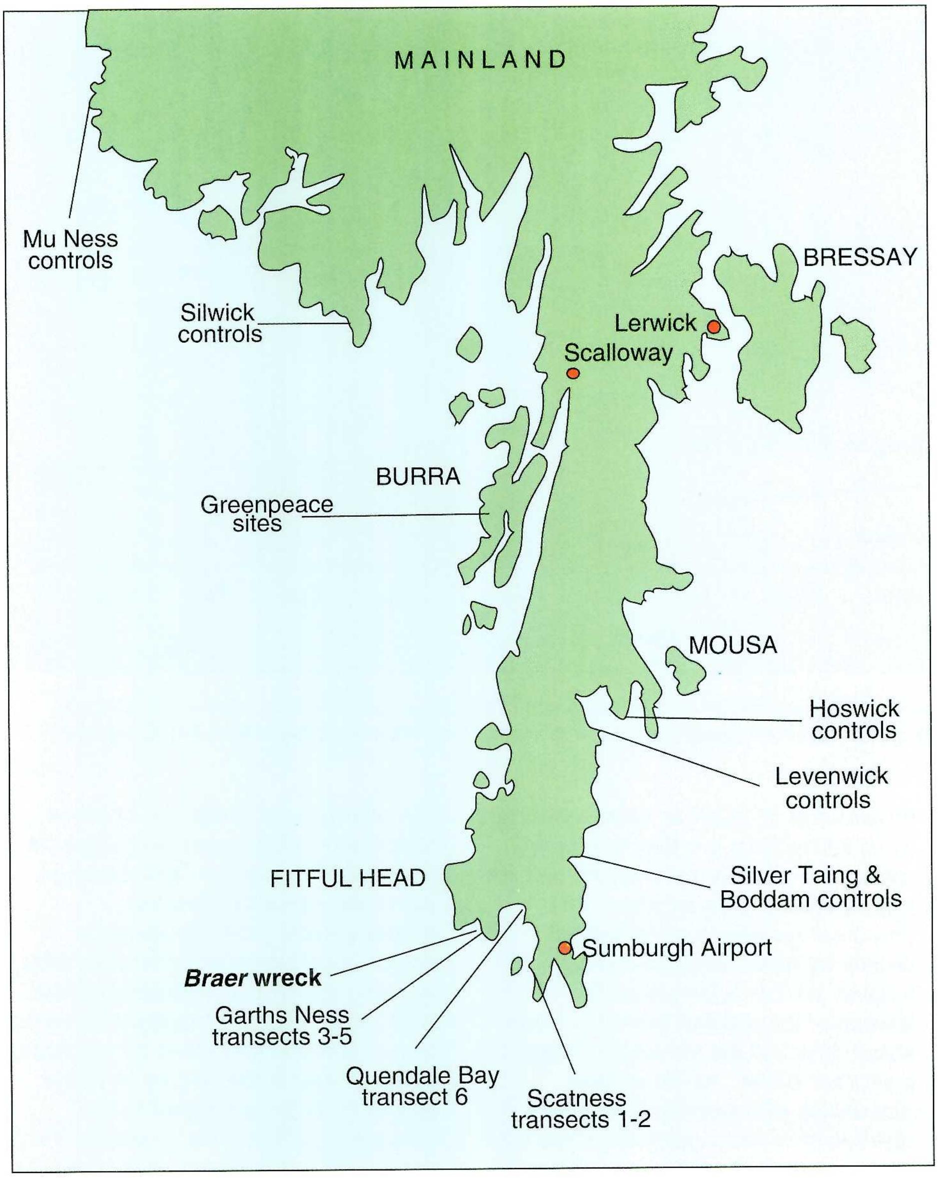


Figure 32. Sites used for sampling vegetation and soils following the *Braer* incident. The wreck occurred in the south-west corner of the mainland, and was sampled by transects at Garths Ness, Quendale and Scatness. The remaining sites were controls

ii. record the extent and type of damage to vegetation;

iii. examine recovery growth;

iv. compare numbers and species of invertebrates hatching from contaminated and uncontaminated cores;

v. assess the effects of the pollution on soil enchytraeid and earthworm populations.

The sampling comprised the cutting of soil and vegetation cores at six polluted sites, and at control areas on both the east and west coasts (Plate 25). Figure 32 indicates the sample locations. Cores of 10 cm in diameter were cut and used for:

- chemical analysis of petrogenic hybrocarbons;
- bioassay of vegetation growth;
- bioassay of emerging invertebrates;
- estimation of enchytraeid and earthworm numbers.

Preliminary results indicate that there has been scorching of vegetation by oil near the coast, with some evergreen species, such as heather, particularly affected. Recovery growth has generally been vigorous, although some species (notably plantain (*Plantago*) and dock (*Rumex*) species) appear to be making more growth than others.

Studies of soil invertebrates so far indicate no significant impact of the pollution on numbers or depth distribution within sample cores.

Longer-term effects

The short-term impacts of the wreck suggest that the severity of the disaster was much less than might have been feared. It will be important, however, to examine the longer-term effects. In particular, ITE has initiated research to determine effects on otter populations, and on inshore fish which form their food. There are indications of substantial falls in fish populations along polluted shores. Scientists at ITE Banchory have begun a detailed study of fish populations at selected locations, polluted and unpolluted, and including Lunna Ness where extensive research on otters, their diet and populations, was conducted in the 1980s.

In 1989, a census of otters on the islands indicated a population of about 700–900 adults. The area of Shetland affected by oil from the *Braer* contained about 10% of the total otter population. The 1989 census will be repeated later this year, and population and reproductive success will be compared for affected and unaffected coasts.

There will also be further study of impacts on vegetation and soil invertebrate communities. All these investigations will complement studies by other organisations, such as the Plymouth Marine Laboratory on offshore fish, the Macaulay Land Use Research Institute and Scottish Agricultural Colleges on impacts on agriculture, and the Royal Botanic Garden, Edinburgh, on lichens.

N G Bayfield and J W H Conroy

Ground-level ozone concentrations in the UK

(This work was funded by the Department of the Environment)

Interest in atmospheric ozone during the last decade has been concentrated on the stratospheric zone at altitudes above 10 km and its depletion as a consequence of chlorofluorocarbon emissions. Ozone is also present in the troposphere (approx 0–10 km) and in the planetary boundary layer (the lowest 1 km) at concentrations in the range 10–200 parts per billion (10^9) by volume (ppbV). The presence of ozone in surface air is not new, and terrestrial then be used to calculate the correct spatial distribution of in-summer mean ozone concentration. The resulting map (Figure 36) combines the effects of altitude and exposure with regional patterns of ozone concentration, and shows the largest summer mean values in the uplands of Wales and southern England. Additional areas of high concentrations occur elsewhere in upland areas of northern and western Britain.

However, most concern over ground-level ozone lies not with the mean values but with the episodes during which concentrations are reached that are toxic to plants or to man. Such thresholds differ between species and with environmental conditions. For vegetation, some sensitive plant species show visible or physiological effects following exposure to 40 or 50 ppbV, ie within a factor of two of the mean concentration at some sites. Choosing a threshold which correctly quantifies the potential risks from ozone effects and its geographical distribution is therefore a difficult task.

We have selected a threshold of 60 ppbV, as it is a concentration above which crop yields have been shown to be reduced and for which the number of episode hours throughout the UK are large enough to be suitable for the mapping tasks.

Topography influences the duration of episodes in much the same way as mean concentrations are increased by altitude. A similar procedure may, therefore, be used to map the episode hours for the UK as has been used for mean concentrations. An underlying surface of episode hours for the well-mixed part of the day is increased in areas of high ground, according to the following relation:

$$h_{60} = 1.3 + 2.1 \times 10^{-3} z \times t_{60}$$

where h_{60} is the exposure (in hours) to ozone concentrations in excess of 40 ppb; z is the altitude (in metres) of the location; and t_{60} is the duration of ozone concentration in excess of 40 ppbV during the well-mixed part of the day (1200–1800 GMT).

The resulting map (Figure 37) shows substantial areas of southern England and Wales with 700–900 hours per year of O_3 concentration in excess of

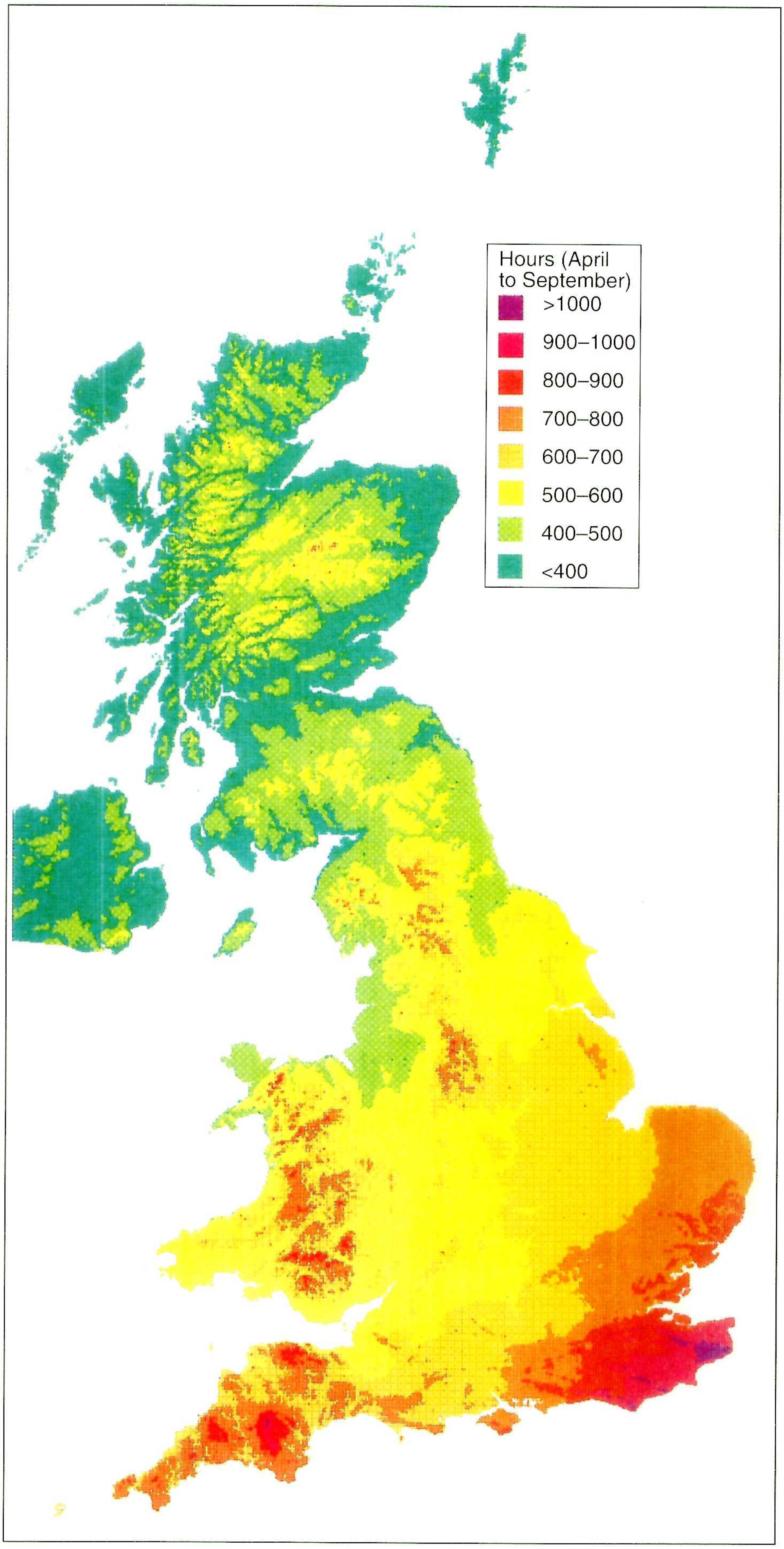


Figure 37. Geographical distribution of ozone exposure hours exceeding 40 ppb throughout the UK

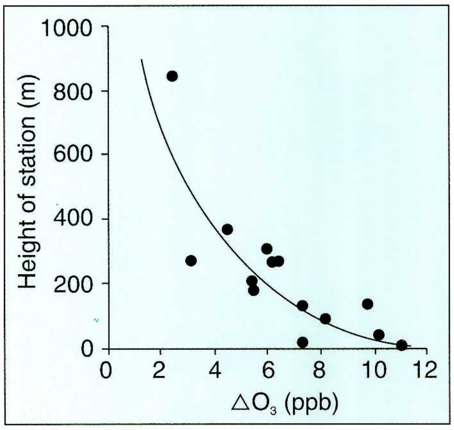


Figure 35. The relationship between height of sampling station and the difference in concentration between daytime 'well-mixed' period (10.00–18.00 GMT) and the daily mean

continuously. In contrast, sheltered valley locations are only well connected during windy weather or when strong solar heating of the ground generates mixing by convective processes. For the rest of the time and particularly at night and in winter, a strong stable stratification of the air near the ground isolates the surface air from the bulk of the atmosphere. In these conditions, ozone concentrations near the ground decline as ozone is deposited on to the ground more rapidly than its supply from higher levels. These processes lead to a marked diurnal cycle in ozone concentrations at sheltered sites, as illustrated in Figure 34. The amplitude of the diurnal cycle at the top of Great Dun Fell is typically 2–5 ppb, whereas at a valley site (Wharleycroft) it is 20–30 ppb and, of course, strongly dependent on the weather. In the six summer months (April–September) and during the part of the day when solar radiation provides good mixing of the boundary layer even at small windspeeds (10.00–18.00 GMT), the topographic effects on ozone are very small, and regional (100–200 km) patterns in ozone concentration may be mapped from the relatively well-mixed part of the day. The topographic effects may then be superimposed on this underlying structure using the relationship between altitude and the magnitude of the diurnal cycle in ozone concentration which has the form:

$$O_3 = -1.71 - 8.4e^{-3z}$$

where O_3 is the difference in concentration between the daytime well-mixed period (10.00–18.00 GMT) and the daily mean; and z is the altitude (in metres) of the monitoring station (Figure 35). A topographical map of the UK can

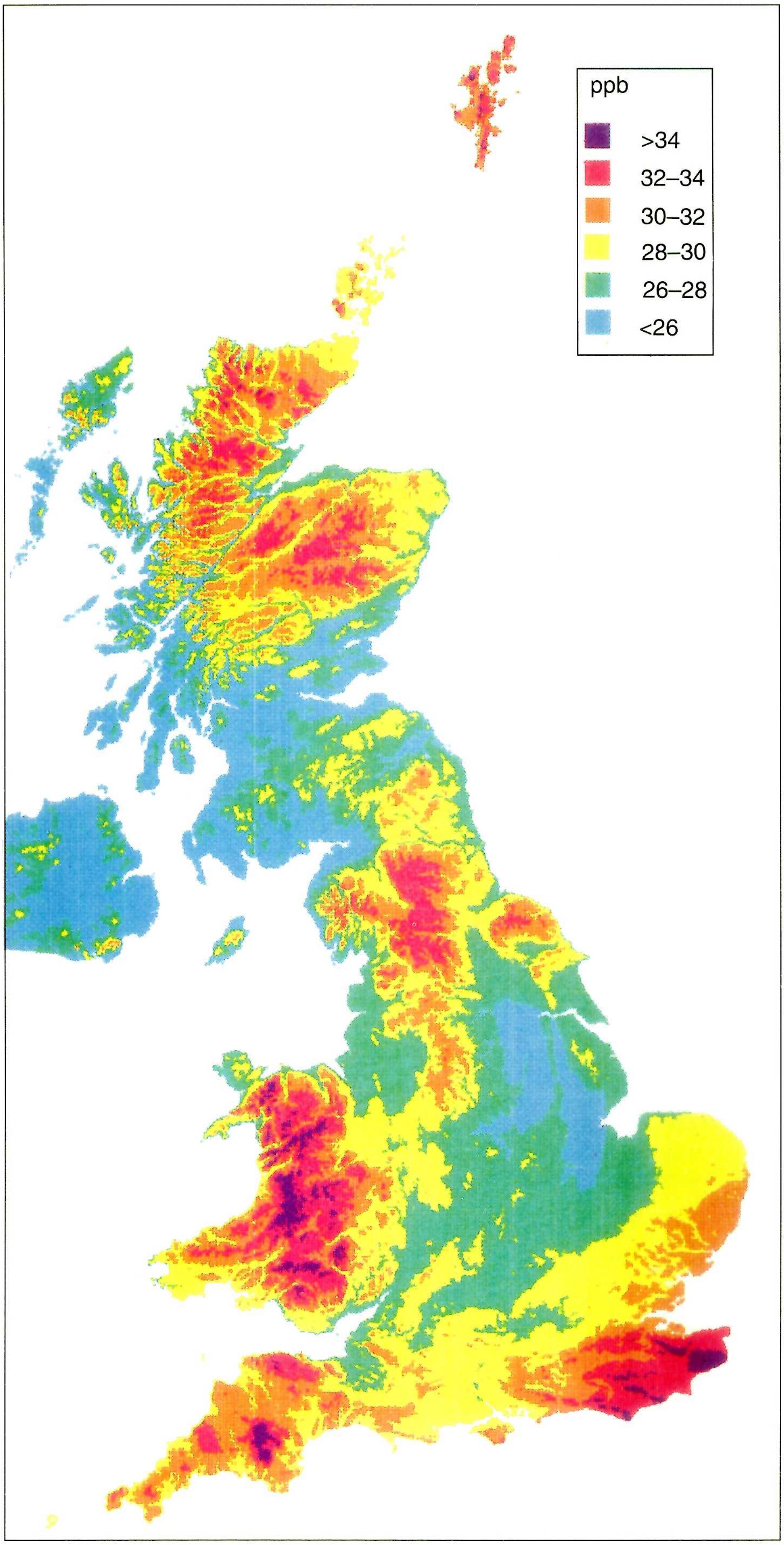


Figure 36. Summer mean ozone concentrations in the UK

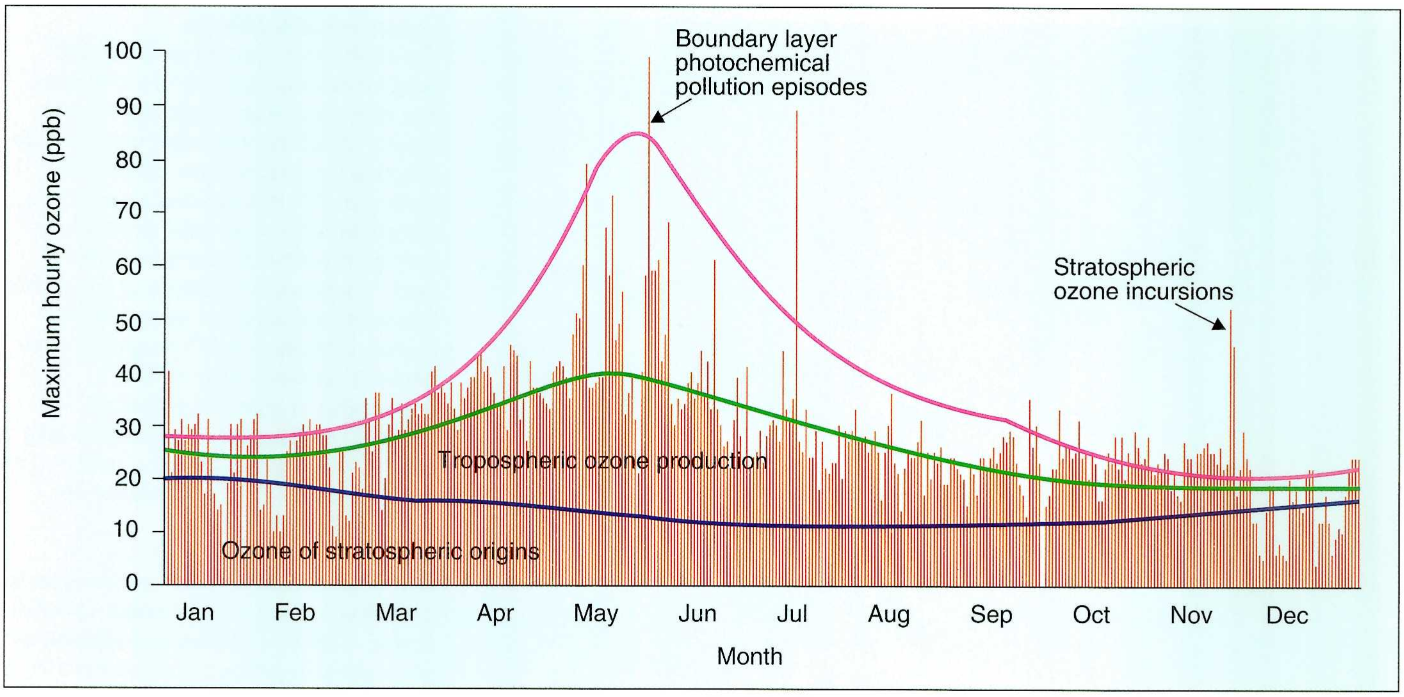


Figure 33. The maximum hourly average ozone concentration for each day of 1978, measured 2 m above a Scots pine (*Pinus sylvestris*) canopy in Devilla Forest, central Scotland, showing the annual cycle in maximum concentration and the occurrence of photochemical episodes during which concentrations exceed 60 ppb

vegetation evolved in the presence of small ozone concentrations. The earliest measurements of ozone in surface air during the second half of the last century at several sites in Europe indicated mean concentrations in the range 10–15 ppbV. Current measurements at the same sites show mean ozone concentrations about a factor of two larger (20–30 ppbV). The large increase in mean concentrations is accompanied by episodes during which concentrations reach peaks of between 100 and 200 ppbV. These episodes occur when the precursor gases for photochemical ozone production (oxides of nitrogen NO and NO_2 and volatile organic compounds, VOC) are present in suitable meteorological conditions for the chemical processing to occur. The ideal conditions are hot summer days with surface air temperatures in excess of 20°C, high solar radiation levels (clear skies), and low windspeeds to reduce the dispersion of the reactants.

The data plotted in Figure 33 show the maximum hourly mean ozone concentration each day for a year at a site in central Scotland. These data illustrate the annual cycle in concentrations, with a spring maximum which is commonly observed in northern Europe. Also clearly illustrated are the occasional days during which peak concentrations reach 100 ppbV, the episode days. Taking an arbitrary threshold of 60 ppbV to define episode days, it is evident that they are largely spring and summer phenomena and are closely linked to the weather. The precursor emissions are provided largely by motor vehicles and by industry so that episodes occur whenever the weather provides warm sunny days and the air contains these precursor gases. As the source distribution of emissions is unevenly distributed in the country and there are also pronounced gradients in climate across the country, then the conditions favouring photochemical oxidant production also vary.

A network of ozone monitoring stations was established by the Department of the Environment during the late 1980s. These stations have provided the data necessary to describe the major features of surface ozone concentrations in the UK. A detailed description of the methodology, sites and data is provided by the third report, in 1993, of the Photochemical Oxidants Review Group, set up by the Department of the Environment. One of the major achievements of the Review Group has been the provision of detailed maps of summer mean surface ozone concentrations and of episode frequency throughout the UK to identify the ecosystems and populations at risk from the effects of photochemical oxidant episodes.

The relatively small number of ozone monitoring stations operating in the UK (currently 15), together with the large site-to-site variability in mean concentrations, preclude simple interpolation procedures between sites to obtain the spatial concentration field. Much of the inter-site variability in mean concentration is generated by variability between sites in the degree to which ground level air is coupled to the atmosphere above. For example, windy, exposed hill-tops remain well coupled to the bulk of the atmosphere almost

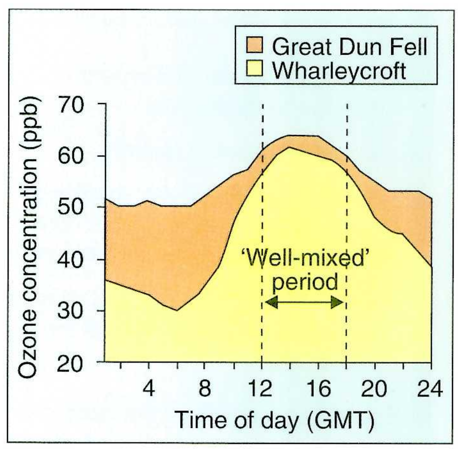


Figure 34. Ambient ozone concentrations measured at a sheltered valley site (Wharleycroft) at 200 m asl and at a windy exposed hilltop (Great Dun Fell) at 847 m asl, showing the large differences in diurnal cycle at exposed and sheltered locations

40 ppb, while in the uplands of Scotland the duration of such concentrations is smaller with between 400 and 600 hours.

These regional distributions of ozone are particularly valuable in assessing areas of vegetation or the population at risk from the effects of ozone. Recent studies of the human health effects of ozone and related photochemical oxidants suggest that this aspect of concern is likely to grow and may in time be the main argument for emission controls of the major precursor gases.

D Fowler and R I Smith

Aerial transport of ammonia from agriculture to forest

(This work was partly funded by the Department of the Environment)

There is an increasing realisation of the importance of ammonia (NH_3) emissions and their role in acid and excess nitrogen (N) deposition to terrestrial ecosystems. Emissions in certain European countries may well have doubled since 1950. Agriculture is responsible for about 95% of the anthropogenic NH_3 emissions over Europe, with the major contribution coming from animal waste. Particularly affected are countries in northern Europe, including The Netherlands, Denmark, Belgium and northern Germany, yet we have very few data on the current situation in the UK.

Although ammonia is able to neutralise acids in the atmosphere, far from

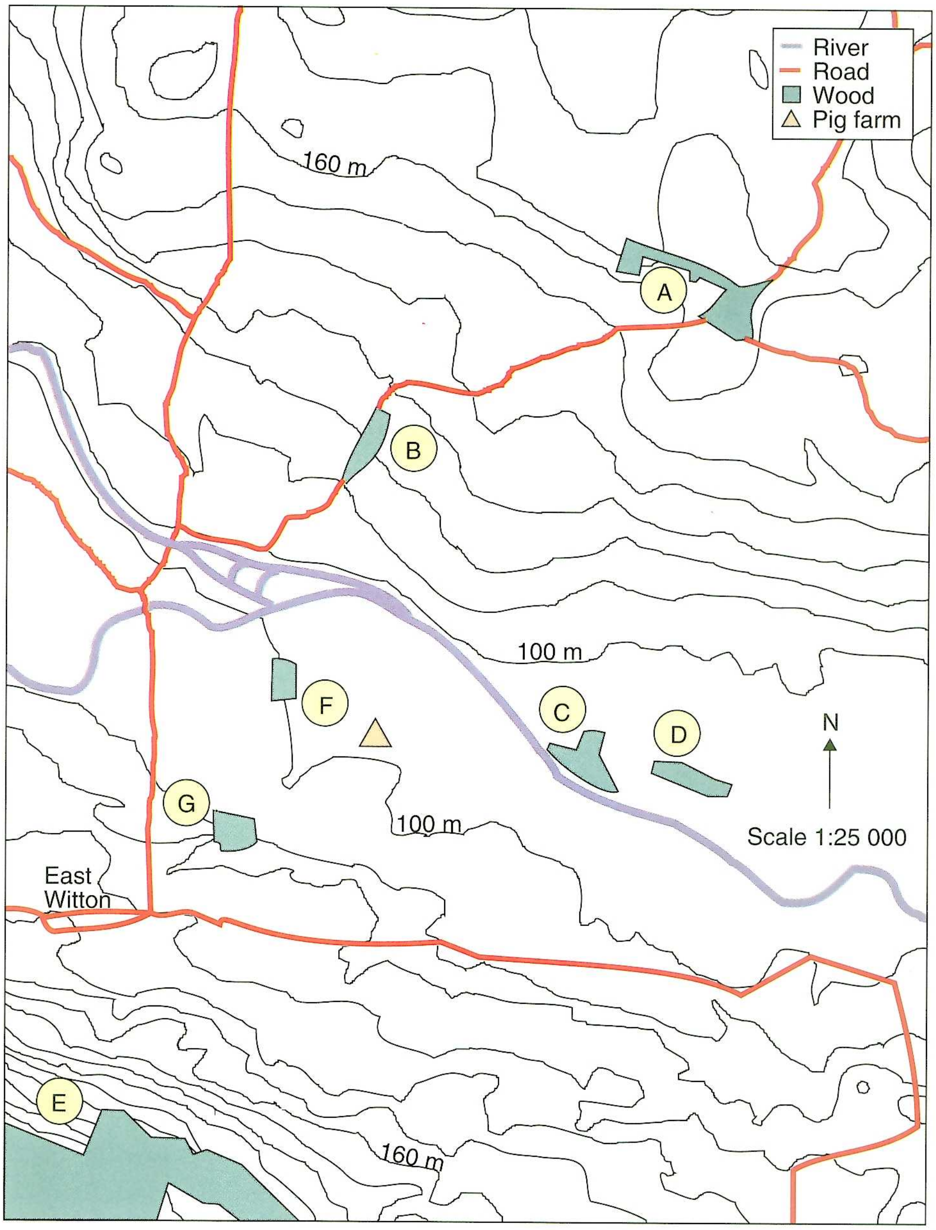


Figure 38. Map showing the location of the study area, and the forests being monitored. The prevailing wind direction is typically from west to east

Plate 26. A view across the valley under study, as seen from the southern side. The pig farm is located at the centre of the photograph and some of the forest sites can also be seen (see Figure 38)

ameliorating against acid deposition to forest ecosystems, increased deposition of SO_4 appears to occur in areas with high NH_3; NH_3 appears to promote the deposition of sulphur dioxide (see Sutton & Fowler, pp59–62). Increased NH_3 deposition may lead to saturation with N in forest ecosystems, and can be a contributory factor to forest dieback in some regions. Increased acidification of forest soils due to increased nitrification, and the disturbance of the nutrient balance in tree leaves due to increased nitrogen uptake are two of the mechanisms by which damage may occur.

The landscape in the UK is characterised by an intimate mixture of agriculture and

40 ppb, while in the uplands of Scotland the duration of such concentrations is smaller with between 400 and 600 hours.

These regional distributions of ozone are particularly valuable in assessing areas of vegetation or the population at risk from the effects of ozone. Recent studies of the human health effects of ozone and related photochemical oxidants suggest that this aspect of concern is likely to grow and may in time be the main argument for emission controls of the major precursor gases.

D Fowler and R I Smith

Aerial transport of ammonia from agriculture to forest

(This work was partly funded by the Department of the Environment)

There is an increasing realisation of the importance of ammonia (NH_3) emissions and their role in acid and excess nitrogen (N) deposition to terrestrial ecosystems. Emissions in certain European countries may well have doubled since 1950. Agriculture is responsible for about 95% of the anthropogenic NH_3 emissions over Europe, with the major contribution coming from animal waste. Particularly affected are countries in northern Europe, including The Netherlands, Denmark, Belgium and northern Germany, yet we have very few data on the current situation in the UK.

Although ammonia is able to neutralise acids in the atmosphere, far from

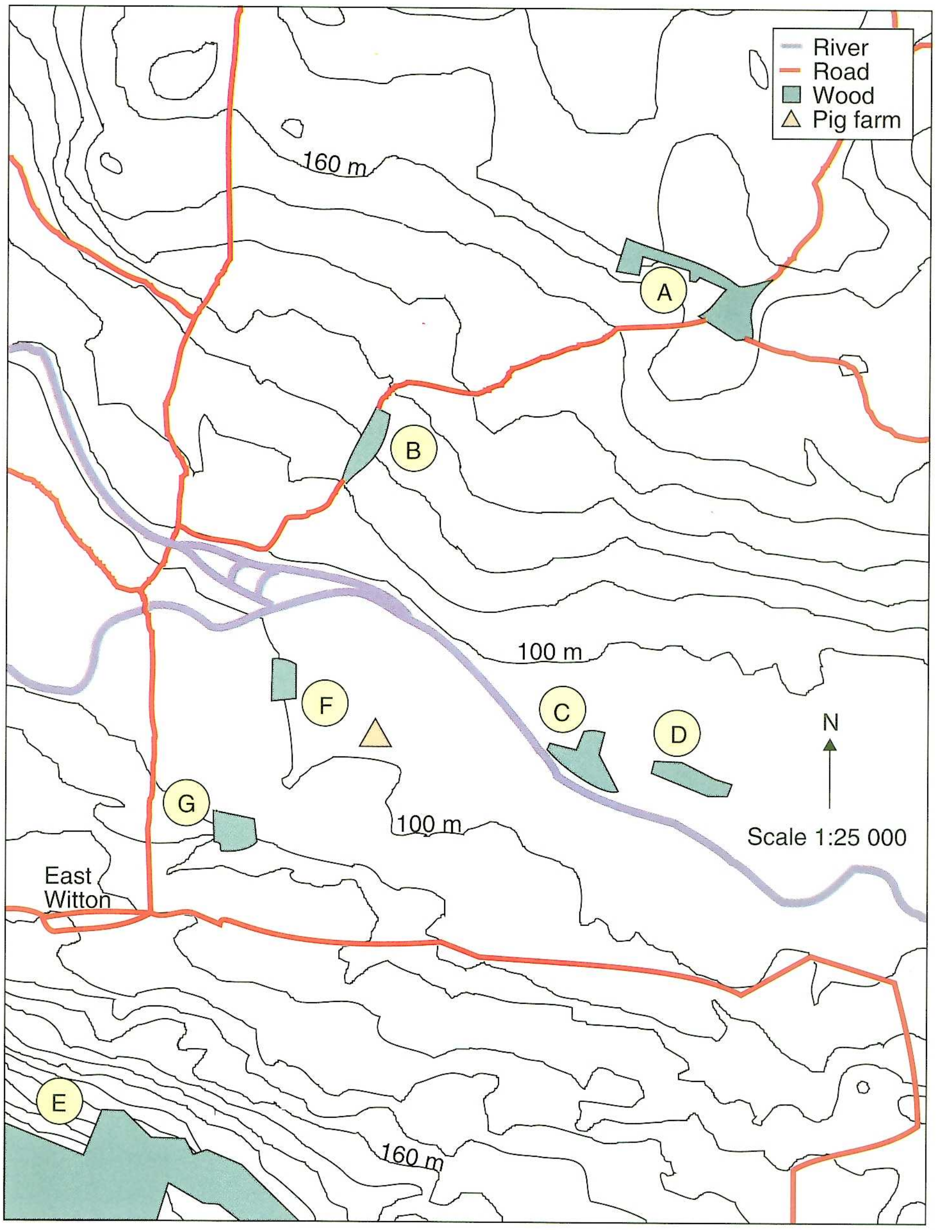


Figure 38. Map showing the location of the study area, and the forests being monitored. The prevailing wind direction is typically from west to east

Plate 26. A view across the valley under study, as seen from the southern side. The pig farm is located at the centre of the photograph and some of the forest sites can also be seen (see Figure 38)

ameliorating against acid deposition to forest ecosystems, increased deposition of SO_4 appears to occur in areas with high NH_3; NH_3 appears to promote the deposition of sulphur dioxide (see Sutton & Fowler, pp59–62). Increased NH_3 deposition may lead to saturation with N in forest ecosystems, and can be a contributory factor to forest dieback in some regions. Increased acidification of forest soils due to increased nitrification, and the disturbance of the nutrient balance in tree leaves due to increased nitrogen uptake are two of the mechanisms by which damage may occur.

The landscape in the UK is characterised by an intimate mixture of agriculture and

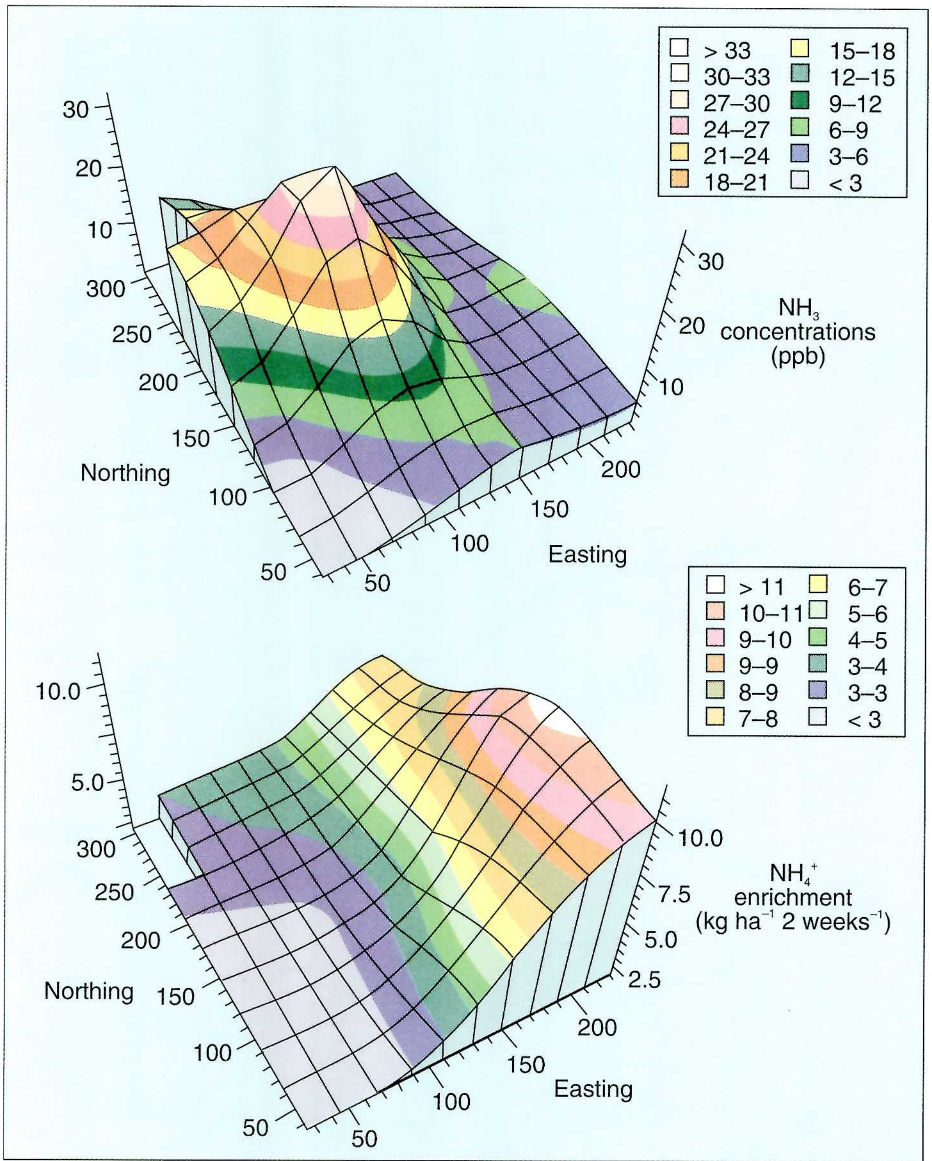


Figure 39. Surface plots showing (i) NH_3 concentrations (ppb) and (ii) NH_4^+ enrichment (kg N ha^{-1} 2 $weeks^{-1}$) beneath the canopy for one week of the study. The peak in NH_3 concentration lies immediately above the pig farm. The plots are extrapolated from only a few measurement points but clearly show the trends observed during the course of the study

woodland/forest, providing the ideal conditions for the potential aerial transport of NH_3 between agriculture and natural/semi-natural ecosystems. However, very little attention has been paid to N deposition to forest ecosystems from agricultural sources in the UK, and the aim of the current work was to investigate the extent to which forests in agricultural areas may be subject to high N deposition. We have also investigated the spatial distribution of NH_3 concentrations in the air around an agricultural point source and looked for evidence of the co-deposition of NH_3 and SO_2.

Site description

In order to investigate N deposition around a single identifiable agricultural point source of NH_3, a site was selected in lower Wensleydale in the Yorkshire Dales. Wensleydale is a valley which runs from west to east, with a wide valley bottom and asymmetric slopes (Plate 26; Figure 38). Prevailing winds are predominantly from the south-west, with an annual rainfall and temperature of approximately 700 mm and 8.5°C, respectively.

An area of 3 km x 4 km around a pig farm on the valley bottom was chosen for the current monitoring programmme, with six Norway spruce (*Picea abies*) forests and one Scots pine (*Pinus sylvestris*) forest being chosen for study at selected locations around the farm.

The seven sites are referred to as sites A to F, with A, C and D located downwind of the farm and the remainder upwind. Sites C, D and F were located in the valley bottom; A and E were located on the top of the valley sides (see Figure 38). Site B was situated only 60 m away from a small low-intensive farm, and the monitoring of the rainfall and NH_3 concentation in the air took place very near to the farm buildings. The wood at this site was surrounded by grassland which was grazed by sheep and cattle during the summer. In winter the animals were brought on to the farm, and the ewes were also kept inside during spring for lambing.

NH_3 concentration in the air

Four passive diffusion tubes, one of which was used as a blank, were used to measure the NH_3 concentration in the air at each of the seven sites. The tubes were fastened to wooden posts 1 m above the ground, placed within 50 m of the edge of each of the forests. Each tube was fitted with a disc of glass-fibre filter paper coated with diluted sulphuric acid (an absorbant for NH_3), which was held in place at the upper end of the tube by a polyethylene cap. NH_3 in the air was converted to ammonium ions (in the ratio 1:1) by reaction with the sulphuric acid, as described by Hargreaves (1989).

The data from one week of sampling are presented in Figure 39 in the form of a contour surface. Although there were too few sampling points to construct a precise 3D image of NH_3 concentrations in the air in the valley, Figure 39 conveys well the types of results obtained during the course of the study, and the importance of the pig farm in dominating air quality in the valley.

Analysis of variance divided the sites into two groups: the high-concentration sites B and F had mean ammonia concentrations of 29.3 ppb and 24.0 ppb respectively, and the low-concentration sites, which comprised the remainder, had concentrations between 4 and 11 ppb. The low-concentration sites showed a clear seasonal pattern, with maximum concentrations in spring and autumn and lowest concentrations in winter. The air concentrations of NH_3 at site F also showed a very similar seasonal pattern, with maximum concentrations also found in spring, late summer and autumn, reaching levels up to 120 ppb after the spreading of manure. In contrast to all the other sites, the concentrations at site B

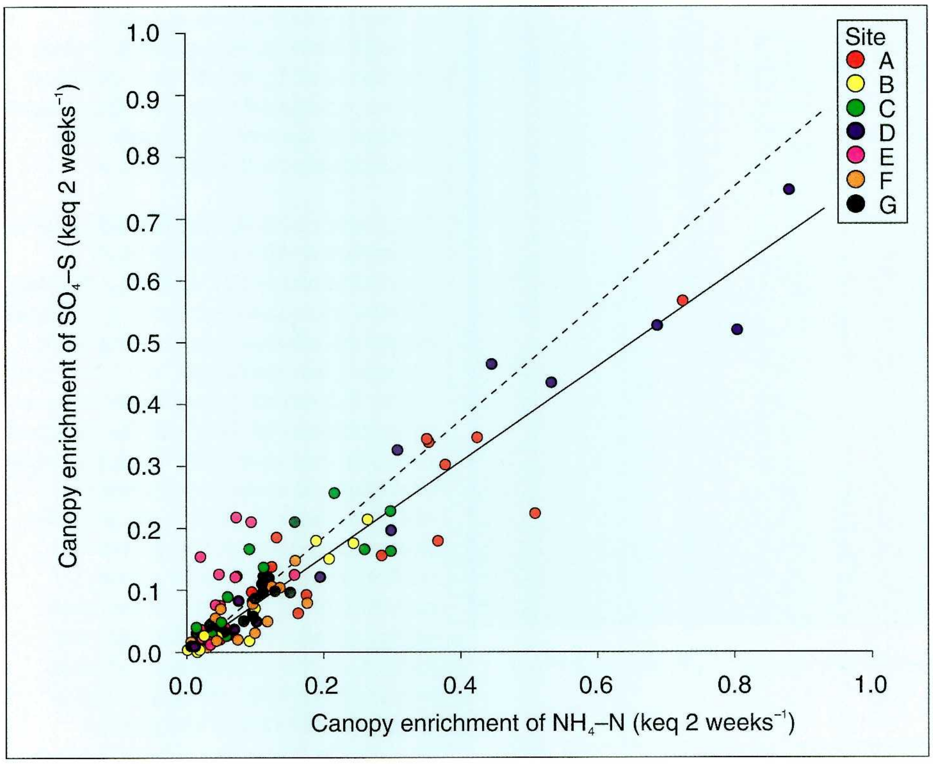


Figure 40. Relationship between canopy enrichment of SO_4–S and NH_4–N across all sites. The dotted line represents theoretical equivalence and the solid line is the actual regression line

showed a totally different seasonal pattern: here, the concentrations rose during winter and spring and fell during summer. Periods with high NH_3 concentration at site B were related to the time that the animals were brought on to the farm.

Throughfall and rainfall

Three rainfall collectors, consisting of polyethlene funnels (diameter *ca* 200 mm) fastened to the posts supporting the diffusion tubes at a height of 1.25 m, were placed adjacent to each wood. Each wood was also equipped with six throughfall gauges, similar to the rainfall collectors, but located under the canopy. Rainfall and throughfall samples were collected every two weeks and analysed for pH, ammonium, nitrate and sulphate.

The mean amount of rainfall for the whole sampling period was 588 mm, with the highest rainfall at site C (652 mm) and the lowest at site F (492 mm). Analysis of variance showed no significant difference between the amount of rainfall at all sites, but the concentrations of NH_4^+ were slightly higher at sites B and F, suggesting some dry deposition of NH_3 gas to the rain gauges. In contrast, analysis of NO_3 and SO_4 concentrations in rainwater showed no significant difference between the rainfall monitoring points.

Remarkably high amounts of NH_4^+ were found in the throughfall at sites D and A, 112 and 104 kg N ha^{-1} yr^{-1}, respectively. These two sites were located downwind of the pig farm and, to our knowledge, have the highest N deposition in throughfall ever recorded in the UK. Immediately downwind from the pig farm was site C, with an NH_4–N enrichment of 51 kg N ha^{-1} yr^{-1}, still far higher than normally expected but much lower than sites D and A. The highest deposition rates were expected at site C, but the trees were young and consequently the canopy was poorly formed, which may explain why deposition was not great to this forest. The lowest NH_4–N enrichment was found at site E (34 kg N ha^{-1} yr^{-1}), the site furthest upwind from the farm and downwind of a large area of moorland. Figure 39 shows the general pattern of NH_4^+ enrichment under canopies across the study site.

Figure 40 shows the amount of canopy enrichment of SO_4–S and NH_4–N found at all the study sites, plotted in equivalence units. Again, the highest amounts of SO_4^{2-} deposition were recorded at sites D (86.9 kg S ha^{-1} yr^{-1}) and A (81.0 kg S ha^{-1} yr^{-1}), with lowest amounts at the sites upwind from the pig farm. There was a very strong correlation between canopy enrichment of SO_4^{2-} and NH_4^+ (r^2=0.87), indicating the co-deposition of these two ions.

The N deposition rates measured at Wensleydale are as great as those recorded in The Netherlands, where similar high N loads have been considered a serious threat to the vitality of forest ecosystems. Extremely high deposition rates of 117.3 kg N ha^{-1} yr^{-1}, of which 83% was NH_4–N, have been reported in The Netherlands, which correspond with the fluxes reported here.

There are no strong sources for SO_2 within the study area, and we have no reason to suspect any strong SO_2 gradient across it. The high SO_4–S enrichment rates occurring at those sites having high NH_4–N deposition suggest that NH_3 from the pig farm is co-depositing with SO_2, presumably derived from industrial and combustion sources outside the study area. Measurements of the stable isotope composition of throughfall enrichments at the Wensleydale site are being made in order to identify the source of the deposited S.

P Ineson, S M C Robertson and P Thomson

References

Hargreaves, K. 1989. *The development and application of diffusion tubes for air pollution measurements.* PhD thesis, University of Nottingham.

Interactions of atmospheric NH_3 and SO_2 exchange with plant communities

(This work was funded by the Department of the Environment, and involved collaboration with the University of Nottingham)

Atmospheric depositions of sulphur dioxide (SO_2) and ammonia (NH_3) both contribute to the acidifying inputs received by ecosystems. In the case of NH_3, which normally acts as a base, it may be oxidised through microbial action in the soil to produce nitric acid. In addition, NH_3 deposition is important because it represents an input of nitrogen which may affect nitrogen-limited ecosystems. It is, therefore, necessary to quantify the atmospheric

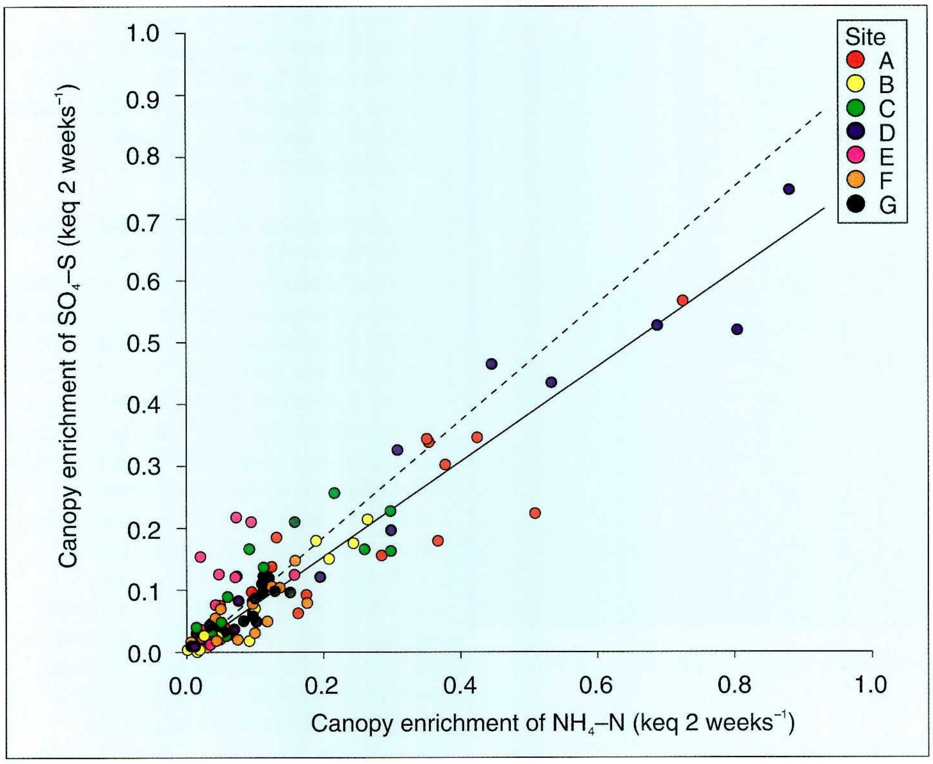


Figure 40. Relationship between canopy enrichment of SO_4–S and NH_4–N across all sites. The dotted line represents theoretical equivalence and the solid line is the actual regression line

showed a totally different seasonal pattern: here, the concentrations rose during winter and spring and fell during summer. Periods with high NH_3 concentration at site B were related to the time that the animals were brought on to the farm.

Throughfall and rainfall

Three rainfall collectors, consisting of polyethlene funnels (diameter *ca* 200 mm) fastened to the posts supporting the diffusion tubes at a height of 1.25 m, were placed adjacent to each wood. Each wood was also equipped with six throughfall gauges, similar to the rainfall collectors, but located under the canopy. Rainfall and throughfall samples were collected every two weeks and analysed for pH, ammonium, nitrate and sulphate.

The mean amount of rainfall for the whole sampling period was 588 mm, with the highest rainfall at site C (652 mm) and the lowest at site F (492 mm). Analysis of variance showed no significant difference between the amount of rainfall at all sites, but the concentrations of NH_4^+ were slightly higher at sites B and F, suggesting some dry deposition of NH_3 gas to the rain gauges. In contrast, analysis of NO_3 and SO_4 concentrations in rainwater showed no significant difference between the rainfall monitoring points.

Remarkably high amounts of NH_4^+ were found in the throughfall at sites D and A, 112 and 104 kg N ha^{-1} yr^{-1}, respectively. These two sites were located downwind of the pig farm and, to our knowledge, have the highest N deposition in throughfall ever recorded in the UK. Immediately downwind from the pig farm was site C, with an NH_4–N enrichment of 51 kg N ha^{-1} yr^{-1}, still far higher than normally expected but much lower than sites D and A. The highest deposition rates were expected at site C, but the trees were young and consequently the canopy was poorly formed, which may explain why deposition was not great to this forest. The lowest NH_4–N enrichment was found at site E (34 kg N ha^{-1} yr^{-1}), the site furthest upwind from the farm and downwind of a large area of moorland. Figure 39 shows the general pattern of NH_4^+ enrichment under canopies across the study site.

Figure 40 shows the amount of canopy enrichment of SO_4–S and NH_4–N found at all the study sites, plotted in equivalence units. Again, the highest amounts of SO_4^{2-} deposition were recorded at sites D (86.9 kg S ha^{-1} yr^{-1}) and A (81.0 kg S ha^{-1} yr^{-1}), with lowest amounts at the sites upwind from the pig farm. There was a very strong correlation between canopy enrichment of SO_4^{2-} and NH_4^+ (r^2=0.87), indicating the co-deposition of these two ions.

The N deposition rates measured at Wensleydale are as great as those recorded in The Netherlands, where similar high N loads have been considered a serious threat to the vitality of forest ecosystems. Extremely high deposition rates of 117.3 kg N ha^{-1} yr^{-1}, of which 83% was NH_4–N, have been reported in The Netherlands, which correspond with the fluxes reported here.

There are no strong sources for SO_2 within the study area, and we have no reason to suspect any strong SO_2 gradient across it. The high SO_4–S enrichment rates occurring at those sites having high NH_4–N deposition suggest that NH_3 from the pig farm is co-depositing with SO_2, presumably derived from industrial and combustion sources outside the study area. Measurements of the stable isotope composition of throughfall enrichments at the Wensleydale site are being made in order to identify the source of the deposited S.

P Ineson, S M C Robertson and P Thomson

References

Hargreaves, K. 1989. *The development and application of diffusion tubes for air pollution measurements.* PhD thesis, University of Nottingham.

Interactions of atmospheric NH_3 and SO_2 exchange with plant communities

(This work was funded by the Department of the Environment, and involved collaboration with the University of Nottingham)

Atmospheric depositions of sulphur dioxide (SO_2) and ammonia (NH_3) both contribute to the acidifying inputs received by ecosystems. In the case of NH_3, which normally acts as a base, it may be oxidised through microbial action in the soil to produce nitric acid. In addition, NH_3 deposition is important because it represents an input of nitrogen which may affect nitrogen-limited ecosystems. It is, therefore, necessary to quantify the atmospheric

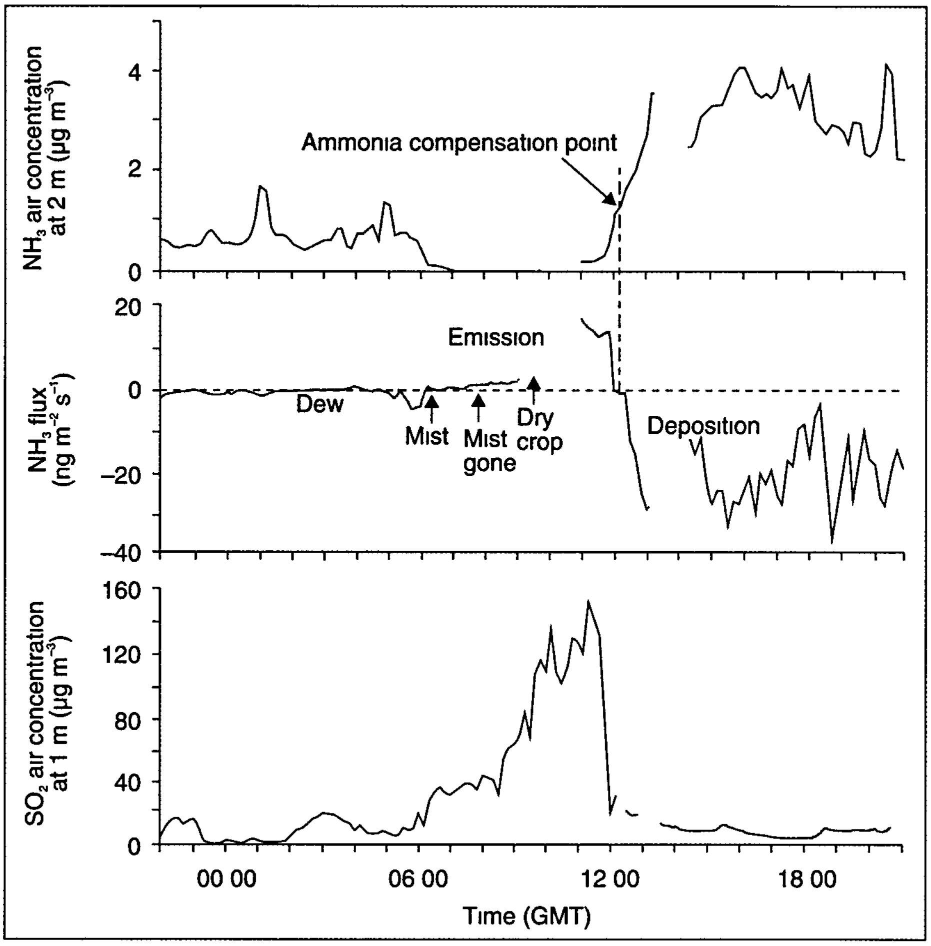


Figure 41 NH_3 flux over a wheat crop in relation to air concentrations of NH_3 and SO_2 NH_3 concentrations may be reduced by reaction with acidic species in the presence of large SO_2 concentrations, allowing net NH_3 emission from the canopy Small NH_3 fluxes at night are related to reduced atmospheric turbulence during this period

than deposition, was measured A likely explanation is that emission and deposition of NH_3 over agricultural cropland are also controlled by the existence of a 'compensation point' in plant tissues when air con-centrations are less than this point, NH_3 emission occurs, when air concentrations are larger, deposition occurs During the period of very large SO_2 air concentrations, NH_3 concentrations were greatly reduced, probably because of an atmospheric reaction with acidic aerosols associated with the SO_2 With the NH_3 air concentration smaller than the compensation point, NH_3 emission was therefore possible By contrast, the SO_2 results confirmed the expectation of little effect of NH_3 on SO_2 fluxes, because SO_2 concentrations were typically one to two orders of magnitude larger than those of NH_3, and resistances for SO_2 deposition to the canopy were consistent with uptake via the plant stomata

Conclusions

The present investigation and the study by Ineson *et al* show how two very different approaches may be used to examine interactions between NH_3 and SO_2 deposition From the throughfall work, it has been suggested that elevated NH_3 concentrations near a livestock farm substantially enhance local SO_2 deposition In contrast, SO_2 deposition is expected to be much less affected in situations of low NH_3 concentrations and excess SO_2, and this hypothesis was supported by the micro-meteorological measurements over the wheat crop However, the results over the wheat crop do not support the converse expectation, of a simple increase in deposition of NH_3 in the presence of excess SO_2 The existence of the NH_3 compensation point, and the effects of SO_2 and its products on NH_3 air concentrations can result in periods of NH_3 emission Such reactions clearly act in competition with the enhancement of deposition to leaf surfaces, so that the net effect on NH_3 fluxes would depend on the relative efficiency of the different processes (Figure 42)

In the high nitrogen status agricultural cropland, during the dry daytime conditions examined, it is possible that leaf surface reaction was less efficient, and a larger NH_3 compensation point allowed net emission Over forest vegetation, with lower nitrogen status, a smaller compensation point may be expected, so that the leaf surface reactions of NH_3 and SO_2 enhance acidic deposition The present requirement is to define these relationships to develop a functional dependence of the co-deposition interaction between ecosystem type and environmental conditions The complexities in the process show that simple assumptions, so often implied by the use of the co-deposition concept, may not be generally applicable, and a more rigorous

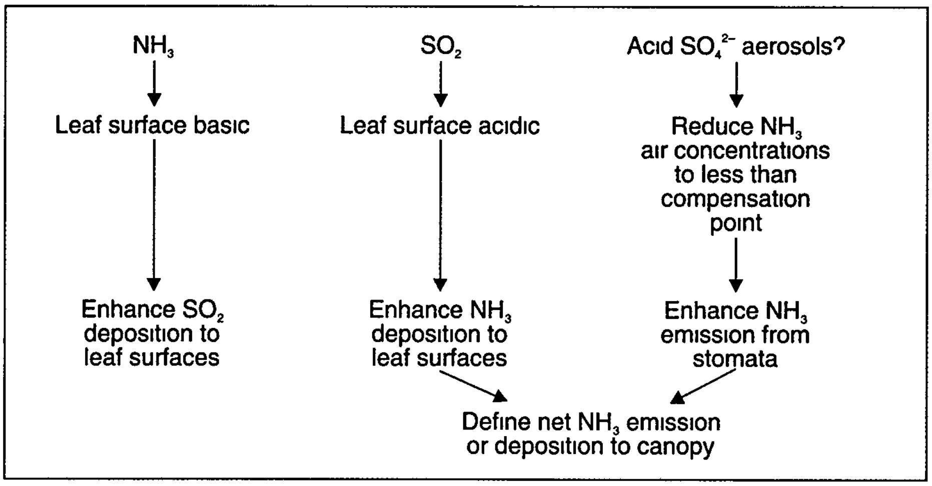


Figure 42 The co-deposition theory describes the enhancement of NH_3 and SO_2 deposition on to leaf surfaces in the presence of both gases However, the net effect on NH_3 fluxes may also depend on the simultaneous competition for emission from the canopy because of the existence of the NH_3 compensation point

Plate 27. Micrometeorological instrumentation for determining NH_3 and SO_2 fluxes over a wheat canopy at the field site

exchange of these gases with vegetation, and, in order to describe the atmospheric budget of these gases, information on the surface/atmosphere exchange is required for the whole range of major land uses.

Interactions between the deposition of SO_2 and NH_3 are possible because of their respective acidic and basic nature. The basic nature of NH_3 may allow more dissolution and uptake of acidic SO_2, while SO_2 itself may promote NH_3 deposition. These factors are particularly important when leaf surfaces are wet or when humidities are above the deliquescence point of ions already deposited on to leaf surfaces. These processes have been used to explain the observation of equivalent amounts of ammonium (NH_4^+) and sulphate (SO_4^{2-}) in throughfall beneath forest canopies (eg Ivens *et al.* 1989), an effect often referred to as 'co-deposition'. In such studies, precipitation chemistry is compared with that under a plant canopy, and the difference, termed 'net throughfall', is related to the wash-off of species dry-deposited to leaf surfaces.

The major atmospheric SO_2 sources are large combustion plants. By contrast, NH_3 is emitted primarily as a result of intensive livestock production, from the volatilisation of animal waste. The existence of interactions between SO_2 and NH_3 has important consequences for the control of acidic deposition. Such interactions could mean that controlling the emissions of only one of these gases would fail to provide an equivalent reduction in acidic inputs to sensitive ecosystems.

Two approaches have been used to examine these possible interactions. Measurements by staff at ITE Merlewood have examined the local pattern of ammonium and sulphur throughfall fluxes in woodlands around a livestock farm (see Ineson, Robertson & Thomson, pp57–59), while staff at ITE Edinburgh have used micrometeorological methods to study the chemical and physical processes of NH_3 and SO_2 interactions on a canopy of winter wheat.

Micrometeorological measurement of NH_3 and SO_2 fluxes over a wheat crop

In the micrometeorological approach, measurements are made of turbulence and changes in air concentrations a few metres above the surface to infer deposition or emission fluxes at the surface itself. The method used is the aerodynamic gradient technique, whereby turbulent diffusion is quantified from vertical profiles of windspeed (u) and temperature, and combined with vertical NH_3 and SO_2 profiles (c) to calculate fluxes to the ground (Sutton, Moncrieff & Fowler 1992). The flux (F_g) may be found from:

$$F_g = -k^2 \frac{du}{d[\ln(z-d) - \Psi_M]} \frac{d\chi}{d[\ln(z-d) - \Psi_H]}$$

where k is the von Karman constant; (z–d) is height above the aerodynamic 'ground' level or zero plane of a plant canopy; and Ψ_M and Ψ_H are correction factors, calculated from the temperature profiles, to account for the effects of atmospheric stability on the flux.

The major constraint on the technique is the accurate determination of air concentrations of NH_3 and SO_2. Sulphur dioxide was measured using a sensitive pulsed fluoresence analyser capable of detecting concentrations of 0.1 part per billion (0.3 mg m^{-3}), but determining air concentrations for NH_3 with the required precision and sensitivity was a particular challenge. In order to separate out interference from ammonium particles present in the air, and to provide a continuous measurement at low concentrations, NH_3 was determined using a continuous annular denuder system. In this technique, which has only recently been developed, air is sampled at 30 l min^{-1} and passed through the 2 mm space between two concentrically arranged glass tubes. These conditions permit the development of laminar flow in the denuder, allowing NH_3 to diffuse to the walls but aerosol ammonium to pass though. The denuders are continuously rotated and wetted with an acidic solution which is analysed for ammonium by a sensitive diffusion technique. The analysis of ammonium in solution has a detection limit of less than 0.1 mg l^{-1}, which permits the detection of air concentrations of NH_3 down to 0.01 mg m^{-3} with a time resolution of two minutes. The denuder system with equipment for measuring the wind profile and the inlets for the SO_2 analysis are shown in position at the field site in Plate 27.

In contrast to the Merlewood throughfall study, the typical conditions in this experiment were very large SO_2 concentrations with much smaller levels of NH_3. In this situation, only a small enhancement of SO_2 deposition is to be expected, because of the limited NH_3 concentrations, and both emission and deposition of NH_3 may occur.

The results, however, show that the opposite pattern can occur. In Figure 41, the NH_3 flux with the wheat canopy for 19 May 1992 is shown alongside the air concentrations of NH_3 and SO_2 (Sutton, Asman & Schjørring 1993). During a period of very large SO_2 concentrations, NH_3 emission, rather approach is required to estimate net NH_3 and SO_2 fluxes on the landscape scale.

M A Sutton and D Fowler

References

Ivens, W.P.M.F., Draaijers, G.P.J., Bleuten, W. & Bos, M.M. 1989. The impact of air-borne ammonia from agricultural sources on fluxes of nitrogen and sulphur towards forest soils. *Catena,* **16**, 535–544.

Sutton, M.A., Moncrieff, J.B. & Fowler, D. 1992. Deposition of atmospheric ammonia to moorlands. *Environmental Pollution,* **75**, 15–24.

Sutton, M.A., Asman, W.A.H. & Schjørring, J.K. 1993. *Dry deposition of reduced nitrogen.* (Background document presented at workshop on deposition of atmospheric pollutants, Göteborg, 1992.) Copenhagen: Nordic Council of Ministers. In press.

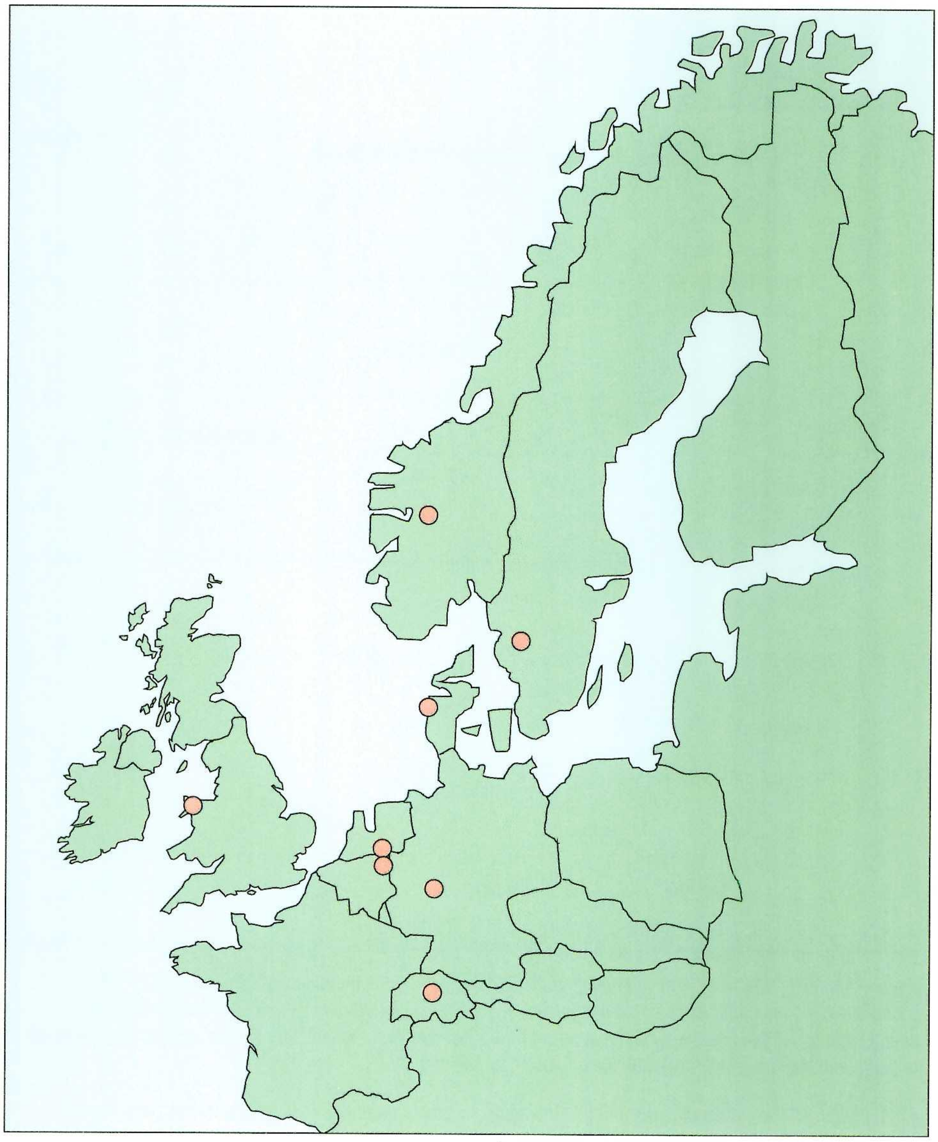


Figure 43. Location of the NITREX study sites at which nitrogen addition or removal experiments are underway

Nitrogen saturation in Welsh Sitka spruce forests

Until recently, it was thought that nitrogen was tightly cycled in forested ecosystems with little or no release to surface or groundwater in the absence of disturbance. However, recent reports elsewhere in Europe have linked enhanced atmospheric nitrogen deposition following industrialisation and intensification of agriculture with nitrate leakage from forest ecosystems. This leakage is assumed to be due to a supply of nitrogen greater than that required by trees and soil micro-organisms, resulting in the leaching of excess nitrogen. There can be a variety of adverse effects, including nutrient imbalances in trees, acidification of soils and waters, and changes in vegetation community structure due to eutrophication (Aber *et al.* 1989).

Plate 28. A view of Aber forest with the automatic weather station in the foreground

In remote pristine areas such as northern Scandinavia, nitrogen deposition is generally less than 5 kg N ha^{-1} yr^{-1}. In highly polluted areas of central Europe, deposition can be as high as 50 –150 kg N ha^{-1} yr^{-1}. There is interest in determining the maximum amount of nitrogen deposition (or the 'critical load') that can be retained by various approach is required to estimate net NH_3 and SO_2 fluxes on the landscape scale.

M A Sutton and D Fowler

References

Ivens, W.P.M.F., Draaijers, G.P.J., Bleuten, W. & Bos, M.M. 1989. The impact of air-borne ammonia from agricultural sources on fluxes of nitrogen and sulphur towards forest soils. *Catena*, **16**, 535–544.

Sutton, M.A., Moncrieff, J.B. & Fowler, D. 1992. Deposition of atmospheric ammonia to moorlands. *Environmental Pollution*, **75**, 15–24.

Sutton, M.A., Asman, W.A.H. & Schjørring, J.K. 1993. *Dry deposition of reduced nitrogen.* (Background document presented at workshop on deposition of atmospheric pollutants, Göteborg, 1992.) Copenhagen: Nordic Council of Ministers. In press.

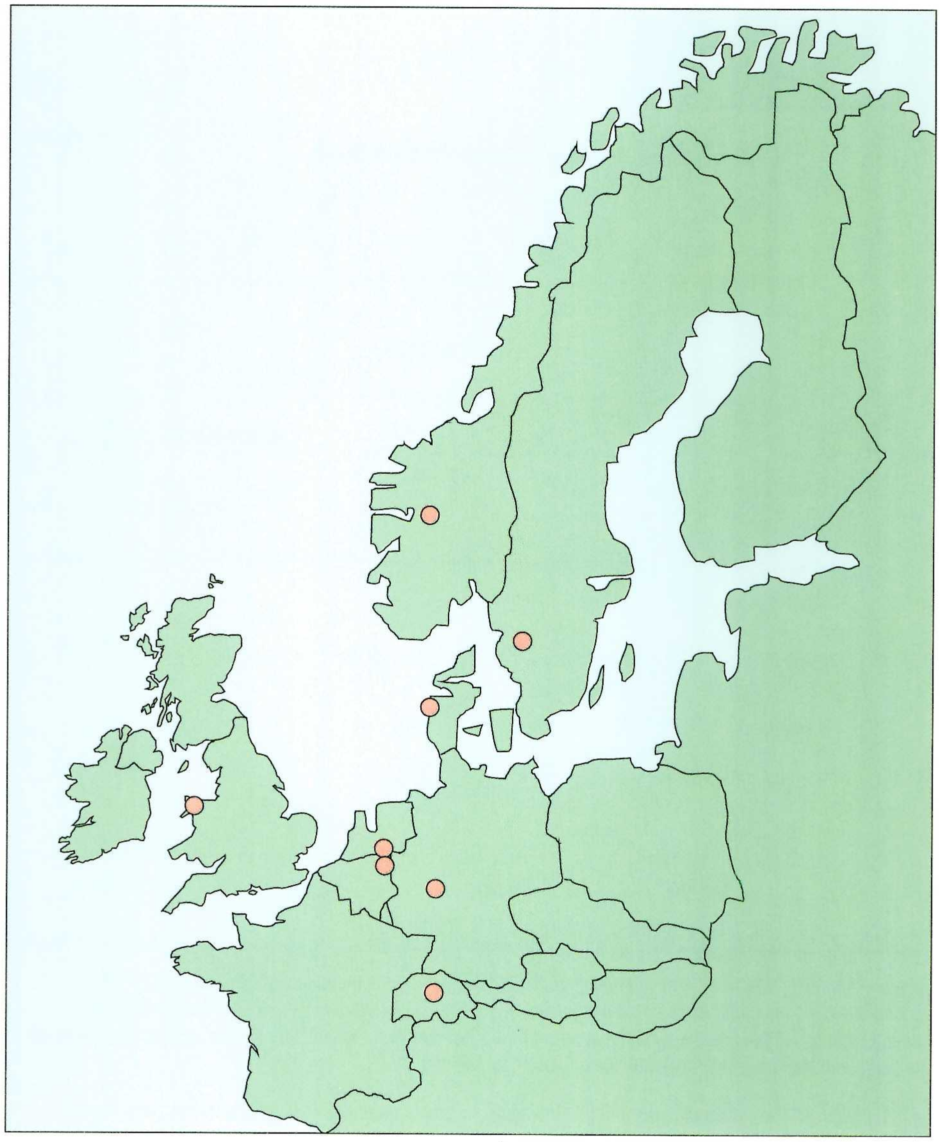


Figure 43. Location of the NITREX study sites at which nitrogen addition or removal experiments are underway

Nitrogen saturation in Welsh Sitka spruce forests

Until recently, it was thought that nitrogen was tightly cycled in forested ecosystems with little or no release to surface or groundwater in the absence of disturbance. However, recent reports elsewhere in Europe have linked enhanced atmospheric nitrogen deposition following industrialisation and intensification of agriculture with nitrate leakage from forest ecosystems. This leakage is assumed to be due to a supply of nitrogen greater than that required by trees and soil micro-organisms, resulting in the leaching of excess nitrogen. There can be a variety of adverse effects, including nutrient imbalances in trees, acidification of soils and waters, and changes in vegetation community structure due to eutrophication (Aber *et al.* 1989).

Plate 28. A view of Aber forest with the automatic weather station in the foreground

In remote pristine areas such as northern Scandinavia, nitrogen deposition is generally less than 5 kg N ha^{-1} yr^{-1}. In highly polluted areas of central Europe, deposition can be as high as 50 –150 kg N ha^{-1} yr^{-1}. There is interest in determining the maximum amount of nitrogen deposition (or the 'critical load') that can be retained by various

Plate 30. The water storage tanks at Aber

dry deposition. We are applying a further 35 kg N ha^{-1} yr^{-1} as sodium nitrate and 35 kg N ha^{-1} yr^{-1} ammonium nitrate to sets of three replicate 15 m x 15 m experimental plots (Figure 44). An additional treatment of 75 kg N ha^{-1} yr^{-1} applied as sodium nitrate enables the effects of high inputs of nitrate to be determined. The nitrogen is applied weekly as a spray after mixing with deionised streamwater collected upslope of the plots (Plate 29). The water is stored on site in large storage tanks with 1200 gallons required for each spraying event (Plate 30). As the water alone may have an impact, a water-only control is included in the experimental design.

We are monitoring the effects of the different treatments on soil water chemistry, tree growth and nutrient status, soil processes and 'greenhouse' gas emissions. We hope to determine the responses of the forest ecosystem and to understand what processes are important in these responses.

Results

The results from the first 18 months of spraying indicate that this forest has a very limited capacity to immobilise nitrogen in the form of nitrate. Nitrate is leached from all treatments and increases in proportion to inputs (Figure 45). Indeed, nitrogen losses under ambient deposition rates are 7 kg N ha^{-1} yr^{-1}, equivalent to the nitrate entering as throughfall. Ammonium leaching losses are low in all treatments. In association with increased nitrate leaching, increases are also observed in cation leaching losses. Sodium leaching losses have increased (due to the input of nitrogen as sodium nitrate), but cation exchange has also resulted in increased aluminium concentration in soil waters, although it is variable between replicate treatment plots (Figure 46). Large amounts of aluminium in soil water and streamwater can have an adverse effect on both tree root vitality and streamwater biota. From this information, we can predict that any increase in NO_x emissions, and subsequent deposition, will result in an equal increase in nitrate leaching and possible elevated concentrations of aluminium in both soil water and streamwater.

In contrast to the fate of nitrate inputs, ammonium appears to be effectively retained within the forest system (Figure 45). Although two treatments receive the same total nitrogen input (35 kg N ha^{-1} yr^{-1}), the nitrate losses are dependent primarily on the proportion of nitrate present in the incoming nitrogen. Thus, the increase in nitrate leaching losses in the ammonium nitrate treatment above those in the control is equivalent to the nitrate input only of 17.5 kg N ha^{-1} yr^{-1}. No increase in tree needle nitrogen concentrations has been observed in any

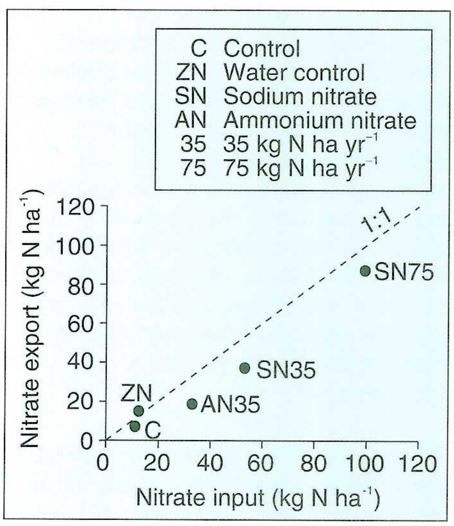


Figure 45. Nitrate leaching losses in relation to nitrate inputs in the different treatment plots during a 15-month period

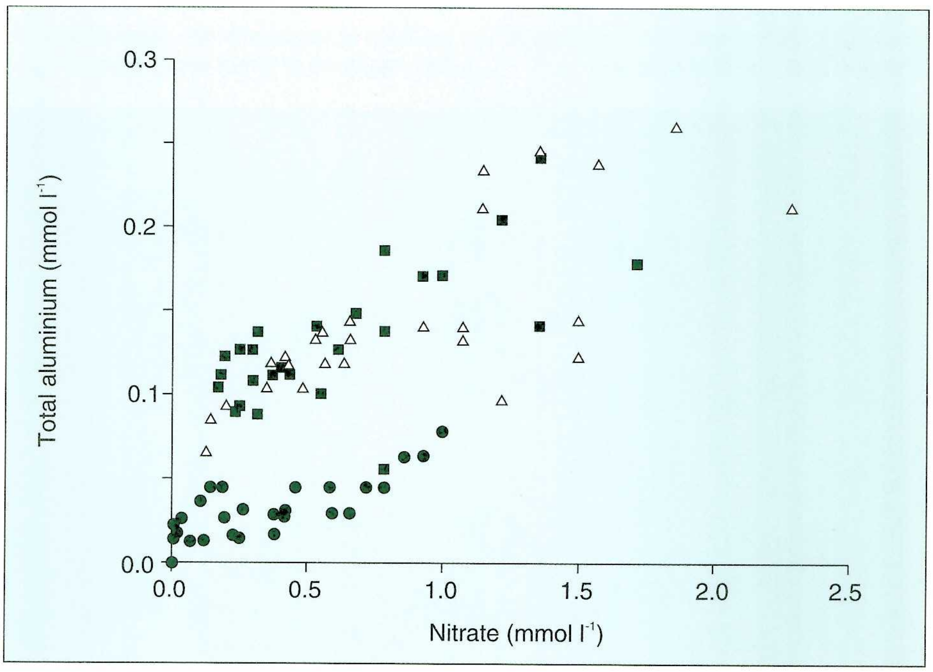


Figure 46. The increase in total aluminium concentrations in the mineral soil as nitrate leaching increases in the 35 kg N ha^{-1} yr^{-1} sodium nitrate treatment (r^2=0.54, P<0.001). The results from the three replicate treatment plots are indicated

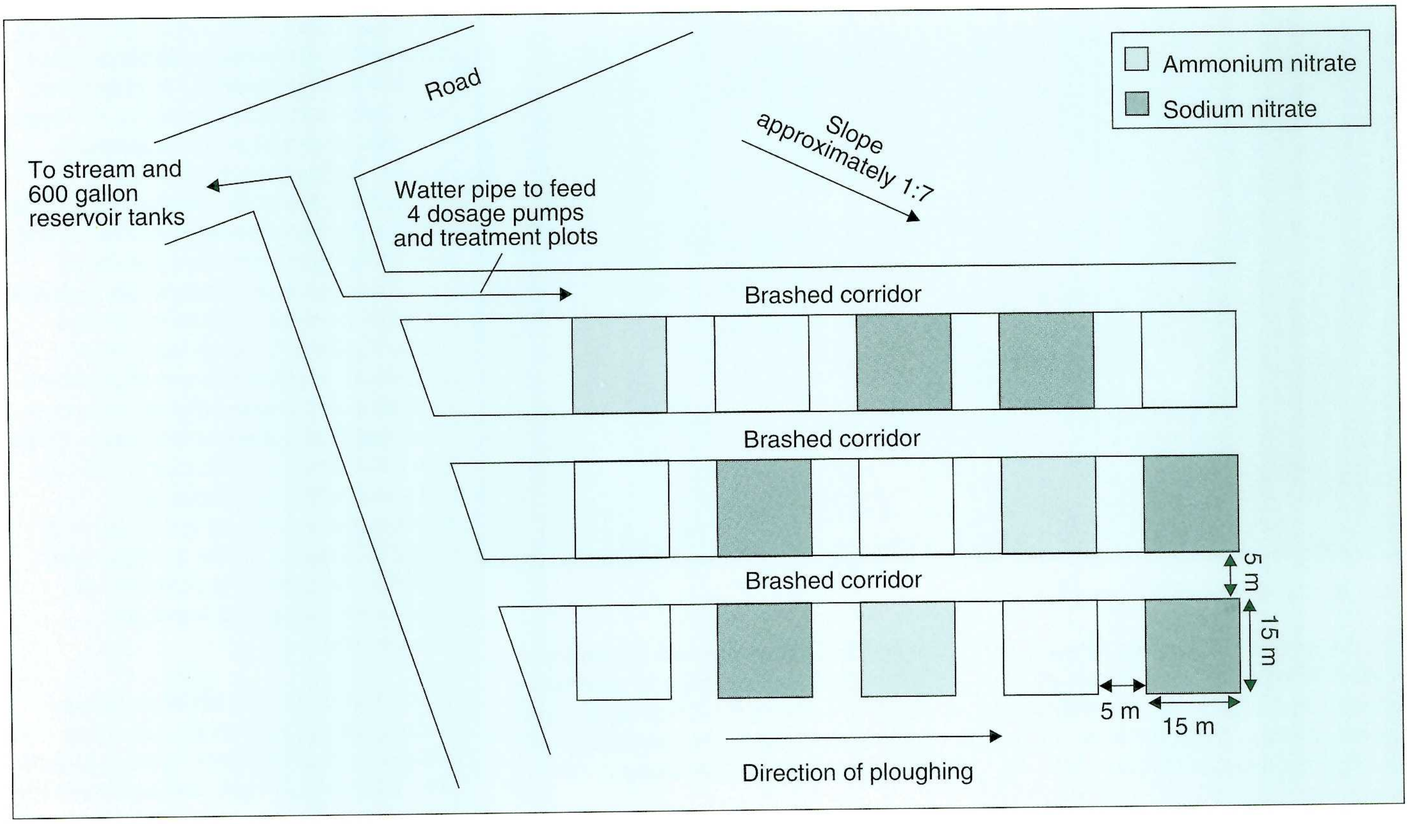


Figure 44. The experimental design at Aber showing the control, water-only and nitrate-N and/or ammonium-N additions

ecosystem types which results in little or no leakage of nitrate and no damage to sensitive parts of the ecosystem.

Together with various collaborators in Europe and Scandinavia in the NITREX (nitrogen saturation experiments) project, ITE is undertaking manipulation experiments across a nitrogen pollution gradient to determine the future role of nitrogen in the acidification and eutrophication of coniferous forest ecosystems (Figure 43). In highly polluted areas such as The Netherlands, roof exclusion experiments are being carried out to determine the reversibility of the effects of chronic nitrogen deposition. In areas where no damage has been reported and nitrogen leakage is low (as in parts of the UK), nitrogen addition experiments are determining the response of forest ecosystems to elevated nitrogen deposition on different soils and under a variety of climates.

Plate 29. Spraying of nitrogen on to the forest floor at Aber

The Aber experiment

We are investigating the potential impact of chronic nitrogen deposition in a 30-year-old Sitka spruce (*Picea sitchensis*) stand in Aber forest in north Wales (Plate 28). The forest is near the coast on an acidic stagnopodzol soil. Pollutant nitrogen can be taken up by the trees, immobilised in the soil, released as a gas back to the atmosphere, or leached from the soil. We are applying nitrogen to the forest floor in Aber forest at different rates and in different forms to determine the response of these different pools and processes to increased nitrogen deposition. Together with a more widespread survey of spruce plantations in north and mid-Wales (Stevens *et al.* 1993), this will enable us to determine the fate of incoming nitrogen (whether as nitrate or ammonium) and the amount of nitrogen deposition that may result in damage to either forest, soil or water resources.

The treatments were chosen to compare the effects of ammonium (originating as NH_3 gas from agricultural sources) and nitrate (emitted as NO_x gas during fossil fuel combustion). Inputs to the site at present are 17 kg N ha^{-1} yr^{-1} in wet and treatment, thus tree uptake does not account for the ammonium retention. It is most likely that soil micro-organisms have immobilised the ammonium-N into the soil organic matter, which may eventually result in a stimulation of the nitrification process (the conversion of ammonium to nitrate by soil micro-organisms) and an increase in nitrate leaching (Aber *et al.* 1989). However, no such stimulation has been found in soil incubation studies carried out so far.

Nitrification is an acidifying process and, therefore, conversion of ammonium to nitrate and subsequent leaching would be more damaging than direct leaching of incoming atmospheric nitrate. There may be a time lag in the stimulation of this process in our ammonium nitrate treatment. The process could be accelerated if an increase in ammonium availability were combined with an increase in soil temperature (Ineson 1992). Such a situation would occur if tree canopy cover was reduced because of root damage and decreased tree growth, in response to the increased aluminium concentrations (Ineson 1992). We will continue to monitor all treatment plots for a further two years to determine the effects of the incoming ammonium on both tree growth and health, soil processes and soil water chemistry. In addition, we are using the stable isotope ^{15}N to answer some of the questions raised concerning the fate of ammonium.

The Aber experiment has demonstrated that this forest already receives nitrogen inputs in excess of its requirements. Any increase in nitrate (or NO_x) inputs will result in acidification of soil water and streamwater. Continuation of the experiment is required to determine the effect of increased ammonium deposition in the absence of tree decline. Information from a survey throughout mid- and north Wales of catchments dominated by different-aged spruce stands indicates that the findings at Aber are applicable to most spruce stands on freely draining acidic soils over 30 years of age (Stevens *et al.* 1993). However, younger stands are highly efficient at immobilising nitrogen because of their greater nitrogen uptake requirements, and may not respond as rapidly to increased nitrogen inputs.

The data collected during this study and in the studies by our European collaborators are being used to test two forest/catchment nitrogen models being developed by other research groups in Europe and the USA. These models will help us to predict, on a wider geographical basis, the future impacts of different nitrogen deposition loads to coniferous forest ecosystems.

B Emmett, A Brittain, D Norris and B Reynolds

References

Aber, J.D., Nadelhoffer, K.J., Steudler, P. & Melillo, J.M. 1989. Nitrogen staturation in northern forest ecosystems. *Bioscience,* **39**, 378–386.

Ineson, P. 1992. Mechanisms of nutrient turnover in forest soils. *Annual Report of the Institute of Terrestrial Ecology 1991–92,* 40–42.

Stevens, P., Norris, D., Sparks, T.H. & Hodgson, A.L. 1993. Nitrate leaching from Sitka spruce forest and moorland in Wales. *Water, Air and Soil Pollution.* In press.

Effects of excess nitrogen on calcareous grasslands

(This work was partly funded by the Joint Environment Programme of National Power and PowerGen)

The deposition of nitrogen from the atmosphere over large areas of Europe has increased during the last decade from 2–6 kg N ha^{-1} yr^{-1} to values of between 15 and 60 kg N ha^{-1} yr^{-1}. There is widespread concern, especially among Dutch ecologists, that for some natural ecosystems the increase in nitrogen deposition is causing changes in species composition and, in particular, a loss of species richness and diversity. Much attention has focused on chalk grasslands, partly because they contain a large number of uncommon and rare species and partly because they are widespread in western Europe and are highly valued by conservationists. Studies of chalk grasslands in South Limburg (The Netherlands) have shown that the rhizomatous, perennial tor grass (*Brachypodium pinnatum*) has spread considerably in the past 50 years, leading to changes in species composition and a loss of diversity (Bobbink & Willems 1987). Ambient nitrogen deposition at these sites has been estimated to be between 35 and 40 kg N ha^{-1} yr^{-1}. It has been suggested that the major driving variable initiating and promoting these changes has been the increased deposition of atmospheric nitrogen, although it is conceded that the cessation of grazing about 50 years ago and its replacement by an autumn mowing regime may also have been contributory factors.

In Britain, chalk grasslands have traditionally been grazed by sheep, cattle and rabbits, with wide variations in the stocking density and the time of year of grazing. Mowing is rare. The present-day composition of any area of

Plate 31. Parsonage Down, Wiltshire – a species-rich grassland grazed by a mixture of sheep and cattle. There is no evidence that the deposition of atmospheric nutrients is affecting the species composition of the site (photograph: Mr P Wakely, English Nature)

treatment, thus tree uptake does not account for the ammonium retention. It is most likely that soil micro-organisms have immobilised the ammonium-N into the soil organic matter, which may eventually result in a stimulation of the nitrification process (the conversion of ammonium to nitrate by soil micro-organisms) and an increase in nitrate leaching (Aber *et al.* 1989). However, no such stimulation has been found in soil incubation studies carried out so far.

Nitrification is an acidifying process and, therefore, conversion of ammonium to nitrate and subsequent leaching would be more damaging than direct leaching of incoming atmospheric nitrate. There may be a time lag in the stimulation of this process in our ammonium nitrate treatment. The process could be accelerated if an increase in ammonium availability were combined with an increase in soil temperature (Ineson 1992). Such a situation would occur if tree canopy cover was reduced because of root damage and decreased tree growth, in response to the increased aluminium concentrations (Ineson 1992). We will continue to monitor all treatment plots for a further two years to determine the effects of the incoming ammonium on both tree growth and health, soil processes and soil water chemistry. In addition, we are using the stable isotope ^{15}N to answer some of the questions raised concerning the fate of ammonium.

The Aber experiment has demonstrated that this forest already receives nitrogen inputs in excess of its requirements. Any increase in nitrate (or NO_x) inputs will result in acidification of soil water and streamwater. Continuation of the experiment is required to determine the effect of increased ammonium deposition in the absence of tree decline. Information from a survey throughout mid- and north Wales of catchments dominated by different-aged spruce stands indicates that the findings at Aber are applicable to most spruce stands on freely draining acidic soils over 30 years of age (Stevens *et al.* 1993). However, younger stands are highly efficient at immobilising nitrogen because of their greater nitrogen uptake requirements, and may not respond as rapidly to increased nitrogen inputs.

The data collected during this study and in the studies by our European collaborators are being used to test two forest/catchment nitrogen models being developed by other research groups in Europe and the USA. These models will help us to predict, on a wider geographical basis, the future impacts of different nitrogen deposition loads to coniferous forest ecosystems.

B Emmett, A Brittain, D Norris and B Reynolds

References

Aber, J.D., Nadelhoffer, K.J., Steudler, P. & Melillo, J.M. 1989. Nitrogen staturation in northern forest ecosystems. *Bioscience,* **39**, 378–386.

Ineson, P. 1992. Mechanisms of nutrient turnover in forest soils. *Annual Report of the Institute of Terrestrial Ecology 1991–92,* 40–42.

Stevens, P., Norris, D., Sparks, T.H. & Hodgson, A.L. 1993. Nitrate leaching from Sitka spruce forest and moorland in Wales. *Water, Air and Soil Pollution.* In press.

Effects of excess nitrogen on calcareous grasslands

(This work was partly funded by the Joint Environment Programme of National Power and PowerGen)

The deposition of nitrogen from the atmosphere over large areas of Europe has increased during the last decade from 2–6 kg N ha^{-1} yr^{-1} to values of between 15 and 60 kg N ha^{-1} yr^{-1}. There is widespread concern, especially among Dutch ecologists, that for some natural ecosystems the increase in nitrogen deposition is causing changes in species composition and, in particular, a loss of species richness and diversity. Much attention has focused on chalk grasslands, partly because they contain a large number of uncommon and rare species and partly because they are widespread in western Europe and are highly valued by conservationists. Studies of chalk grasslands in South Limburg (The Netherlands) have shown that the rhizomatous, perennial tor grass (*Brachypodium pinnatum*) has spread considerably in the past 50 years, leading to changes in species composition and a loss of diversity (Bobbink & Willems 1987). Ambient nitrogen deposition at these sites has been estimated to be between 35 and 40 kg N ha^{-1} yr^{-1}. It has been suggested that the major driving variable initiating and promoting these changes has been the increased deposition of atmospheric nitrogen, although it is conceded that the cessation of grazing about 50 years ago and its replacement by an autumn mowing regime may also have been contributory factors.

In Britain, chalk grasslands have traditionally been grazed by sheep, cattle and rabbits, with wide variations in the stocking density and the time of year of grazing. Mowing is rare. The present-day composition of any area of

Plate 31. Parsonage Down, Wiltshire – a species-rich grassland grazed by a mixture of sheep and cattle. There is no evidence that the deposition of atmospheric nutrients is affecting the species composition of the site (photograph: Mr P Wakely, English Nature)

Table 8 Summary of the main features of the vegetation on four transects at Parsonage Down, 1970 and 1990

Transect no	No of species per quadrat 1970	1990	No of grasses 1970	1990	No of sedges 1970	1990	No of forbs 1970	1990	No of mosses 1970	1990
1 (n=38)	23 80	20 8	10	8	3	3	38	35	5	4
2 (n=35)	22 10	21 5	9	9	3	3	39	37	5	2
3 (n=21)	22 10	22 3	8	11	3	3	34	32	4	2
4 (n=21)	26 05	19 2	9	10	3	3	39	35	3	2

chalk grassland reflects not only physical factors such as soil type, soil depth, slope, aspect and climate, but also the management which the site has received in the past Any attempt, therefore, to assess whether increased nitrogen *deposition is affecting* the composition of chalk grassland must take into account the confounding effect of management, and the possible effect of shifts in climate in the same time period

One approach to this problem, and the one described here, is to compare the composition of the grassland at some previous point in time with its present composition, at a site where the management has remained the same and where variation in the climate has been small

The site

Use was made of four permanent transects, laid down in 1970 at Parsonage Down (Plate 31), Wiltshire, to study the floristic composition of ancient chalk grassland overlying Celtic field systems (Wells 1985)

The same 115 quadrats recorded in 1970 were recorded in 1990, using exactly the same methods as employed 20 years previously, soil samples were taken and analysed, again using the same methods The significance of changes in floristic composition and in soil chemistry was tested using the paired comparison 't' test

The 276 ha site had been in one ownership for 53 years until it was taken over by the Nature Conservancy Council in 1973, who has continued with the same method of management The site has been maintained for the past 70 years by mixed grazing (sheep and cattle), stocking levels being carefully controlled so that the grassland was neither under- or over-grazed The sward was kept short (usually less than 5 cm high), occasionally chain-harrowed, but no fertilizers were ever applied Stocking levels were maintained at 0 25 dairy cow equivalents per acre For all practical purposes, the site can be considered to have had the same management for the past 70 years, any changes in floristic composition could reasonably be ascribed to changes in external factors, such as incoming nutrients or climate

Results

In 1990 Parsonage Down had much the same appearance as it had in 1970 The turf still had that fine-grain appearance which is characteristic of ancient,

Table 9 Flux in composition of vegetation on four transects at Parsonage Down, 1970 and 1990

Transect no	Species present in 1970, not 1990 (with frequency)	Species present in 1990, not 1970 (with frequency)
1 (n=38)	*Acrocladium cuspidatum* (3) *Crepis capillaris* (3) *Hippocrepis comosa* (10) *Luzula campestris* (3) *Orchis morio* (3) *Phleum nodosum* (3) *Rhytidiadelphus squarrosus* (3) *Thymus drucei* (5) *Zerna erecta* (3)	*Cirsium arvense* (3) *Fissidens* spp (3) *Picris hieracioides* (8)
2 (n=35)	*Acrocladium cuspidatum* (1) *Asperula cynanchica* (14) *Crepis capillaris* (3) *Hieracium pilosella* (3) *Ligustrum vulgare* (3) *Mnium* sp (3) *Orchis morio* (6) *Rhytid squarrosus* (9) *Thymus drucei* (3) *Tragopogon pratense* (3) *Trifolium campestre* (3) *Veronica chamaedrys* (3)	*Achillea millefolium* (3) *Anthyllis vulneraria* (6) *Cirsium arvense* (6) *Gentianella amarella* (3) *Ononis spinosa* (6) *Polygala calcarea* (3) *Senecio jacobaea* (9)
3 (n=21)	*Acrocladium cuspidatum* (10) *Bellis perennis* (5) *Crepis capillaris* (5) *Luzula campestris* (14) *Picris hieracioides* (33) *Polygala calcarea* (10) *Rhytid squarrosus* (14) *Senecio jacobaea* (5) *Spiranthes spiralis* (5) *Asperula cynanchica* (10)	*Anthyllis vulneraria* (6) *Cirsium arvense* (19) *Leucanthemum vulgare* (10) *Lolium perenne* (5) *Ononis spinosa* (19) *Phleum bertolonii* (19) *Taraxacum laevigatum* (19) *Trisetum flavescens* (5) *Gentianella amarella* (5)
4 (n=21)	*Campanula glomerata* (19) *Crepis capillaris* (5) *Hieracium pilosella* (14) *Orchis ustulata* (10) *Pseudoscleropodium purum* (28) *Spiranthes spiralis* (5) *Taraxacum laevigatum* (10) *Trifolium campestre* (10)	*Leontodon autumnalis* (33) *Leucanthemum vulgare* (5) *Lolium perenne* (10) *Thymus drucei* (5)

species-rich turf, consisting of a mosaic of small, closely grazed forbs and grasses. The most frequent grasses and sedges were quaking grass (*Briza media*), spring sedge (*Carex caryophyllea*), carnation grass (*C. flacca*), slender sedge (*C. humilis*), cock's-foot (*Dactylis glomerata*), sheep's fescue (*Festuca ovina*), meadow oat (*Avenula pratense*), hairy oat (*A. pubescens*) and crested hair-grass (*Koeleria macrantha*). There was no evidence of any one species becoming dominant.

The main qualitative characteristics of the grassland in 1970 and 1990 are summarised in Tables 8 and 9. The species composition of the site changed very little between 1970 and 1990: 68 species were recorded in May 1970, five species being recorded in 1990 but not in 1970; 61 species were recorded in August 1990, 12 species being recorded in 1970 but not in 1990. All of those species which were present in one year but not the other were of low frequency, so the probability of recording them was small, or they were species which had no above-ground leaves in August (eg bulbous buttercup (*Ranunculus bulbosus*)) and would therefore not have been seen.

Data were also analysed to detect any significant ($P>0.05$) quantitative changes in the cover of species which may have occurred between 1970 and 1990. Of the 73 species studied, 18 showed no significant change, five increased, 18 decreased, while 32 species with frequencies of less than 10% could not reasonably be assigned to any category.

No relationship was found between a species performance in the period 1970–90 and its Ellenberg nitrogen indicator value (Ellenberg 1988). Species with low Ellenberg values (1 and 2), which might be expected to have diminished as a result of increased atmospheric nitrogen deposition, were as frequent in the 'no significant change' category as in the group of species which showed a decrease in cover.

There were no significant differences ($P>0.05$) in soil organic matter content, extractable K, Ca and P between soils sampled in 1970 and 1990, whereas soil pH increased from 7.5 to 7.8 ($P<0.001$), total N increased from 1.069 to 1.169 ($P<0.001$), total P increased from 0.128 to 0.147 ($P<0.001$) and extractable Mg decreased from 19.55 to 16.56 ($P<0.001$). Total annual nitrogen deposition for that part of southern England which includes Wiltshire is estimated to be between 16 and 20 kg N ha^{-1} yr^{-1} (Williams *et al.* 1989). This value lies within the range (14–25 kg N ha^{-1} yr^{-1}) suggested by Bobbink *et al.* (1992) as the critical load for nitrogen in calcareous grassland. It should be stressed that the conclusions of Bobbink *et al.* are based largely on studies in The Netherlands in which the dominant species was tor grass. It has been suggested that other grasses might behave similarly. The results obtained from Parsonage Down do not support this contention. There is no evidence of substantial changes in the floristic composition of the grassland in the period 1970–90. All species which were present in 1990 were still present in 1990, including 18 species with Ellenberg nitrogen indicator values of 1 and 2 which should have been most susceptible to increased nitrogen deposition.

A possible explanation for the apparent stability (at the macro-scale) of the grassland at Parsonage Down is that any possible advantage which the grasses might expect to gain from their potential ability to utilise incoming nitrogen more effectively than forbs is negated by the grazing animal which reduces their competitive ability by defoliation.

The calcareous nature of the soil, with its huge reserves of calcium, makes it unlikely that acidification will occur in the short or long term as a result of increased nitrogen deposition. It seems likely that any increase in atmospheric N will be cycled in the soil/vegetation/animal system, *provided the current system of management, using the grazing animal, is maintained.*

T C E Wells

References

Bobbink, R. & Willems, J.H. 1987. Increasing dominance of *Brachypodium pinnatum* (L.) Beauv. in chalk grasslands: a threat to a species-rich ecosystem. *Biological Conservation*, **40**, 301–314.

Bobbink, R., Boxman, D., Fremstad, E., Heil, G., Houdijk, A. & Roelofs, J. 1992. *Critical loads for nitrogen eutrophication of terrestrial and wetland ecosystems based upon changes in vegetation and fauna.* (Report.) Copenhagen: Nordic Council of Ministers.

Ellenberg, H. 1988. *Vegetation ecology of central Europe.* Cambridge: Cambridge University Press.

Wells, T.C.E. 1985. The botanical and ecological interest of ancient monuments. In: *Archaeology and nature conservation*, edited by G. Lambrick, 1–9. Oxford: Dept for External Studies, University of Oxford.

Williams, M.L., Atkins, D.H.F., Bower, J.S., Campbell, G.W., Irwin, J.G. & Simpson, D. 1989. *A preliminary assessment of the air pollution climate of the UK.* (Report LR723(AP).) Stevenage: Warren Spring Laboratory.

The ecological effects of liming

(This work was partly funded by the Joint Environment Programme of National Power and PowerGen, and the Nature Conservancy Council, and involved collaboration with Dr S Mackenzie of the Department of Environmental Biology, University of Manchester)

In many countries, acid deposition has led to progressive acidification of surface waters, which has been linked to reductions in numbers of aquatic invertebrates and fish. Upland areas of the UK, including extensive areas of blanket bog and moorland, have been the most severely affected. Liming the catchment areas (watersheds) around lakes is one way to mitigate the effects of acidification; soil water and surface runoff is neutralised before it reaches the lake, and the quality of the lake water is subsequently improved (Howells & Dalziel 1992). This technique is appropriate for lakes with short retention times where direct liming of the water is ineffective.

Blanket bog and moorland are naturally acidic, and there has been concern from conservation interests that liming may be detrimental to the terrestrial ecosystem. In small-scale field trials on blanket bog,

Plate 32. Field plots at Llyn Conwy, north Wales

species-rich turf, consisting of a mosaic of small, closely grazed forbs and grasses. The most frequent grasses and sedges were quaking grass (*Briza media*), spring sedge (*Carex caryophyllea*), carnation grass (*C. flacca*), slender sedge (*C. humilis*), cock's-foot (*Dactylis glomerata*), sheep's fescue (*Festuca ovina*), meadow oat (*Avenula pratense*), hairy oat (*A. pubescens*) and crested hair-grass (*Koeleria macrantha*). There was no evidence of any one species becoming dominant.

The main qualitative characteristics of the grassland in 1970 and 1990 are summarised in Tables 8 and 9. The species composition of the site changed very little between 1970 and 1990: 68 species were recorded in May 1970, five species being recorded in 1990 but not in 1970; 61 species were recorded in August 1990, 12 species being recorded in 1970 but not in 1990. All of those species which were present in one year but not the other were of low frequency, so the probability of recording them was small, or they were species which had no above-ground leaves in August (eg bulbous buttercup (*Ranunculus bulbosus*)) and would therefore not have been seen.

Data were also analysed to detect any significant ($P>0.05$) quantitative changes in the cover of species which may have occurred between 1970 and 1990. Of the 73 species studied, 18 showed no significant change, five increased, 18 decreased, while 32 species with frequencies of less than 10% could not reasonably be assigned to any category.

No relationship was found between a species performance in the period 1970–90 and its Ellenberg nitrogen indicator value (Ellenberg 1988). Species with low Ellenberg values (1 and 2), which might be expected to have diminished as a result of increased atmospheric nitrogen deposition, were as frequent in the 'no significant change' category as in the group of species which showed a decrease in cover.

There were no significant differences ($P>0.05$) in soil organic matter content, extractable K, Ca and P between soils sampled in 1970 and 1990, whereas soil pH increased from 7.5 to 7.8 ($P<0.001$), total N increased from 1.069 to 1.169 ($P<0.001$), total P increased from 0.128 to 0.147 ($P<0.001$) and extractable Mg decreased from 19.55 to 16.56 ($P<0.001$). Total annual nitrogen deposition for that part of southern England which includes Wiltshire is estimated to be between 16 and 20 kg N ha^{-1} yr^{-1} (Williams *et al.* 1989). This value lies within the range (14–25 kg N ha^{-1} yr^{-1}) suggested by Bobbink *et al.* (1992) as the critical load for nitrogen in calcareous grassland. It should be stressed that the conclusions of Bobbink *et al.* are based largely on studies in The Netherlands in which the dominant species was tor grass. It has been suggested that other grasses might behave similarly. The results obtained from Parsonage Down do not support this contention. There is no evidence of substantial changes in the floristic composition of the grassland in the period 1970–90. All species which were present in 1990 were still present in 1990, including 18 species with Ellenberg nitrogen indicator values of 1 and 2 which should have been most susceptible to increased nitrogen deposition.

A possible explanation for the apparent stability (at the macro-scale) of the grassland at Parsonage Down is that any possible advantage which the grasses might expect to gain from their potential ability to utilise incoming nitrogen more effectively than forbs is negated by the grazing animal which reduces their competitive ability by defoliation.

The calcareous nature of the soil, with its huge reserves of calcium, makes it unlikely that acidification will occur in the short or long term as a result of increased nitrogen deposition. It seems likely that any increase in atmospheric N will be cycled in the soil/vegetation/ animal system, *provided the current system of management, using the grazing animal, is maintained.*

T C E Wells

References

Bobbink, R. & Willems, J.H. 1987. Increasing dominance of *Brachypodium pinnatum* (L.) Beauv. in chalk grasslands: a threat to a species-rich ecosystem. *Biological Conservation*, **40**, 301–314.

Bobbink, R., Boxman, D., Fremstad, E., Heil, G., Houdijk, A. & Roelofs, J. 1992. *Critical loads for nitrogen eutrophication of terrestrial and wetland ecosystems based upon changes in vegetation and fauna.* (Report.) Copenhagen: Nordic Council of Ministers.

Ellenberg, H. 1988. *Vegetation ecology of central Europe.* Cambridge: Cambridge University Press.

Wells, T.C.E. 1985. The botanical and ecological interest of ancient monuments. In: *Archaeology and nature conservation*, edited by G. Lambrick, 1–9. Oxford: Dept for External Studies, University of Oxford.

Williams, M.L., Atkins, D.H.F., Bower, J.S., Campbell, G.W., Irwin, J.G. & Simpson, D. 1989. *A preliminary assessment of the air pollution climate of the UK.* (Report LR723(AP).) Stevenage: Warren Spring Laboratory.

The ecological effects of liming

(This work was partly funded by the Joint Environment Programme of National Power and PowerGen, and the Nature Conservancy Council, and involved collaboration with Dr S Mackenzie of the Department of Environmental Biology, University of Manchester)

In many countries, acid deposition has led to progressive acidification of surface waters, which has been linked to reductions in numbers of aquatic invertebrates and fish. Upland areas of the UK, including extensive areas of blanket bog and moorland, have been the most severely affected. Liming the catchment areas (watersheds) around lakes is one way to mitigate the effects of acidification; soil water and surface runoff is neutralised before it reaches the lake, and the quality of the lake water is subsequently improved (Howells & Dalziel 1992). This technique is appropriate for lakes with short retention times where direct liming of the water is ineffective.

Blanket bog and moorland are naturally acidic, and there has been concern from conservation interests that liming may be detrimental to the terrestrial ecosystem. In small-scale field trials on blanket bog,

Plate 32. Field plots at Llyn Conwy, north Wales

Table 10. The numbers of pygmy shrews and common shrews captured inadvertently in pitfall traps on limed and control experimental plots in channels, flushes and ombrogenous parts of the blanket bog at Llyn Conwy

Plot type	Months after application of lime	Pygmy shrews captured per trap		Common shrews captured per trap	
		Lime	Control	Lime	Control
Channel	0-3	1.75	4.0*	1.05	2.4*
	4-6	0.3	1.5*	0.4	0.4
	11-18	0.2	0.7	0.9	0.5
Flush and ombrogenous	7-13	0.3	1.2*	0.2	0.2
	18-24	2.3	3.3	0.3	1.0
	7-24	2.6	4.5*	0.5	1.2

* Indicates that capture success on the limed and control plots was significantly different (chi-squared test: $P<0.05$ at least)

liming killed some of the *Sphagnum* species and reduced the abundance of some invertebrate groups. Preliminary studies also demonstrated that the total numbers of shrews inadvertently captured in pitfall traps were significantly lower on limed than on control (not limed) plots (Mackenzie, Lee & Wright 1990).

Pygmy shrews (*Sorex minutus*) and common shrews (*S. araneus*) dominate the small mammal fauna on blanket bog and moorland, and are an important component of the terrestrial ecosystem; they are voracious predators of invertebrates and can have a major impact on invertebrate communities (Churchfield, Hollier & Brown 1991). The aim of the present study was to determine whether liming moorland affected shrew abundance and, if so, to investigate the mechanism.

Shrews were trapped on blanket bog moorland at Llyn Conwy (Plate 32), north Wales, and on acidic moorland at Loch Fleet, Scotland, and Llyn Brianne, mid-Wales. At Llyn Conwy, traps were set in 26 small (4–25 m^2) limed and control plots located in channels, flushes and in ombrogenous parts of the bog. At Loch Fleet and Llyn Brianne, small mammals were trapped on six pairs of larger (0.5–34 ha) limed and control plots; five pairs of plots were on acidic moorland, the other was under a planted conifer forest. Liming methods and dates of application varied between sites (Shore & Mackenzie 1993).

Pygmy shrews were adversely affected by liming. At Llyn Conwy, fewer animals than expected were captured on the limed plots (Table 10) up to between one and two years after liming. The small size of the Llyn Conwy plots, relative to shrew home range size, meant that lower numbers of captures on limed plots reflected a reduction in activity rather than in abundance. On the large plots at Loch Fleet and Llyn Brianne, captures of pygmy shrews were 30–55% lower on limed moorland plots than on controls (Figure 43); only under the conifer forest were shrews captured in equal numbers on the limed and control areas. Because these plots were large, trap captures provided some index of abundance combined with activity, and these results suggested that pygmy shrews inhabit limed areas but in reduced numbers. The duration of lime-mediated reductions in pygmy shrew abundance was highly site-specific, varying between only five months to at least three years on the sites examined in the present study. This variability presumably reflected differences between sites in hydrology, vegetation structure and liming methods, each of which may alter the effect and persistence of lime.

Liming had little impact on common shrews. At Llyn Conwy, common shrews were captured less frequently than expected on limed plots but, in contrast to pygmy shrews, only for the first three months after the application of lime (Table 10). At Loch Fleet and Llyn Brianne, common shrews tended to be trapped more frequently on limed plots than on controls (Figure 47), but this difference in numbers between the two types of plot was not statistically significant. Investigation of the population dynamics of common shrews on an experimentally limed 0.5 ha plot and an equivalent control area at Llyn Conwy revealed no effects of lime on common shrew numbers and overwinter survival (Shore & Mackenzie 1993).

The mechanism by which liming affects pygmy shrews may be a reduction in prey availability. The number of pygmy shrews captured at Llyn Conwy was positively correlated with the numbers of Opiliones, Araneae, Heteroptera, Coleoptera and Diptera captured in pitfall traps. These invertebrates comprise the bulk of the diet of pygmy shrews on peat moorland (Butterfield, Coulson & Wanless 1981) and their abundance can be reduced by lime (Mackenzie *et al.* 1990). However, common shrews take similar invertebrate prey to pygmy shrews on peat bog but were little affected by liming. In an attempt to explain this apparent anomaly, we examined the effect of lime on the abundance of invertebrates of different size because pygmy shrews generally take smaller prey than the larger common shrew. We found that fewer small invertebrates (0–6 mm body length) were captured on limed plots at Llyn Conwy than on the controls. In contrast, there was no difference between treatment and control plots in the numbers of large invertebrates (6–12 mm body length) trapped. These results suggest that liming reduced the availability of small invertebrates preyed upon by pygmy shrews, and so may have indirectly

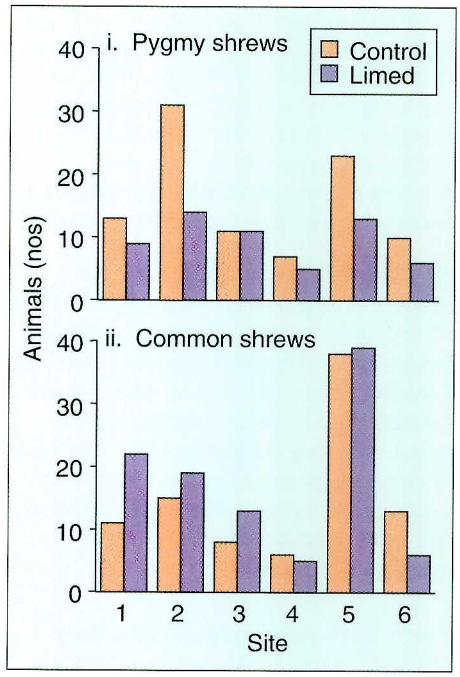


Figure 47. Number of (i) pygmy shrews and (ii) common shrews captured on control and limed plots on six pairs of sites at Loch Fleet and Llyn Brianne. The number of individuals captured on limed and control plots was significantly different (matched sets test: $P<0.05$) for pygmy shrews but not common shrews

reduced the abundance of the shrews themselves. In contrast, liming did not alter the availability of the larger prey taken by common shrews, and this fact may account for why lime had little effect on this shrew species.

We do not know the way in which lime has an impact on invertebrates or why small rather than large individuals are affected, and these aspects merit further study.

R F Shore

References

Butterfield, J., Coulson, J.C. & Wanless, S. 1981. Studies on the distribution, food, breeding biology and relative abundance of the pygmy and common shrews (*Sorex minutus* and *S. araneus*) in upland areas of northern England. *Journal of Zoology,* **195,** 169–180.

Churchfield, S., Hollier, J. & Brown, V.K. 1991. The effects of small mammal predators on grassland invertebrates, investigated by field exclosure experiment. *Oikos,* **60,** 283–290.

Howells, G. & Dalziel, T.R.K., eds. 1992. *Restoring acid waters: Loch Fleet 1984–1990.* Barking: Elsevier.

Mackenzie, S., Lee, J.A. & Wright J.M. 1990. *Ecological impact of liming blanket bog.* (Report to the Nature Conservancy Council.) Peterborough.

Shore, R.F. & Mackenzie, S. 1993. The effects of catchment liming on shrews *Sorex* spp. *Biological Conservation,* **64**, 101–111.

Pesticide interactions

(This work was partly funded by the Department of the Environment, and involved collaboration with the University of Reading)

Before any pesticide can be registered for use, it has to undergo a rigorous series of tests to establish its toxicity to non-target organisms and to determine its environmental fate. In such tests, each pesticide is treated individually. In practice, wildlife on agricultural land will be exposed to combinations of pesticides, either because more than one has been applied to a particular crop (together or sequentially), or because an animal has moved from one crop to another. Birds are particularly susceptible to the latter situation. In most cases, the effects of a combination of pesticides will simply be additive, but, in some cases, the combined effect is much greater than would be expected from their individual toxicities (synergism). Clearly, it is totally impractical to test every possible combination of pesticides, but with a knowledge of how pesticides behave biochemically it is possible to select combinations which could cause potential problems.

Broadly, synergisms can occur in two ways. The first is where one pesticide increases the activation of another, and the second is where one pesticide decreases the detoxification of another. In a joint study with Drs Johnston and Walker of the University of Reading, ITE has been examining both types of synergism. The study began by examining an example of the former type of interaction, in which a fungicide increased the activation, and hence the toxicity, of an insecticide.

Most modern pesticides are complicated molecules which interfere with the biochemistry of their target organism. Ideally, they are made to be specific, ie non-toxic to non-target organisms, by designing the molecule so that it affects an aspect of biochemistry which is unique to the target organism(s). The fungicide we used (prochloraz) is one of a group of fungicides known as ergosterol biosynthesis inhibiting (EBI) fungicides. As the name suggests, they act by inhibiting the synthesis of a sterol. The sterol is essential to fungi but not to other organisms, and so EBIs are specific; they are toxic to fungi but not to other organisms. The insecticide (malathion) is one of the organophosphorus insecticides. They inhibit cholinesterases, which are essential in all organisms. However, these insecticides are of comparatively low toxicity to vertebrates because vertebrates, unlike insects, have an enzyme which can rapidly detoxify the insecticide. In insects, instead of being detoxified, all of the insecticide is transformed from the inactive molecule, which is applied to the crop, to an active toxic molecule which potently inhibits cholinesterases.

Although these two pesticides are themselves good specific pesticides with comparatively low toxicity to vertebrates,

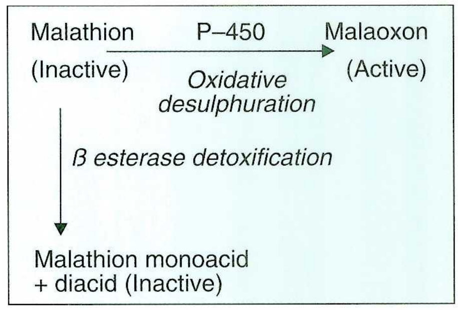


Figure 48. Activation of malathion by P-450 mono-oxygenases. In birds and mammals, but not insects, most malathion is normally rapidly detoxified by β esterases, but, if P-450s have been induced, a higher proportion is converted to the active toxic malaoxon

the combination of the two can have toxic consequences in laboratory studies, because a side effect of exposure to EBI fungicides is the induction of a family of enzymes called P-450s, ie the amount of these enzymes increases markedly. This is not a harmful consequence: it is a natural reaction to a variety of foreign compounds. However, the enzymes which transform organophosphorus insecticides from their inactive form to their active toxic form are also P-450s. So, if a bird, for example, which has undergone P-450 induction as a result of exposure to an EBI, is exposed to an organophosphorus insecticide, instead of most of that insecticide being rapidly detoxified, much more than usual will be transformed to its toxic form (Figure 48).

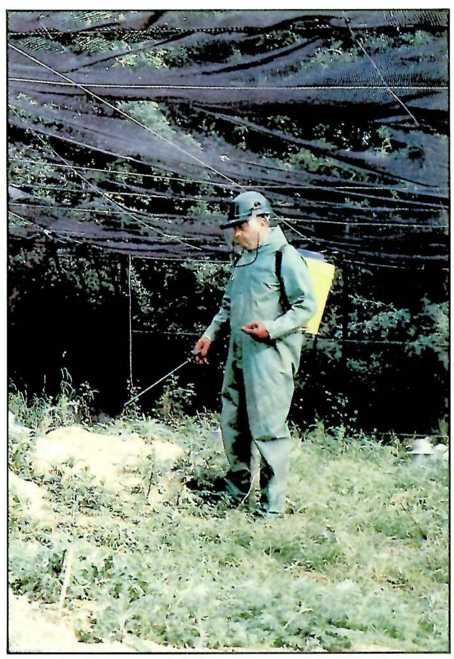


Plate 33. Spraying enclosures with the organophosphorus insecticide malathion, to determine whether toxicity is increased in partridges which have eaten fungicide-dressed wheat

reduced the abundance of the shrews themselves. In contrast, liming did not alter the availability of the larger prey taken by common shrews, and this fact may account for why lime had little effect on this shrew species.

We do not know the way in which lime has an impact on invertebrates or why small rather than large individuals are affected, and these aspects merit further study.

R F Shore

References

Butterfield, J., Coulson, J.C. & Wanless, S. 1981. Studies on the distribution, food, breeding biology and relative abundance of the pygmy and common shrews (*Sorex minutus* and *S. araneus*) in upland areas of northern England. *Journal of Zoology*, **195,** 169–180.

Churchfield, S., Hollier, J. & Brown, V.K. 1991. The effects of small mammal predators on grassland invertebrates, investigated by field exclosure experiment. *Oikos*, **60,** 283–290.

Howells, G. & Dalziel, T.R.K., eds. 1992. *Restoring acid waters: Loch Fleet 1984–1990.* Barking: Elsevier.

Mackenzie, S., Lee, J.A. & Wright J.M. 1990. *Ecological impact of liming blanket bog.* (Report to the Nature Conservancy Council.) Peterborough.

Shore, R.F. & Mackenzie, S. 1993. The effects of catchment liming on shrews *Sorex* spp. *Biological Conservation*, **64**, 101–111.

Pesticide interactions

(This work was partly funded by the Department of the Environment, and involved collaboration with the University of Reading)

Before any pesticide can be registered for use, it has to undergo a rigorous series of tests to establish its toxicity to non-target organisms and to determine its environmental fate. In such tests, each pesticide is treated individually. In practice, wildlife on agricultural land will be exposed to combinations of pesticides, either because more than one has been applied to a particular crop (together or sequentially), or because an animal has moved from one crop to another. Birds are particularly susceptible to the latter situation. In most cases, the effects of a combination of pesticides will simply be additive, but, in some cases, the combined effect is much greater than would be expected from their individual toxicities (synergism). Clearly, it is totally impractical to test every possible combination of pesticides, but with a knowledge of how pesticides behave biochemically it is possible to select combinations which could cause potential problems.

Broadly, synergisms can occur in two ways. The first is where one pesticide increases the activation of another, and the second is where one pesticide decreases the detoxification of another. In a joint study with Drs Johnston and Walker of the University of Reading, ITE has been examining both types of synergism. The study began by examining an example of the former type of interaction, in which a fungicide increased the activation, and hence the toxicity, of an insecticide.

Most modern pesticides are complicated molecules which interfere with the biochemistry of their target organism. Ideally, they are made to be specific, ie non-toxic to non-target organisms, by designing the molecule so that it affects an aspect of biochemistry which is unique to the target organism(s). The fungicide we used (prochloraz) is one of a group of fungicides known as ergosterol biosynthesis inhibiting (EBI) fungicides. As the name suggests, they act by inhibiting the synthesis of a sterol. The sterol is essential to fungi but not to other organisms, and so EBIs are specific; they are toxic to fungi but not to other organisms. The insecticide (malathion) is one of the organophosphorus insecticides. They inhibit cholinesterases, which are essential in all organisms. However, these insecticides are of comparatively low toxicity to vertebrates because vertebrates, unlike insects, have an enzyme which can rapidly detoxify the insecticide. In insects, instead of being detoxified, all of the insecticide is transformed from the inactive molecule, which is applied to the crop, to an active toxic molecule which potently inhibits cholinesterases.

Although these two pesticides are themselves good specific pesticides with comparatively low toxicity to vertebrates,

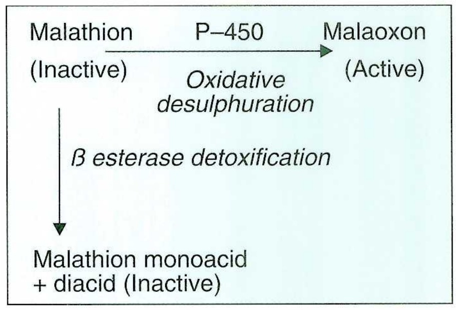


Figure 48. Activation of malathion by P-450 mono-oxygenases. In birds and mammals, but not insects, most malathion is normally rapidly detoxified by β esterases, but, if P-450s have been induced, a higher proportion is converted to the active toxic malaoxon

the combination of the two can have toxic consequences in laboratory studies, because a side effect of exposure to EBI fungicides is the induction of a family of enzymes called P-450s, ie the amount of these enzymes increases markedly. This is not a harmful consequence: it is a natural reaction to a variety of foreign compounds. However, the enzymes which transform organophosphorus insecticides from their inactive form to their active toxic form are also P-450s. So, if a bird, for example, which has undergone P-450 induction as a result of exposure to an EBI, is exposed to an organophosphorus insecticide, instead of most of that insecticide being rapidly detoxified, much more than usual will be transformed to its toxic form (Figure 48).

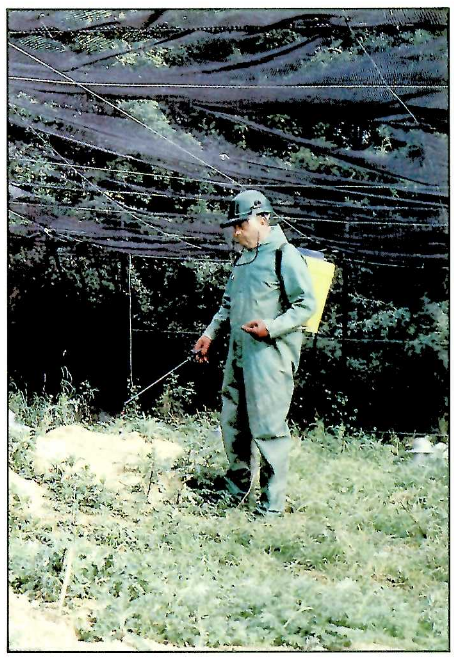


Plate 33. Spraying enclosures with the organophosphorus insecticide malathion, to determine whether toxicity is increased in partridges which have eaten fungicide-dressed wheat

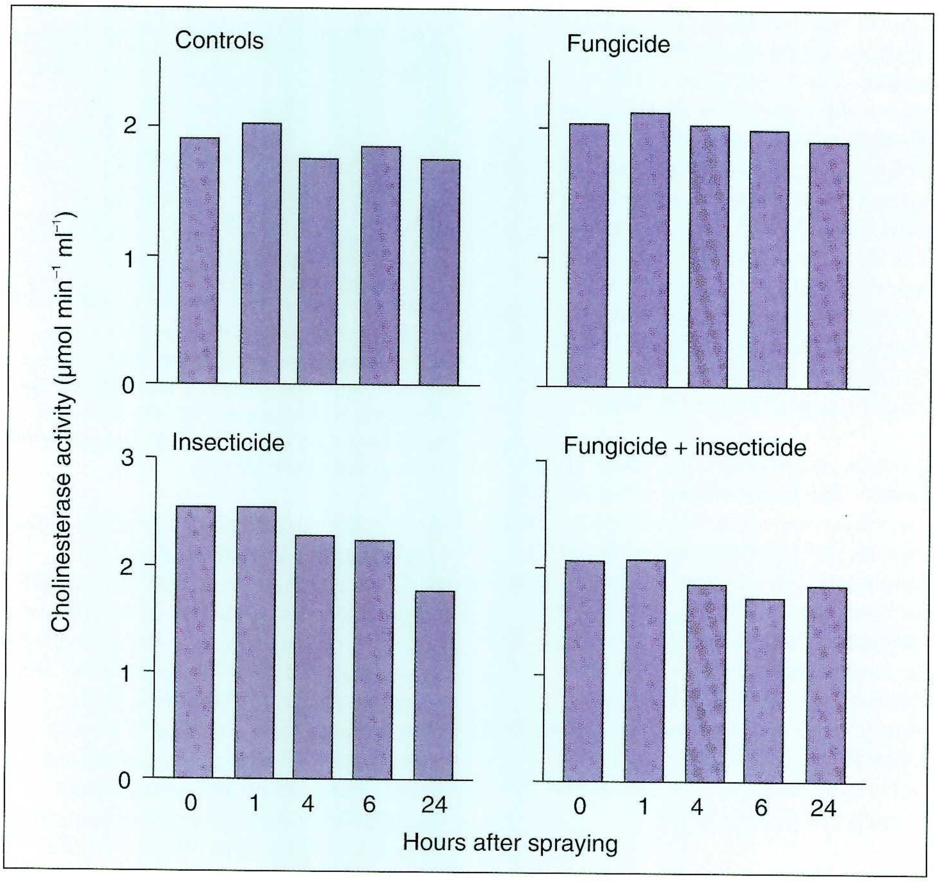


Figure 49. Changes in blood cholinesterase activity in control partridges and partridges sprayed with the organophosphorus insecticide malathion. Half of the partridges had eaten fungicide-dressed wheat during the week before spraying. The insecticide caused significant cholinesterase inhibition, but previous exposure to the fungicide did not cause significantly increased inhibition

Initial laboratory studies on our test species, the red-legged partridge (*Alectoris rufa*), showed that exposure to an EBI dramatically enhanced the toxicity of organophosphorus insecticides. The next, and important, question is whether this interaction is likely to have significant consequences in the field. There are two aspects to this question: first, whether, under normal agricultural practice, these two pesticides are likely to be encountered in the same area at roughly the same time, and, second, whether application rates of the pesticides to crops are such that wildlife would be exposed to levels high enough to result in a synergistic interaction. We examined the latter aspect by conducting a semi-natural field trial in which birds were exposed to levels of the two pesticides as high as is ever likely to occur. Partridges were exposed to the fungicide, in the form of dressed wheat, followed by the insecticide applied as a spray.

Partridges were divided into four groups of ten. Each group was put into a 10 m x 5 m enclosure on pasture. Birds in two of the enclosures were provided with wheat dressed with fungicide containing prochloraz (donated by Schering Agrochemicals). In the other two enclosures, birds were given undressed wheat of the same variety. One week later, two of the enclosures (one with dressed wheat and one with untreated wheat) were sprayed with malathion insecticide at the application rate recommended for horticultural use (Plate 33). Blood samples were taken from all birds before, and 1, 4, 6 and 24 hours after spraying, and were assessed for cholinesterase activity.

Both groups sprayed with insecticide showed highly significant inhibition of cholinesterase activity ($P<0.0001$); the unsprayed groups did not. In the insecticide-sprayed group which had untreated wheat, cholinesterase was significantly inhibited after 24 hours. In the insecticide-sprayed group which had been given fungicide-dressed wheat, there was significant inhibition after only 4 hours. However, the inhibition was not significantly greater, at any time, in those given dressed wheat and sprayed with insecticide than in those which were sprayed only (Figure 49). The birds demonstrated no abnormal behaviour and none died. In fact, maximum cholinesterase inhibition was about 20%, compared to a lethal inhibition of about 80%.

This result suggests that this combination of pesticides probably does not present a threat to wildlife; birds given prochloraz-dressed wheat showed no marked increase in susceptibility to the insecticide malathion, in contrast to an earlier laboratory trial, in which birds fed on fungicide-dressed wheat and then given an oral dose of insecticide did show increased susceptibility. The difference is probably because, in the semi-natural trial, birds were in lush vegetation and therefore had natural food available to them as well as dressed wheat. Indeed, the amount of wheat consumed was considerably less than in laboratory trials when wheat alone was available. This is probably why later analysis of the birds revealed that, whilst the fungicide had caused P-450 induction, the level was much less than in earlier studies.

It is unlikely that this combination of pesticides in this particular species represents the worst possible scenario. We are, therefore, doing further research with different species, different combinations of EBIs and insecticides, and pesticide combinations in which one pesticide inhibits the detoxification of others. We are also examining the potential indirect effects of P-450 induction on steroid hormones, as P-450s are involved in the biosynthesis and metabolism of these hormones.

A S Dawson

References

Johnston, G., Collett, G., Walker, C., Dawson, A., Boyd, I. & Osborn, D. 1989. Enhancement of malathion toxicity to the red-legged partridge following exposure to prochloraz. *Pesticide Biochemistry and Physiology,* **35,** 107–118.

Johnston, G., Walker, C., Dawson, A. & Furnell, A. 1990. Interactive effects of pesticides in the hybrid red-legged partridge. *Functional Ecology,* **4,** 309–314.