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Lichens in a changing pollution environment

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Lichens in a changing pollution environment

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Edited by

Peter Lambley
Pat Wolseley

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Preface

This volume presents the results of a workshop on *Nitrogen in the Environment* held at the Field Study Centre at Nettlecombe Court, Taunton, Somerset from 24-27 February 2003. A changing pollution environment across Europe has led to concern that increasing levels of nitrogen compounds are now affecting lichen communities in both urban and rural environments. Interest has focused on two main aspects, the use of lichens as bioindicators and potential threats to lichen species and their communities. Concern within English Nature that nitrogen may affect lichen-rich sites of conservation importance led to the provision of funding to the British Lichen Society to run a workshop to address these issues and to make recommendations for further work. The workshop was organised from the Natural History Museum, London, and the grant allowed us to bring together specialists concerned with nitrogen from five European countries.

There were 32 participants from Britain, France, the Netherlands, Germany and Italy, representing both researchers and agencies concerned with changing environmental conditions. The West Country location allowed participants time to visit the medieval parkland around Nettlecombe Court and the Institute for Grassland and Environmental Research (IGER) at North Wyke in Devon where a preliminary study of epiphytic lichens had been undertaken for DEFRA. This allowed participants time for discussion on agricultural implications of nitrogen and on the effects of variations in climatic conditions on lichen growth and distribution.

This publication includes most of the papers presented within the four sessions and a plenary address by Professor Mark Seaward providing an overview of the use of lichens in detecting changes in the pollution environment. *A changing lichen flora* reflects ongoing changes in nitrogen deposition across Europe (van Herk), and approaches used to detect and map aerial pollution in urban areas using cryptogams from France, Germany (Kricke & Feige, Stapper & Franzen-Rejer), Italy (Loppi, Giordani) and Britain (Davies). *The pollution environment* presents the policies of UK and European strategies to controlling air pollution (Vipond) and the concerns of the statutory nature conservation agencies (Whitfield & Bareham), followed by spatial distribution of atmospheric nitrogen (Sutton and others) and an outline of changes in agricultural use of both N and P including ways of reducing impacts of N on the environment (Chadwick & Scholefield). An address by Scheidegger (published elsewhere) in *Selecting and monitoring species and communities* outlined preliminary results of a large European project to assess changes in lichen diversity with land use. Other papers include the selection and use of indicator species in environmentally variable regions of Italy (Giordani) and a wider look at the global distribution pattern of nitrophytic lichens (Aptroot). The paper on the experimental effects of NH_3 on an acid heathland lichen *Cladonia portentosa* (Sheppard and others) contrasts with a community-based sampling strategy to detect changes in epiphytic lichen communities due to increased NH_3 deposition (Wolseley & James). A session on *Conserving lichen communities and species diversity* provided an overview from the site conservation point of view (Woods, not included here); the important contribution made by wayside trees to lichen distribution (Edwards) and an overview of the potential threats to SSSIs in England (Lambley and others). Several other papers are absent from this volume and will be published elsewhere; Frantzen-Reuter on nitrogen metabolism in lichens (PhD thesis), Ellis & Crittenden on using heathland lichens to map nitrogen and acid deposition, and Headley and others on changes in lichen distribution with atmospheric deposition of nitrogen.

Questions raised during each session were discussed in four groups and the results presented to participants. These are outlined at the end of the papers for each session which led to a general discussion on further developments to establish lichens as indicators and associated research methodologies.

The organisers wish to thank all those people and organisations who helped to make this meeting a success; the Natural History Museum for providing facilities from which to organise this workshop in particular Gill Stevens who co-ordinated everything while Pat Wolseley was away and who together with Jenny Duckworth of Plantlife ensured that the discussion groups targeted questions raised during the workshop. Mark Bolland and staff at Field Studies Centre provided us with excellent facilities and Sara Hayward helped everyone to travel to and from a rather remote place The day spent at IGER at North Wyke, with Professor Steve Jarvis and Jerry Tallowin provided us with a combination of interesting lichens in a place with a long history of agricultural use that led to some very good discussions. Most of all we would like to thank participants for their contribution to the meeting and to the authors who have contributed papers to this volume and we hope that we have provided a faithful record of the meeting and the discussions that took place. We would also like to thank Professor Clifford Smith for his immense help in helping to edit the final report. Many thanks also to Mary Roberts for the final typing and preparation of this volume.

Pat Wolseley and Peter Lambley

Contents

1. Plenary address

Lichens and hypertrophication

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The use of the term 'hypertrophication', rather than 'eutrophicated' ('eutrophiated' *sensu* Hawksworth & Rose 1970) to describe current lichen phenomena arising from excessive environmental enrichment of nutrients is strongly advocated. Whether lichens can be used for biomonitoring scales to determine levels of nutrient-enrichment under changing or indeed stable pollution burdens remains to be seen. Nitrogen is one of the commonest elements on our planet. Its natural occurrence in ecosystems and fundamental use in organisms require no elaboration, but you can have too much of a good thing! As a result of increasing human impact, nitrogen, in various forms, is one of many contaminants affecting the natural environment. To determine the average concentration of pollutants, including nitrogen, to which plants are exposed is highly complex due to the multiplicity of environmental factors implicated in their deposition, uptake and retention, as well as the sensitivity of the particular species to them. Studies on the dispersion of these pollutants over our planet, their chemical and physical transformation, their direct deposition as gases onto surfaces (dry deposition) and their removal from the atmosphere by precipitation (wet deposition) have received considerable attention over recent years due to short-term but significant quantitative and qualitative shifts in pollution burdens and their long-range (global) dispersion.

In the case of nitrogen, it is delivered as a primary pollutant in the form of ammonia or nitric oxide; ammonia neutralises SO_2 as SO_4^{2-} partially to NH_4HSO_4 or fully to $(NH_4)_2SO_4$, and nitric oxide is readily oxidised by ozone to form NO_2 (Fowler 2002). Further secondary pollutants and chemical complexes, in gaseous or aerosol phase, arise from oxidation products of the atmospheric chemistry of NO_X; the presence of NO_X is also instrumental in the generation of photochemical oxidants, particularly ozone, resulting from the oxidation of volatile organic compounds.

Ammonia occurs as a consequence of natural processes derived from animal waste and plant decomposition, its exchange with the atmosphere is analogous with CO_2 exchange in which a compensation point defines the concentration at which no net exchange occurs; clearly this compensation point is exceeded by modern agricultural and animal husbandry practices and NH_3 is deposited. As levels of SO_2 and NO_X decline due to control measures, NH_3 becomes more important as a contributor to eutrophication, and indeed hypertrophication.

NO_X emissions are mainly composed of NO, which is readily oxidised to NO_2; for plants, the latter is regulated by stomata, but for lichens and bryophytes its effects are not controlled. Furthermore, in very low doses, NO_X can stimulate plant growth; the switch to toxicity occurs rapidly as the dose increases, but no reliable data for a dose-response curve are available, since the impacts of NO and NO_2 may be fundamentally different at the cellular level (Mansfield 2002). In the past, NO_2 was found to be considerably less toxic than SO_2 and O_3, and since it was very rarely likely to be encountered alone, its phytotoxicity was usually considered only in the context of its interactions with other pollutants. The recognition that low doses of NO_2 would be beneficial in supplying N to nitrogen-deficient soils, has more recently been superseded by consideration of the ecological consequences of additional N deposition (NO_X and NH_3) which are both complex and rarely considered to be advantageous in the long-term (Bobbink & Lamers 2002).

Until recently, due to the overriding effects of SO_2, other than the researches of Nash and colleagues on the impact of oxidant air pollution in western North America (eg Nash 1988, Boonpragob, K. and others 1989), little work has been undertaken on the response of lichens to primary and secondary products resulting from NO_X and NH_3 emissions. More recently, the damaging effects of N compounds on lichens have been noted in many European countries, but research on the topic has been mainly concentrated on The Netherlands and Denmark, where, since the early 1980s, ammonia emissions, particularly from animal husbandry, have been shown to be a, if not the, major influence on the ecology and distribution of epiphytic lichens (eg de Bakker 1989, van Dobben & de Bakker 1996, van Herk 1999, van Herk and others 2002). The long-term monitoring programme in The Netherlands has also detected that some of these changes would suggest that lichens are also responding to global warming (van Herk and others 2002).

Generally speaking, from the 1970s onwards, when SO_2 levels were falling but NH_3 emissions were rising, nitrophilous lichens progressively replaced acidophilous species, which, according to de Bakker (1989), was due to the rise in pH of the bark rather than the nitrogen supply. The re-establishment of lichens in areas, mainly urban, formerly polluted by SO_2 but now adopting clean policy has been dramatic over a relatively short time-scale; lichen recovery is not immediate, necessitating a time-lag, modelled in one instance to be about 5 years (Henderson-Sellers & Seaward 1979). Here the once-more available wide range of urban habitats now subjected to new atmospheric burdens, including hypertrophication, are exploited to great advantage (Seaward 1997). Further factors influencing these changes include the widespread use of garden chemicals, particularly fertilizers, industrial and transport emissions, and de-acidification of tree bark brought about by increasing NH_3 deposition and alkaline dusts.

Similar impacts on the epiphytic lichen flora are to be observed in rural areas; these effects are, as yet, less dramatic in the United Kingdom than those encountered in The Netherlands. However, it is clear that the lichen floras of both countries are greatly influenced by agricultural practices, the effects of which were formerly not manifested due to over-riding blanket SO_2 pollution. The effects of agrochemicals and fertilizers on lichens are reviewed by Brown (1992), who highlights the complexity of chemical and physical conditions involved. Further agricultural practices, such as slurry treatment of land and animal husbandry, bring about environmental hypertrophication which profoundly affects lichen floras; in addition, waste-treatment, industrial and quarrying activities exacerbate this effect.

Rather interestingly, the resultant lichen floras are not a new phenomenon: the effects of this type of pollution have influenced natural habitats since the development of human settlements and the domestication of plants and animals. While there are no records to show how such contamination affected lichen floras, it is clear from an examination of late 18th and early 19th century collections in herbaria that substrata, particularly lignum (which was then more widely used as building and fencing material), were impregnated with nutrient-rich chemicals.

Despite the ever-increasing use of agrochemicals over the next century or more, this effect became less noticeable due to the overriding influence of an acidic environment generated to a large extent by SO_2 emissions, the quantitative and qualitative delivery of which changed over time. The implementation of clean air policies in recent decades has undoubtedly led to a dramatic improvement in lichen floras, but the effects of hypertrophication, so long masked by acidic conditions, and more recently exacerbated by increases in NO_X, are clearly dictating the nature of emerging lichen assemblages, not only at a local level but also regionally. Widescale changes in agricultural practices, especially the use of agrochemicals and the expansion in animal husbandry, have impacted on the lichen flora: noticeable expansions in the range and extensions in habitat preference of many species in the British Isles have been documented in recent floras (Purvis and others 1992, Dobson 2000), atlases (Seaward 1995 on-going) and biogeographical studies (eg Seaward 1998, Coppins and others 2001).

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2. A changing lichen flora

The effects of short and long distance nitrogen deposition on epiphytic lichens

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Summary

Current knowledge concerning the effects of NH_3, NH_4^+, NO_x, and NO_3^- on epiphytic lichens in urban, rural and background situations is discussed. The effects of NH_3 are strongest near the animal sources. Severe NH_3 pollution results in communities with abundant nitrophytic and few acidophytic lichens. Several scales are tested, and are compared with NH_3 concentrations. NH_4^+ is particularly harmful to acidophytic lichens, and some like *Bryoria*'s disappear in background situations at extremely low deposition values.

Direct effects of gaseous NO_x on lichens are uncertain so far, but NO_x adds considerably to the NO_3^- deposition. NO_3^- in wet precipitation and in fine dust deposition is likely to have a strong influence on lichens in urban areas, and might also affect acidophytes in background situations. Non-eutrophicated urban trees with an acid bark have often remained very poor until now, although the lower SO_2 levels would permit a well-developed lichen vegetation. Recent urban studies suggest that lichen diversity counts seem to have lost their predictive value for SO_2 air pollution in this environment.

Introduction

Over the last 50 years, the European deposition levels of ammonia (NH_3), ammonium (NH_4^+), nitrogen oxides (NO, NO_2) and nitrate (NO_3^-) have increased considerably. At least some of these compounds are likely to affect epiphytic lichens, but the nature of the impact is not comparable (Table 1).

Table 1 Nitrogen compounds, likely to affect epiphytic lichens.

Compound:		**Source**	**Effect**
Ammonia	NH_3	Cattle	Basic, increases both bark pH and nitrogen content of bark
Ammonium	NH_4^+	Atmosphere	No effect on bark pH, but increases nitrogen content of bark
Nitrogen oxides	NO_x	Traffic	Acid, but not readily absorbed by bark
Nitrate	NO_3^-	Atmosphere	Increases nitrogen content of bark
Dust		Traffic, industry	Often basic, increases both bark pH and nitrogen content of bark

Ammonia is a basic compound, mainly originating from animal dung, and produced in large quantities by intensive cattle husbandry. It has been shown that gaseous ammonia increases bark pH up to 2 units (de Bakker & van Dobben 1988; van Herk 2001), especially close (< 1 km) to large sources. Ammonium is not emitted but arises in the atmosphere from ammonia. Ammonium is neither basic nor acid, and has therefore probably only slight pH effects. Ammonium in contrast to ammonia, adds particularly to background levels of dry and wet nitrogen deposition at medium to large distances from the source (> 10 km).

Traffic is the main source of nitrogen oxides (NO_x), which are acid by nature, but not readily absorbed by bark, and therefore not significantly affecting the bark pH like ammonia does. In the atmosphere, a relatively large proportion of the NO_x emission gets transformed into nitrogen loaded aerosol, finally to be deposited as NO_3^- in dust particles, or as dissolved NO_3^- in wet precipitation.

Ammonia and ammonium in the rural areas

The Netherlands have by far the highest ammonia and ammonium (NH_x) deposition levels in Europe, locally exceeding 3200 mol ha^{-1} yr^{-1} (Asman and others 1998). Adjacent Belgium and Germany, and parts of Denmark, England and France (Brittany) have deposition levels of c. 800-3000 mol ha^{-1} yr^{-1}. As a consequence, epiphytic lichen flora and vegetation of the Netherlands is strongly affected by NH_3. In rural areas with excessive NH_3-deposition, trees are strikingly covered by nitrophytic lichens like *Candelariella reflexa*, *Phaeophyscia orbicularis* and *Physcia caesia*, while *acidophytic* lichens such as *Lecanora conizaeoides* and *Hypogymnia physodes* are absent.

Nitrophytes can be characterized as species needing both a relatively high bark pH and at least some additional nitrogen as well. On acid bark the pH rather than nitrogen is a limiting factor for their occurrence. On basic bark nitrogen can be limiting but this is hardly the case in urban or rural circumstances, where nitrogen is usually widely available (see Table 1 in van Herk 2001).
Acidophytes need by definition an acid substrate, but many of them are sensitive to increased levels of nitrogen as well. It appears that some species disappear in background situations already at *c*. 0.3 mg N.l^{-1} in precipitation.

During recent decades, large parts of the Netherlands were mapped for the effects of NH_3 using the abundance of nitrophytes (expressed in NIW) and acidophytes (AIW) on roadside oak (*Quercus robur*) trees (van Herk 1999, 2002), and similar mappings have started in Germany (Franzen and others 2002). The Dutch programme involves some 6000 sites with 10 trees each. Most of the areas mapped are part of a monitoring programme with up to four resurveys so far. Changes in the occurrence of the lichen species in a representative area (Utrecht) over the last 22 years were published recently (van Herk and others 2002). Several different tree species (Figure 1) and methods to express the abundance of nitrophytes (Figure 2) have been tested so far by comparing this to ammonia pollution levels (see also Figure 6 (NIW) in van Herk 2001).

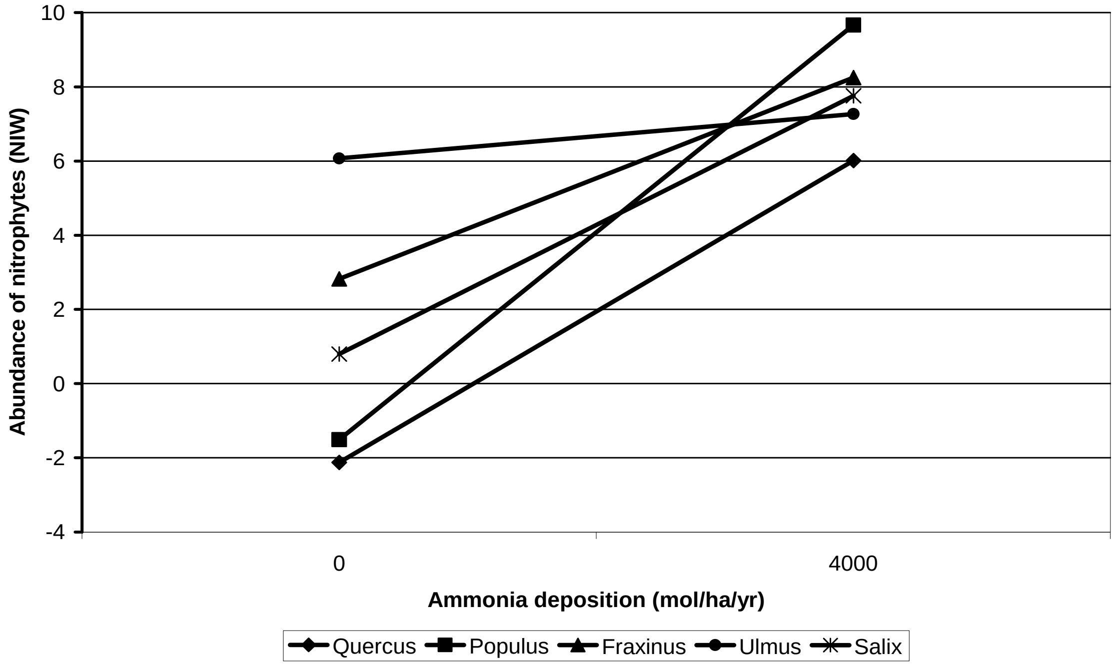
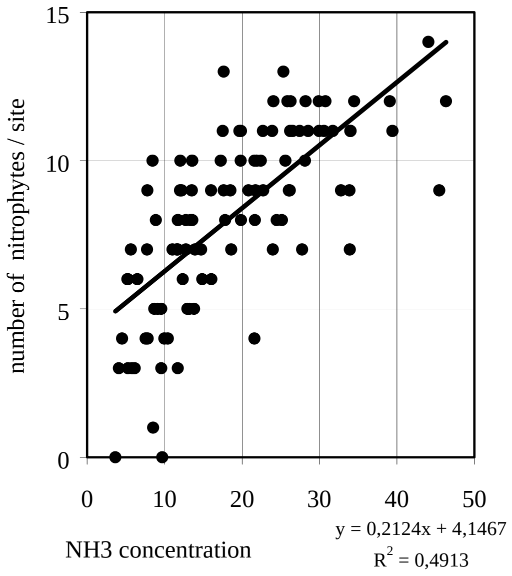


Figure 1
The abundance of nitrophytes (expressed as NIW) as a function of the annual ammonia deposition (mol/ha; grid estimates). Shown are regression lines for five different tree species: *Quercus robur*, *Populus* x *canadensis*, *Salix alba*, *Fraxinus excelsior* and *Ulmus* sp. Explained variances (R^2) 0.18 ($p<0.05$), 0.10 ($p<0.05$), 0.04 (n.s.), 0.03 (n.s.) and 0.00 (n.s.) respectively. Variances are generally low because only grid estimates based on cattle density were available. The differences between the tree species (R^2 and slope), however, show clearly whether each of these five tree species is suitable for mapping ammonia.



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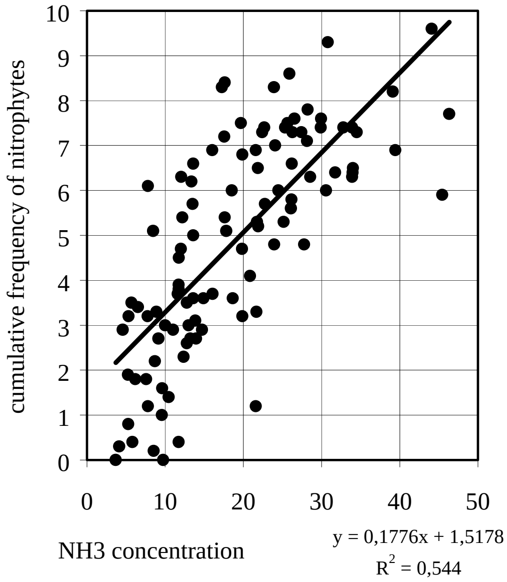

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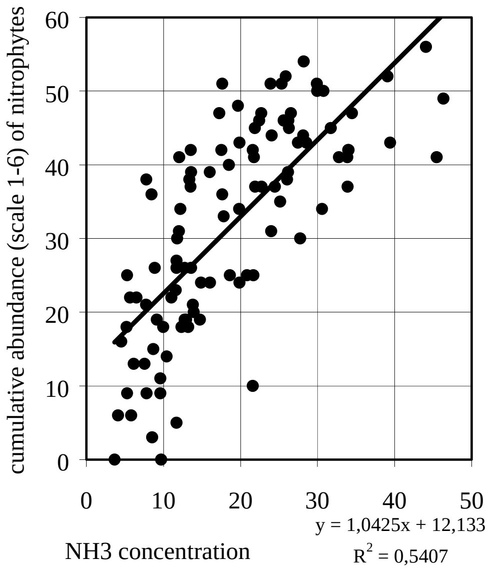

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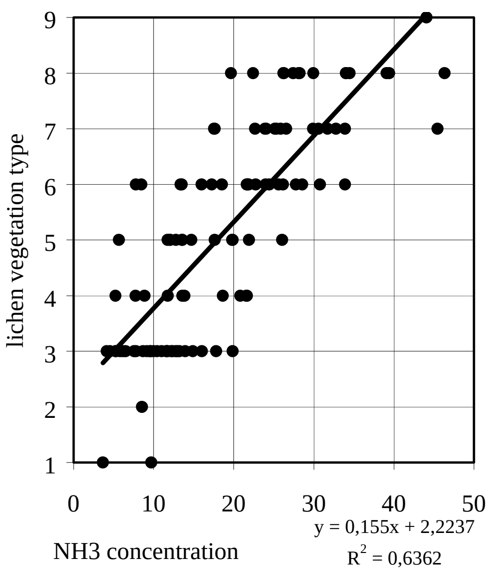

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Figure 2
Four different ways to express the abundance of nitrophytes as a function of the annual mean ammonia concentration:
A: Based on the number of nitrophytic species per site (each site with 10 trees).
B: Based on the cumulative frequency of nitrophytic species per tree.
C: Based on the cumulative abundance (scale 1-6) of nitrophytic species per site.
D: Based on the lichen vegetation type (see table 2).
Only sites with *Quercus robur* (n= 104), NH_3 concentrations are taken in situ using diffusion tubes over one year. Further methods and the species considered as nitrophytes: see Van Herk (2001). The explained variances (R^2) appear higher the more accurate the figures for total diversities and amounts of nitrophytes.

Table 2 Definition of 9 lichen vegetation types, as used in Figure 1D to estimate the NH_3 air concentrations (only used for rural *Quercus robur* sites; effects of dogs are excluded, and bark wounds avoided). Occasional: few thalli on few (<50%) trees; frequent: few thalli, but on most trees; abundant: large sheets (>1dm^2) on most trees

type 1	none of the species mentioned below present
type 2	*Physcia tenella* and/or *Xanthoria polycarpa* occasionally present, but none of the other species mentioned below
type 3	*P. tenella* and/or *X. polycarpa* frequently present, or... *Physcia adscendens, Xanthoria parietina* and/or *X. candelaria* occasionally present (but none of the others below)
type 4	*Lecanora dispersa* and/or *Phaeophyscia orbicularis* occasionally present
type 5	like type 4, but *Physcia dubia* and/or *P. caesia* occasionally present as well
type 6	*L. dispersa* and/or *P. orbicularis* frequently present
type 7	*L. dispersa*, *P. orbicularis* and/or *P. caesia* abundantly present, or *P. dubia* frequently present, or together with frequent (not abundant) *P. orbicularis* either *Phaeophyscia nigricans*, *Candelariella aurella, Caloplaca holocarpa* or *Rinodina gennarii* occasionally present.
type 8	besides abundant *L. dispersa*, *P. orbicularis* and/or *P. caesia,* also *P. nigricans*, *C. aurella*, *C. holocarpa* or *R. gennarii* occasionally present as well.
type 9	like type 8, but *P. nigricans*, *C. aurella*, *C. holocarpa* or *R. gennarii* frequently present.

Nitrogen oxides and nitrate in urban areas

In contrast to ammonia, for which a clear dose-effect relationship on lichens could be shown with decreasing distance from animal sources (van Herk 1999; Wolseley & James 2002), there has so far no or only a very weak short-distance relationship been shown between epiphytic lichens and the traffic density. For example, calculations carried out with some 400 Dutch roadside *Quercus* sites (only rural) showed that there was no different lichen flora present along busy roads (highways) compared to similar sites along quiet roads (van Herk 2001); existent differences could be attributed mainly to differences in SO_2 and NH_3 background levels, age of the trees, and the way in which the trees were managed (low branches present or not).

Urban areas of many Western European cities and towns have been invaded by nitrophytes on a large scale during the last decade. The role of nitrogen in this, especially traffic fumes, was discussed recently by many authors (Davies and others 2002; Purvis and others 2003; Vanholen 2003; Stapper & Kricke 2003). A correlation between the nitrogen concentration in two epiphytic lichens and the traffic density was found by Gombert and others (2003). It is, however, not very plausible to link the current spatial lichen patterns in urban areas directly with gaseous NO_x emissions. No clear "roadside" effect on the performance of lichen species, attributable to NO_x, has been demonstrated so far. This does not imply that effects of NO_x are non-existent: the current urban lichen patterns, particularly of the nitrophytes, probably show a clear positive correlation with patterns of basic dust pollution (in dry and wet precipitation) (Loppi, this volume), in which nitrate (originating from NO_x) is an important constituent. Recent measurements of fine dust appear to be particularly high in urban areas (Anon 2000), showing deposition patterns very much like the current distribution pattern of some nitrophytic lichens.

Long-distance effects

Nitrophytes have been shown to respond positively to ammonia, but a response to ammonium could not be shown. This distinction is most probably due to the basic properties of ammonia (lacking in ammonium). From acidophytes, however, it appeared that many of them are sensitive to both ammonia and ammonium (van Herk 2001), showing that they do need more than just acid conditions; many of them are sensitive to increased N-levels as well.

Lecanora conizaeoides was once a perfect indicator for SO_2 air pollution, but recently it has become rare in large areas. The main Dutch strongholds of this species that are left are in the coastal dune strip and in the centre of a few large forests, paradoxically the areas with lowest air pollution levels of both SO_2 and NH_x now. This shows that decreasing SO_2 levels are not the only reason why *L. conizaeoides* has disappeared; its current behaviour clearly shows a sensitivity to N compounds as well.

Until recently it was thought that the effects of nitrogen are quite localized in the areas around sources. However, analysis of data from 25 European ICP-IM monitoring sites, all situated far from emission sources, has shown that even in background situations the N-deposition has a marked influence on sensitive species (van Herk and others 2003). From acidophytic species like *Bryoria capillaris*, *Cetraria pinastri* and *Imshaugia aleurites* it could be shown that a decrease of their probability of occurrence can be observed already at 0.3 mg N/l in precipitation. Such deposition values are recorded now even in remote parts of Scandinavia. Unfortunately it appeared difficult to make a distinction between ammonium and nitrate due to strongly intercorrelated deposition patterns, but correlations with ammonium are strongest. It is, however, very possible that the total N-dose is decisive. It can be concluded that at medium to long distances from the source NH_4^+ and/or NO_3^- affect many *acidophytic* species just by adding N, particularly in wet precipitation.

Relationship to SO_2

In the near future more work is needed to determine how decreasing levels of SO_2 may be interfering with temporal changes of lichens responding to eutrophication, especially basic dust and NH_3. Many authors consider that recent recovery of all lichen species except *L. conizaeoides* is a result of better SO_2 air quality, but observations indicate that a recovery in acidified areas is slowed down or even absent if there is no eutrophication. Especially non-eutrophicated urban trees with acid bark are currently still very poor, although the SO_2 levels would permit well-developed lichen vegetation. This suggests that a natural recovery is inhibited, and only occurs where sufficiently high pH-values and nitrogen supply exist. Recent urban studies (eg Franzen 2001) suggest that lichen diversity counts seem to have lost their predictive value for SO_2 air pollution, and such counts, as eg proposed by Asta and others (2003), should be applied with caution for this purpose.

The spatial and temporal pattern of the abundance of nitrophytes in the Netherlands appears to be determined in the first place by ammonia, at least in the rural areas. No correlation could be shown with current levels of SO_2 (van Herk 1999, 2001). Lichen changes in a small area, intensively monitored since 1989 show that a decrease of the ammonia concentration by c. 30% (diffusion tubes) since 1997 has now been followed by a similar decrease of nitrophytes over that period (Figure 3).

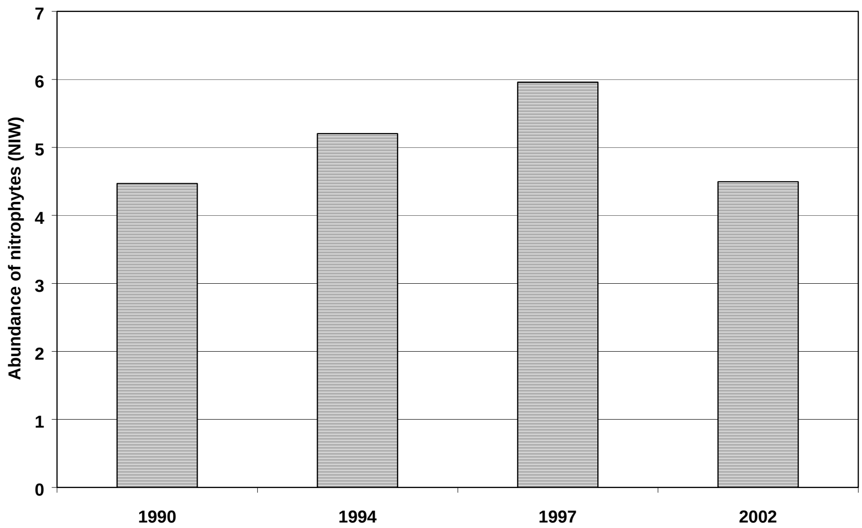

Figure 3
Changes in the abundance of nitrophytes (expressed as NIW) in the central part of the Netherlands (Gelderse Valley) between 1990 and 2002 (n= 146, all *Quercus robur*). All changes (adjacent years) are significant. Over the period 1997 – 2002 there has been a considerable decrease of nitrophytes, coinciding with a decrease of ammonia emission over that same period due to a clean air policy.

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A study of epiphytes in London under a climate of low sulphur dioxide

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Summary

This study records the distribution, diversity, frequency and community structure of corticolous lichens and bryophytes in London under a climate of low sulphur dioxide and increased nitrogenous compounds and other pollutants primarily generated by transport. A short review of air quality, the approach and methodology used for the study and results from the first eight surveys are presented.

Air quality in London

Following the completion of the review and assessment of London's air quality in accordance with EU and UK statutory requirements, the London Mayor published his Air Quality Strategy in 2002 (Greater London Authority). Concentrations of sulphur dioxide have fallen from their high of over 350 σg m^{-3} in the seventies to an average of 3 σg m^{-3} in 2001. Exceedances of the EU Objectives (EUO) for health and vegetation are not expected for this pollutant. A UK Objective of 266 σg m^{-3} as a fifteen-minute mean is however exceeded in some parts of the London primarily due to emissions from power stations and refineries operating along the banks of the Thames. Ozone has increased by an average of 15% (ERG, King's College 2002) peaking in 2000 with some exceedances of the EU Limit values for human health (8 hour running mean of 100 σg m^{-3}) mainly in the outer London boroughs.

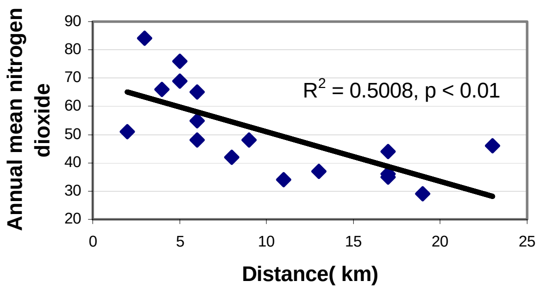

Figure 1. Annual mean nitrogen dioxide (σg m^{-3}) with distance from the city center.

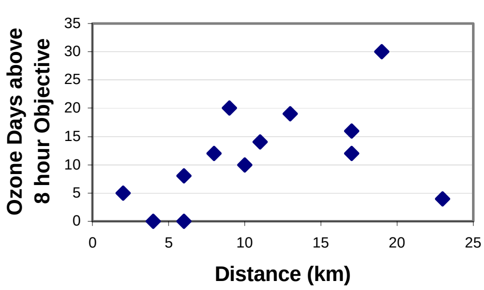

Figure 2. Ozone - Number of days exceeding the 8 hour EU Limit Value for human health.

Conversely the EUOs for human health for nitrogen dioxide and particulate matter (PM10) are widely exceeded. PM10 exceedances, expressed as the number of days exceeding the 24-hour EUO of 50 σg m^{-3}, occur mainly in the inner London boroughs and where major traffic routes converge. 106 days above the threshold were recorded in the central London borough of Westminster in 2001 with annual concentrations reaching 44 σg m^{-3}. Over the past five years average NO_2 concentrations in London have declined by 10% and NOx by 26%.

However, the annual mean EUO for NO_2 of 40 σg m^{-3} (21 ppb) was exceeded at the majority of kerbside and roadside monitoring sites (Figure 3) and many background sites in inner London. Annual mean concentrations in central London reached 60 σg m^{-3} (32 ppb) at some inner city sites exceeding the EUO for vegetation and sensitive ecosystems for NOx (NO and NO_2) by more than a factor of 2. The additional concentrations of NO mean that areas in all London boroughs exceed the annual mean NOx EUO of 30 σg m^{-3} (16 ppb). It is against this climate that epiphytic flora is investigated although the EHOs for vegetation are not applicable in urban areas for economic reasons.

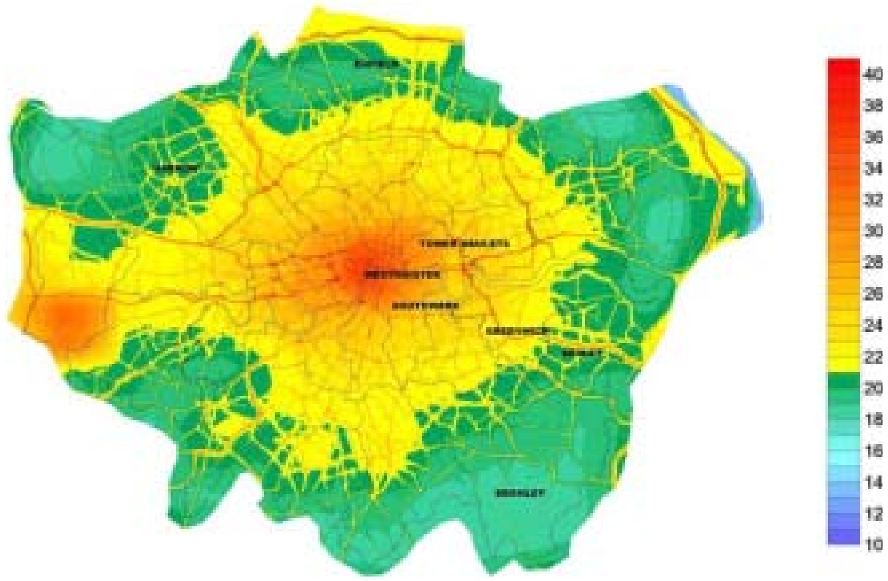

Figure 3. Modelled annual average concentrations of nitrogen dioxide in London (ppb). Eight survey sites are named. Source: Greater London Authority and Transport for London (2001) using 1997 weather data.

Approach

There are three elements to the main study. The first is a survey of epiphytes on ash trees (*Fraxinus excelsior*) at selected sites in 32 London boroughs and the City of London. These data will be mapped using a geographical information system and overlaid with maps of modelled concentrations of five pollutants (NOx, NO_2, PM10, sulphur dioxide and ozone) produced using ADMS-urban (CERC 2003) and regional climatic data. Canonical correspondence analysis will be used to investigate the relationship between the environmental variables, including bark pH and girth, and the epiphytes recorded. Thirdly, temporal trends in urban areas over the past few decades will be reviewed to identify major changes in diversity and community structure.

Methodology

Species diversity, frequency, and abundance were recorded on the trunks of ash between a height of 50 cm and 200 cm. Eight bark pH measurements were taken from each tree using a flat head electrode. Health and size of thalli were also recorded for selected species. The EU draft protocol for bio-monitoring was applied (Asta and others 2002, James and others 2002) to the first eight sites. However, the quadrat recording method (Figure 5) was used at the southern aspect only for the remainder of the survey due to time constraints. A range of different tree species were examined at selected sites to provide an indication of general trends in diversity in London at the present time.

Results from the pilot study were used to determine the optimum number of trees per site for the full survey. Groups of four trees, eight trees, ten and eleven trees (4 classes) were analysed in relation to the number of species recorded and the total diversity for the site (Figure 4).

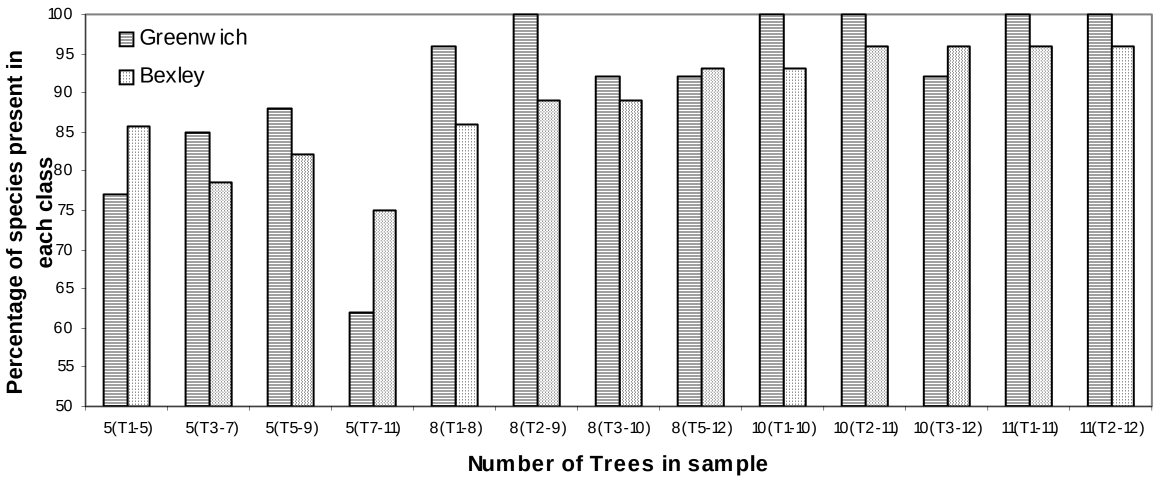

Figure 4: Establishing sample size (no of trees) per plot at Greenwich and Bexley. Four groups of trees corresponding to 5, 8, 10 and 11 trees per group were examined at each site.

The results indicate that at Greenwich all species were identified after sampling 8 trees whereas not all species were identified in Bexley after 11 trees had been surveyed. 85 % of species were identified at both sites when 8 trees were surveyed and 95% when 10 tree were surveyed. It was decided therefore to set the sampling size at a minimum of 8 trees per site.

Survey results from the first 8 sites

Early results indicate that the long anticipated increase in lichen diversity in London, hitherto extremely slow, has now gained pace with dramatic increases in the frequency and abundance of common and rare species. Major changes in community structure were also observed. Species richness on ash was significantly correlated (R^2 = 0.8091, P <0.001) with distance from central London, peaking in the northern boroughs (Figure 6). 12 bryophytes were recorded, mainly from suburban sites. (Pilot study sites are shown in Figure 3.)

Species numbers on oak were the highest for many decades and included several lichens new to central London. Of particular interest were *Pleurosticta acetabulum, Parmelina tiliacea,* and *Lecanora albella*, the latter a first record for London, the former rare in London. The abundance of fertile *Bacidia delicata* on plane (*Platanus hybrida)* was also unexpected, particularly on roadside trees where concentrations of oxides of nitrogen and particulate matter are highest.

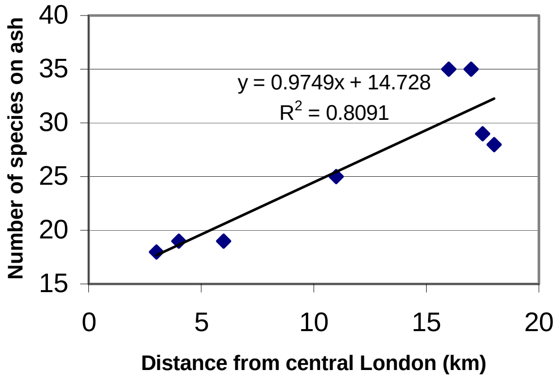

Figure 5. Laddered quadrat on tree at 1.50m

Figure 6. Species number v. distance from central London

A total of 51 lichen species were recorded on ash at the first 8 sites. The most commonly occurring species include many of those designated as nitrophytes in the monitoring scale developed in the Netherlands under conditions of high ammonia from livestock farming (van Herk 2002). These species were: *Candelariella reflexa and C. vitellina Physcia adscendens, P. dubia, P. caesia, P. tenella, Phaeophyscia orbicularis, Xanthoria candelaria, X. parietina, X. polycarpa.* Also common, but not associated with nitrogen were *Flavoparmelia caperata* and *F. soredians, Melanelia subaurifera, Physcia aipolia* and *Strangospora pinicola*. Species associated with acidic conditions were less common, but included *Lecanora conizaeoides, Hypogymnia physode*s and *Usnea subfloridana*. Other interesting species recorded were *Arthonia spadicea, Bacidia arceutina, B. laurocerasi B. naegelii, Dimerella pineti, Lecanora carpinea, L. saligna, and Punctelia subrudecta.* Bryophytes recorded were: *Amblystegium serpens, Brachythecium rutabulum, Dicranoweisia cirrata, Dicranum scoparium, Eurhynchium praelongum, Grimmia pulvinata, Hypnum cupressiforme, H. andoi, Orthotrichum affine, O. diaphanum, O. lyellii,* and *Ulota crispa.* Parasitic fungi recorded were: *Lachnella alboviolascens, Gloniopsis praelonga, Athelia arachnoidea,* and *Lichenococcus erodens.* Liverworts: *Lophocolea bidentata, Frullania sp*. and algae were: *Desmococcus viridis, Prasiola crispa and Trentepohlia* spp.

Ash bark acidity ranged from pH 3.6 to 6 with most values around 5. Average measurements at each site (8 measurements per tree between heights of 100m and 150m) showed lowest values at Bromley, an outer London site, and Westminster, a central London site.

Survey work for all London boroughs is now complete and will be analysed during the summer of 2003.

Conclusions from the pilot surveys

- Most single records are from suburban sites where pollution is lower
- Nitrophytes are ubiquitous
- Bark pH was only very weakly correlated with girth

- Although results indicate highest diversity for many decades some once common species were not recorded

Laundon (1970) recorded only 9 epiphytes in London and reported a gradient of increasing diversity with distance from Charing Cross correlating with ambient sulphur dioxide concentrations.

Hawksworth and McManus (1989) recorded 37 lichens from a range of tree species at 50 sites along a north-western transect from the city centre out to the London suburbs. No gradient of reducing diversity with distance was found and no correlation with pollution reported.

The spatial trend from this pilot study accords with Laundon. Diversity is significantly higher than both earlier studies despite being recorded from a single phorophyte. Community structure has changed.

Higher numbers of nitrophytic species, as defined by van Herk (2002), were recorded during this urban study than were recently found in areas of high ammonia (intensive livestock units) in the UK (Wolseley and James 2002). Ammonia concentrations measured at the only monitoring station in London (located in the highly trafficked Cromwell Road) are consistently higher than normal UK background concentrations. Many of the nitrophytes are also associated with an excess of other nutrients and are associated with fine particles arising from construction, industrial activity and road surfaces.

Air pollution and climatic data and results from the epiphytic surveys from each of the 33 London boroughs will be investigated using canonical correspondence analysis.

Lichen nomenclature follows lists maintained by Brian Coppins of the British Lichen Society (Coppins 2002).

Acknowledgements

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Changes in epiphytic lichen flora in urban environments

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Summary

The recolonisation of epiphytic lichens in the Ruhr Basin is described briefly and general aspects of re-establishing lichen floras are depicted in a model.

Introduction

The Ruhr valley has been one of the most heavily industrialised and urbanised areas in Europe since the beginning of the process of industrialisation on the European continent. It has suffered from some very serious environmental damage, until well after the Second World War, which led to serious damage to the lichen vegetation (Domrös 1966, Schönbeck 1972). In the 1980's, the drastic reduction in acceptable emission levels led to great improvements in air quality, particularly in relation to SO_2 levels.

This study deals with the transformation of the lichen vegetation in the Ruhr Valley area of Germany. Based on a comparison between the information found in this region and in the London area results are presented with a view to discovering if there are any universally valid characteristics of lichen recolonisation in urban areas.

The story of lichens in the Ruhr Area

Unfortunately, only very sketchy information about "historic" lichen flora in the Ruhr Valley is available. In 1800, Carstanjen published the "Flora Duisburgensis", in which he described the flora of Duisburg, and mentioned eight lichen species. However, in spite of this scant information, one can conclude from Carstanjen's findings that the lichen flora of the time must have been rich and varied, eg, Carstanjen describes that *Usnea plicata, Peltigera aphthosa* or *Lobaria pulmonaria* were very common („...ubique copiose...") in the local forests.

More than 160 years later, long after the Ruhr Area had developed into one of Europe's largest industrial areas, Domrös (1966) surveyed the lichen flora. The central parts of the Ruhr Area in particular were literally 'lichen deserts'. Towards the southern edge of the agglomeration Domrös found areas where lichens were struggling for survival as well as the transition zones where *Lecanora conizaeoides* was the dominant species on the trunks, apart from green algae. The observations by Domrös demonstrate that the high levels of air pollution caused by the Industrial Revolution have almost eradicated the original lichen vegetation recorded by Carstanjen. Only a very few highly tolerant species such as *Lecanora conizaeoides* could cope with the environmental stresses of the period.

Due to the enactment of Clean Air Acts, air pollution levels in the Ruhr area began to decrease around the end of the 1960s. There was a decline in sulphur dioxide levels in particular. This enabled the lichen flora to recover, and this process was first observed by Rabe & Wiegel (1985) and Weitkamp (1988). Recent surveys (Kricke 2002, Stapper and others 2000, Franzen 2001) charted the presence of approximately 60 species in the Ruhr Area.

Model illustrating lichen recolonisation

The process of recolonisation is depicted with the aid of the following diagram. It proposes a model for explaining the migration of lichens over time, which is based on different studies.

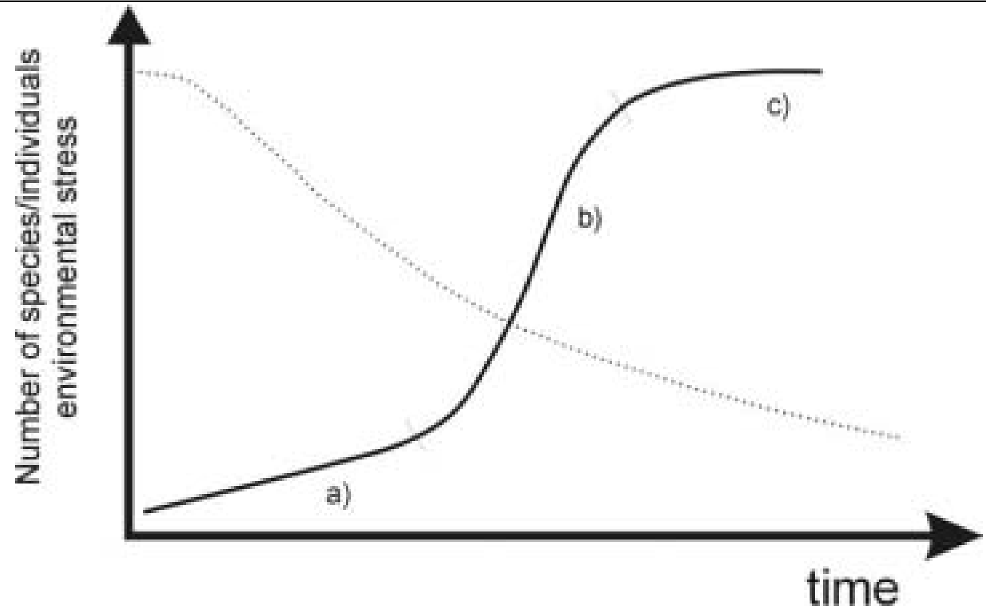

Figure 1: Suggested model for the recolonisation, of lichens ("Three Stage Recolonisation: As pollution and environmental stress (dotted line) decreases, the first lichens to return are those from the immediate environs and from areas to which they had previously retreated (First Stage, a). Later on in the second phase (Second Stage b) the diaspores of the individuals already established in the area play a role. In the Third Stage, c), the levels of individuals and the number of species reaches the maximum for that area.

The first stage of recolonisation is defined by a decline in air pollution. The events in this phase can be divided into two parts: those returning from safe zones/havens in the area and those arising from the re-migration of diaspores from the environs of the area. Some time after the beginning of the first re-migration phase, the second phase begins. This second stage happens with great rapidity, sometimes exponential speed, and is characterised by the development and growth of the newly established lichen. This means that in the second stage, diaspores from the area itself are responsible for the further development of the lichen population. In a process like this, the distances involved are anything from a few centimetres up to a few metres, which means the dispersal is very close to the source. In addition, there are diaspores that arrive in an area from outlying areas. This happens when there is a steady lowering of pollution and toxin levels in the environment and this is followed by a corresponding increase in suitable sites for growth. This may be facilitated, for example, by the toxins simply washed out of the bark or by new trees being planted. New and more sensitive species can begin to thrive in the area, thus increasing not only the number of individual plants, but also the range and number of species. In the third phase, the speed of resettlement slows down at a certain point and the numbers both of species and of individuals begin to level off and remain more or less constant (steady state). One reason for this levelling off is the reduction in suitable sites, which have not already become occupied. The most important factor however, is one which always arises in the process of resettlement of an ecosystem of this kind: at a certain point in time a given habitat becomes occupied by enough suitable species. The generalised species take over during the first phase and throughout the second stage, and they are followed during the third phase by only a small number of more specialised species.

The “Three Stage Recolonisation” model proposed here can be seen as an attempt to structure the process of recolonisation events over time. It may be used to predict the course of future lichen migration, in other words it can be of enormous value in the study of bioindicators in general.

It is important to remember, however, that some simplification of the conditions is necessary when formulating a model of this sort. This suggested model may undergo some modification in the future, once it has been put to the test by further studies on lichen recolonisation in urban areas.

Conclusion

As Seaward (1997) observed, the circumstances of newly migrated lichen vegetation differs considerably from that of the original lichens which had been severely damaged or completely wiped out by air pollution. The global transformation of the habitat, from damage by acidic contaminants to nutrient rich components, leads to an enrichment of the substratum of epiphytic lichens with nitrogen compounds, which encourage neutrophytic and nitrophytic species. This is evident also in the Ruhr area (Table 1). Currently, the most common species is *Physcia tenella*. *Physcia adscendens* or *Phaeophyscia orbicularis* are species which occur in more than half of the areas under study. On the other hand, the most common species up until 20 years ago, *Lecanora conizaeoides,* has disappeared from many sites and now appears in just over one third of the areas under study.

Table 1: Percentage frequency of selected species in the Ruhr Valley

Species	**Relative Frequency (%)**
Physcia tenella	90,1
Physcia adscendens	65,5
Phaeophyscia orbicularis	60,9
Amandinea punctata	53,7
Lecanora conizaeoides	36,9
Xanthoria polycarpa	34,4
Xanthoria parietina	24,6
Lecanora dispersa	24,3

The situation of lichen flora in city environments has undergone enormous and far-reaching change in the past 200 years, as has been demonstrated by research in the Ruhr Valley, and the comparative research with London. Going forward, it is a matter of concern, that the increasing influence of nitrogen emissions will encourage the dominance of nitrophilous and nitrogen-tolerant lichen varieties in urban areas. This may lead to an edging out of the remaining acidophytic lichen flora, and ultimately to their disappearance. This would mean a decline in the richness and number of species, something which could be halted by introducing appropriate regulation.

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Mapping aerial hypertrophication with epiphytic lichens as biomonitors in North Rhine-Westphalia (NRW, Germany)

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Summary

In North Rhine-Westphalia (Germany) the effects of hypertrophicating air pollutants were mapped using epiphytes as biomonitors. Spatial patterns of hypertrophication were obtained based on abundance and frequency of nitrophytic lichen species.

Introduction

Due to a strong decline of atmospheric sulphur dioxide concentrations, epiphytic lichen and bryophyte species diversity has, during the past decade, increased in formerly highly polluted regions of North Rhine-Westphalia (NRW). Emission levels of nitrogen oxides and ammonia have, however, remained largely unchanged. Nitrophytic species have recovered more frequently than others: About 60% of the epiphytic lichens that have recolonised the Ruhr District are dependent on mineral enrichment of the bark or tolerate hypertrophication to a certain extent (Stapper and others 2000, Franzen 2001). To provide information on the regional distribution of epiphytic bryophytes and lichens, in particular, lichens indicating hypertrophication or acidification. The epiphytic flora on 1815 trees uniformly distributed within NRW was investigated. This project was funded by the NRW Ministry of the Environment (grant to J-P Frahm, Univ. Bonn).

Methods

We have applied two standardized methods based on lichen frequencies within grids that allow the differentiation of regions with statistically different Air Quality Values ("AQV"; German VDI guideline; VDI 1995) or Lichen Diversity Values ("LDV", proposed EU protocol; Asta and others 2002). Ash (*Fraxinus excelsior*), Norway maple and sycamore (*Acer platanoides* and *A. pseudoplatanus*) meeting the criteria of both guidelines were selected as the major phorophytes. Sampling units (SU) are based on the 1:25,000 Ordnance map grid. Each of the 234 SU consists of 8 trees placed in the north western quadrant of either map. Relevés were carried out at the face with highest cover and species diversity (20cm x 50cm, 10 quadrates; VDI 1995) and at the four cardinal points (10cm x 50cm, 5 quadrates; Asta and others 2002). The LDV of a SU is the mean sum of the frequencies of all lichen species occurring within the grid. Furthermore, all bryophytes and lichens above 50cm up to 200cm, and the cover of filamentous green algae were recorded. The nitrophytic species index (NIW) was estimated according to van Herk (1999).

Results and discussion

Ninety-one lichen and 43 bryophyte species were recorded, 31 of which are red-listed in NRW, eg *Parmotrema stuppeum* (Taylor) Hale and *Orthotrichum pulchellum* Sw. (cat. 1; Heibel and others 1999, Schmidt & Heinrichs 1999). Species diversity was highest (49 per SU) in the southern highlands, where acidophytic species are dominant, and lower in the agricultural northern lowlands, where nitrophytic lichens, or bryophytes and lichens normally growing on basic rock are very common on bark. Highest mean numbers of nitrophytic lichen species per tree (<10) were recorded north of a line between Krefeld, Duisburg, Münster, and Osnabrück up to the Dutch border. In this area, excessive critical loads of eutrophicating nitrogen in forest ecosystems have been estimated (>25kg/ha/a; Gehrmann & Becker 2000), and livestock is very dense, suggesting that in NRW, like in The Netherlands (van Herk 1999), epiphytic flora is affected by livestock derived ammonia emissions.

Both frequency-based monitoring approaches indicate low emission loads in the southern and south-eastern highlands, and high loads in (formerly) highly industrialized regions like the Ruhr District. The agriculturally formed north-western lowlands, however, show high LDV indicating low emission loads due to high frequencies of nitrophytic lichen species often covering the whole surface of the trunks. Thus frequency-based biodiversity values do not indicate "real" emission loads unless the species specific behaviour of either lichen is respected. By analogy with the LDV we estimated the diversity value of only nitrophytic lichen species (van Herk 1999) that may be regarded as an indicator of hypertrophication. High values were calculated for the agricultural regions in the north of the country, whereas low values indicate no or only slight hypertrophication in the south. This approach yields essentially the same result as van Herk's (1999) method based on the occurrence and number of nitrophytes on the whole trunk instead of their frequency within the grid area.

To study the influence of "traffic" or "agriculture" on the frequency of epiphytes, we estimated whether the respective influence was low, moderate or high. Influence was "low" outside villages, far away from traffic or farming, and "high" along major roads, downtown or near (>100m) farms. Basically, epiphytic species diversity declines with increasing impact of either influence, some species react, however, contrarily and may therefore be regarded as indicators for one or both influences (Table 1). All lichen species reacting positively to increasing "traffic" or "agriculture", are nitrophytes according to van Herk (1999) or are considered as nitrophytes in the United Kingdom (Wolseley & James 2002). Most of the lichens responding negatively to either influence are acidophytes. *Physcia tenella* was found on 94% of all phorophytes and normally yielded high frequencies. Therefore, only negative changes at high traffic stations became visible. All species positively influenced by traffic are not limited to bark, but also may occur on rock and therefore may be regarded as indicators for (alkaline) dust emissions, eg *Physcia dubia* or the moss *Grimmia pulvinata*. Among the epiphytes positively influenced by traffic are the filamentous green algae of the genus *Klebsormidium* (eg *K. crenulatum* (Kützing) Lokhorst) which have become very common on trees in NRW (Frahm 1999). They were recorded at two thirds of all sampling units. Even fast growing foliose lichens like *Parmelia sulcata* or *Physcia tenella* may become overgrown rapidly. Sometimes 50% and more of a trunk's surface is covered by dense mats of algal filaments, in particular in highly populated regions at stations with strong traffic influence. It is unclear whether nitrogen containing emissions or mineral enrichment of the bark by dust are responsible for the positive reaction of some lichens, bryophytes and filamentous green algae to increased traffic influence. Traffic influence includes a chemically complex mixture of different gaseous or particulate components with different physiological effects on lichens and other sensitive organisms. The proposed EU protocol for lichen biomonitoring require a high degree of objectivity and standardization in the sampling procedure. Epiphytic lichen biodiversity on its own, however, is no longer a suitable indicator of emission loads. Future mapping projects should be used to identify indicator species, lichens and bryophytes, and their tolerance range.

Table 1. Epiphytes responding to increasing influence of "traffic" or "agriculture" with decreasing or increasing frequencies ($p<0.05$ between categories "low" and "high"). Only sampling units below 160m above sea level are included. N, A: species identified by van Herk (1999) as nitrophytes and acidophytes, respectively. *, Species considered as nitrophytes in the UK (Wolseley & James 2002).

N*	*Amandinea punctata*	Frequency decreases with increasing impact of traffic
	Dicranoweisia cirrata	
	Hypnum cupressiforme	
N*	*Lecanora expallens*	
A	*Lepraria incana*	
A	*Parmelia saxatilis*	
N	*Physcia tenella*	
	Ramalina farinacea	
	Filamentous green algae	Frequency increases with increasing impact of traffic
	Grimmia pulvinata	
	Orthotrichum diaphanum	
N	*Phaeophyscia nigricans*	
N	*Phaeophyscia orbicularis*	
N	*Physcia dubia*	
N	*Xanthoria parietina*	
A	*Evernia prunastri*	Frequency decreases with increasing impact of agriculture
A	*Hypogymnia physodes*	
A	*Lecanora conizaeoides*	
A	*Lepraria incana*	
	Orthotrichum diaphanum	
	Melanelia subaurifera	
	Parmelia sulcata	
N	*Phaeophyscia orbicularis*	
N	*Physcia caesia*	
N*	*Amandinea punctata*	Frequency increases with increasing impact of agriculture
N	*Caloplaca holocarpa*	
N*	*Lecidella elaeochroma*	
N*	*Physconia grisea*	
N	*Xanthoria parietina*	

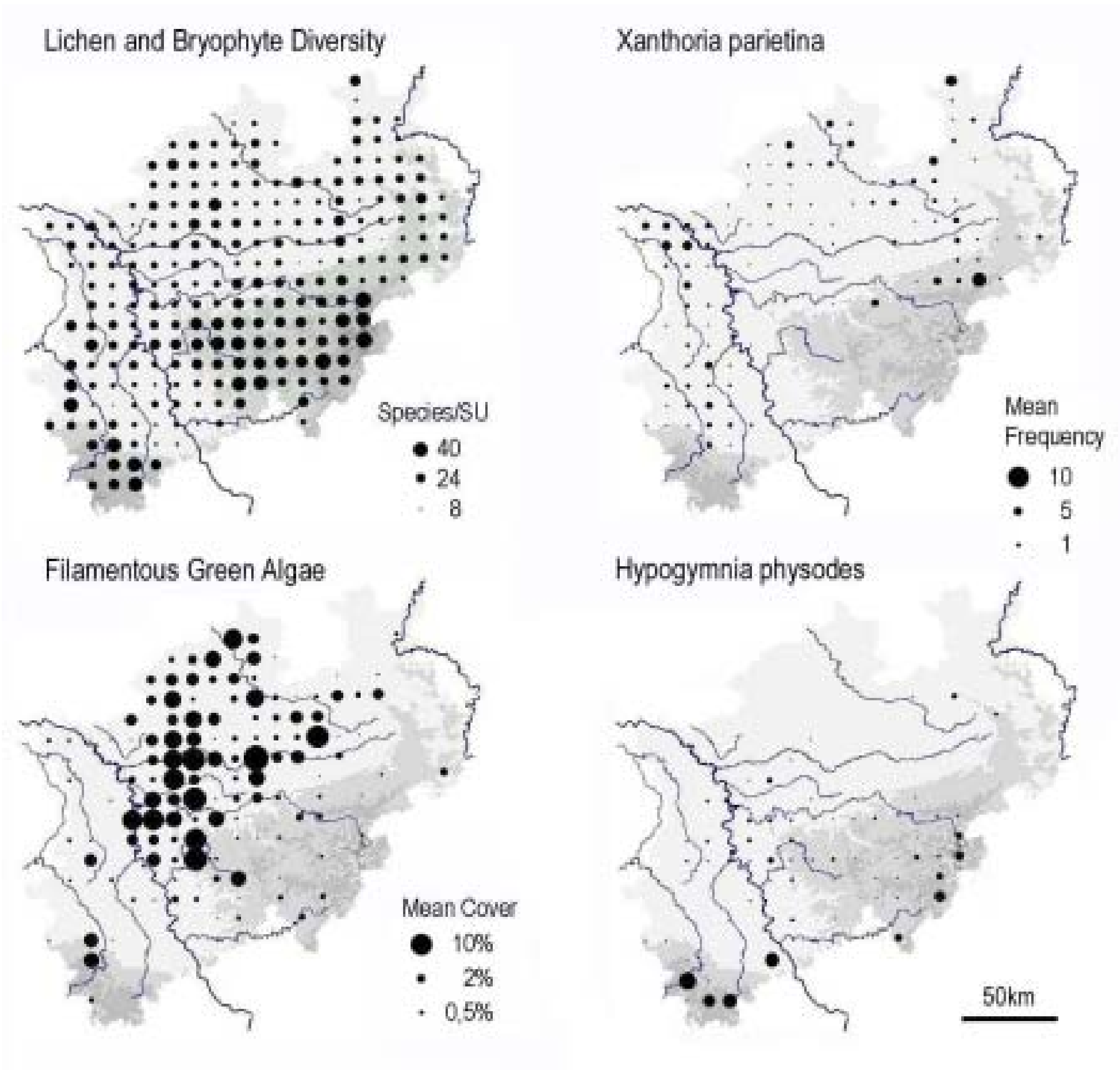

Figure 1: Epiphyte mapping in NRW: Lichen and bryophyte diversity per sampling unit, distribution and mean cover of filamentous green algae on bark, and distribution of nitrophytic lichen *Xanthoria parietina* and acidiophytic lichen *Hypogymnia physodes*. Background indicates height above sea level (15m to 840m).

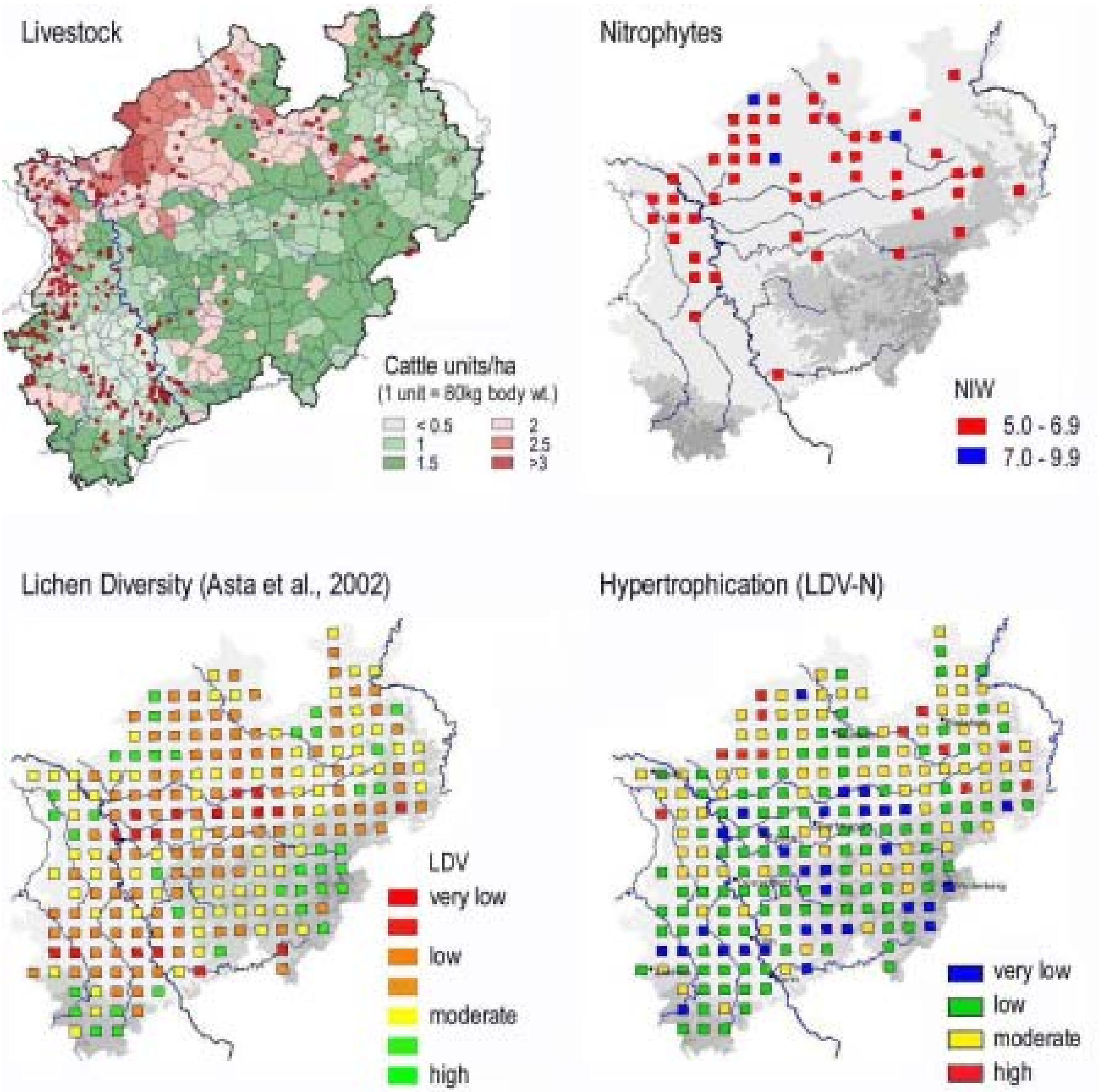

Figure 2. Mapping environmental stress: Livestock density (source: Grundwasserbericht Nordrhein-Westfalen 2000. Ministerium für Umwelt und Naturschutz, Landwirtschaft und Verbraucherschutz des Landes Nordrhein-Westfalen. Düsseldorf, 269p.). Mean number of nitrophytic lichen species per tree (NIW) according to van Herk (1999), where only sampling units with NIW >5 are shown. Map of Lichen diversity (Asta and others 2002) as indicator of environmental stress (5.6 < LDV < 65.5, class width = 9.4). Map of hypertrophication as determined by the frequencies of nitrophytic species only (LDV-N; class width = 12.5).

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Mapping the effects of air pollution, nitrogen deposition, agriculture and dust by the diversity of epipytic lichens in central Italy

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Summary

An example from central Italy is given to show how nitrophytic species affect the diversity of epiphytic lichens. In order to give realistic results reflecting air quality nitrophytic species were excluded from the calculation of the index of lichen diversity. Conversely, using only nitrophytic species total nitrogen deposition in the area was mapped. Mapping using only strictly nitrophytic species showed two peaks in the vicinity of two big sheep and pig stock farms. It was concluded that monitoring of air quality must consider the 'noise' caused by nitrophytic species and that monitoring of ammonia pollution and nitrogen deposition is feasible using regionally selected indicator species.

Introduction

In field monitoring studies it is very difficult to separate the effects of many intercorrelated variables, and the interpretation of lichen biomonitoring surveys in terms of environmental change is open to some controversy, mainly because of the complexity of the system (Van Dobben & Ter Braak 1998). It is well-known that lichens are sensitive to air pollution, however, other ecological variables such as climate, substrate, light etc., also induce variation in the frequency of lichen species. As a consequence, it is often difficult to discriminate between the effects of pollution and those of other environmental changes.

As bioindication is largely a matter of data interpretation (Loppi and others 2002), this does not mean that the effects of individual pollutants cannot be detected, though great care must be taken in the interpretation of the results. In this context, when monitoring the biodiversity of epiphytic lichens (see Asta and others 2002), the interpretation of the results could consist of several maps, each based on different groups of species, and each interpreted in a different way.

In lichen bioindication studies the inclusion of nitrophytic species has been questioned since relatively high values could be due to eutrophication and not to air pollution (van Herk 2001). Furthermore, the use of nitrophytic species could represent an effective tool for mapping the effects of ammonia pollution, as has been accomplished for The Netherlands (van Herk 1999). However, it has been shown that in areas with a warm and dry climate, such as the Mediterranean basin, mapping of nitrogen effects may be hampered by the dominating effect of dust (Loppi & De Dominicis 1996).

The present paper reports a case study from central Italy to show all these situations, and discusses possible issues for follow-up studies and for using lichens in large-scale and/or long-term monitoring programmes.

Data and methods

A lichen biomonitoring survey of air quality was carried out in the Municipality of Colle Val d'Elsa (Tuscany, central Italy). The study area was systematically divided following the network of 3x3 km used for monitoring the effects of air pollution on forest ecosystems (European net EU-UN/ECE). A few stations were chosen in the surroundings of the main urban area, based on a finer network of 1x1 km. 3-10 *Tilia* spp. or deciduous *Quercus* spp. trees were sampled in each station. For each tree, an index of lichen diversity (ILD) was calculated as the sum of frequencies of epiphytic lichens in a sampling grid of 30x50 cm divided into 10 units of 10x15 cm. Two-dimensional zone-maps were drawn using the plotting program SURFER (Golden Software Inc., Colorado).

Results and discussion

The biodiversity map of epiphytic lichens, supposed to reflect the air quality of the area (Asta and others 2002), is shown in Figure 1A. The map demonstrated a general picture of low environmental alteration. However, sources of atmospheric pollution were present in the area, as vehicular traffic, domestic heating and, especially with an important crystal factory emitting large amounts of fluoride. Fluoride is well-known for being highly toxic to lichens (Gilbert 1973). Around the crystal factory, however, lichen species such as *Xanthoria parietina*, *Physcia adscendens*, *Lecidella elaeochroma*, *Lecanora chlarotera*, etc., were well established. Furthermore, the distribution maps of several lichen species (eg *Hyperphyscia adglutinata*, *Phaeophyscia orbicularis*, *Physcia adscendens*, *Xanthoria parietina*, *Physconia grisea*, *Lecanora hagenii*) showed a pattern with peaks in or around urban areas. All these resistant (tolerant) species are regarded as nitrophytic, and it seems reasonable to exclude them from the calculation of the ILD, if such an index is addressed to the estimation of "classical" air pollution by phytotoxic gases. To achieve an objective way of selecting nitrophytic species, we used the "floristic query interface" of the on-line checklist of Italian lichens (Nimis 2000), asking for the epiphytic species of Tuscany with an indicator value for eutrophication of 4 (rather high eutrophication) or 5 (very high eutrophication). These 36 lichens species (Table 1) were excluded from the calculation of the ILD. The resulting map, plotted according to a newly calibrated scale, is shown in Figure 1B.

Table 1. Nitrophytic species of Tuscany (epiphytic) according to Nimis (2000)

Amandinea punctata	*Dirina ceratoniae*	*Physcia aipolia*
Caloplaca cerina	*Hyperphyscia adglutinata*	*Physcia biziana*
Caloplaca cerinelloides	*Lecanora chlarotera*	*Physcia dubia*
Caloplaca herbidella	*Lecanora hagenii*	*Physcia stellaris*
Caloplaca luteoalba	*Lecanora umbrina*	*Physcia tenella*
Caloplaca obscurella	*Lecidella elaeochroma*	*Physcia vitii*
Caloplaca pyracea	*Lecidella flavosorediata*	*Physconia distorta*
Caloplaca ulcerosa	*Phaeophyscia cernohorskyi*	*Physconia enteroxantha*
Candelaria concolor	*Phaeophyscia chloantha*	*Physconia grisea*
Candelariella reflexa	*Phaeophyscia hirsuta*	*Ramalina canariensis*
Diploicia canescens	*Phaeophyscia orbicularis*	*Xanthoria fallax*
Diplotomma alboatrum	*Physcia adscendens*	*Xanthoria parietina*

Although direct measurements of air pollutants in the area are missing, this new map seemed to better reflect the air pollution status of the area, evidencing a high environmental alteration in urban areas, around the crystal factory and along a highway.

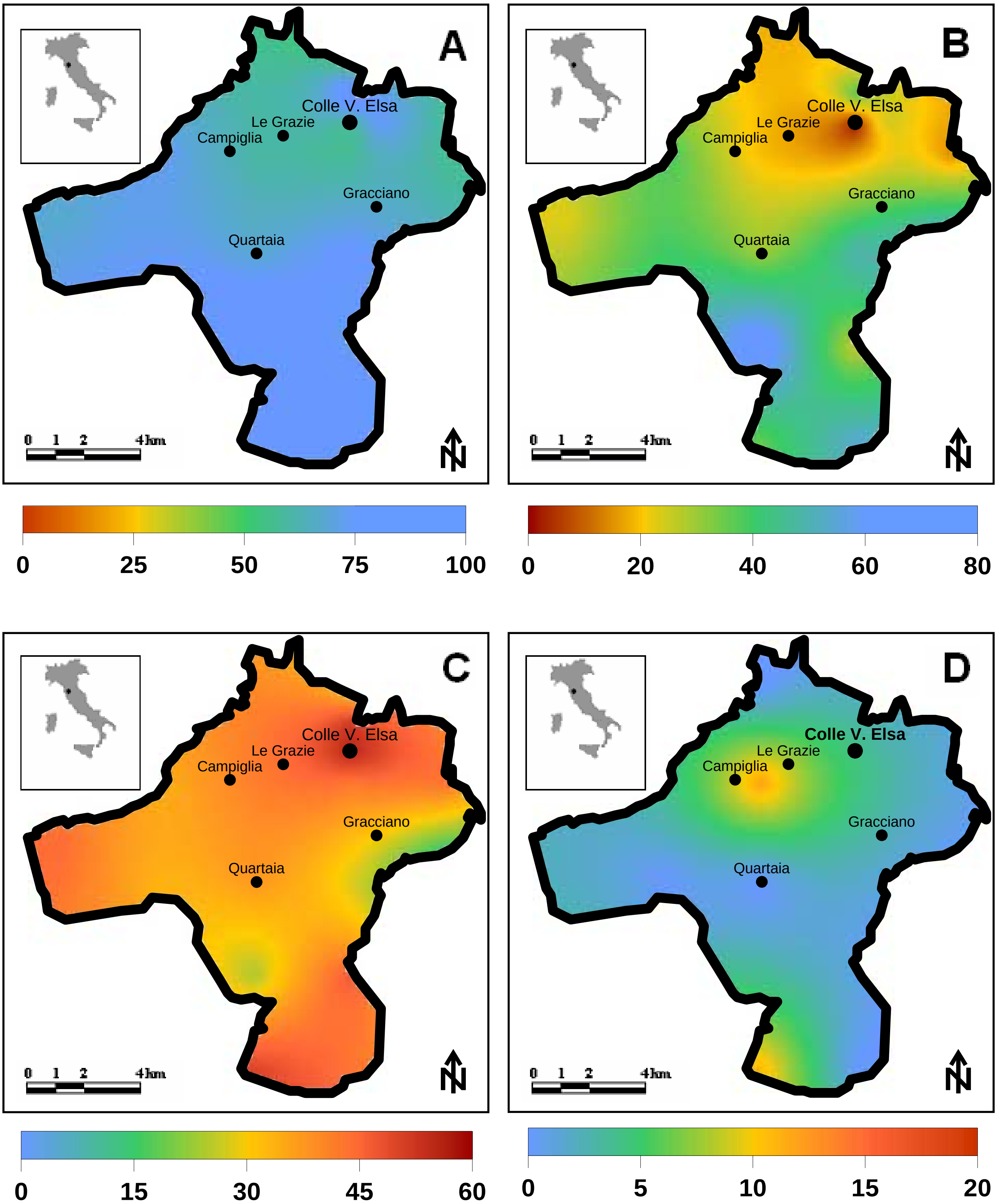

Figure 1. Maps of the biodiversity of epiphytic lichens (A), biodiversity excluding nitrophytic species (B), biodiversity of nitrophytic species (C), biodiversity of strictly nitrophytic species (D). Note that high values in maps A and B correspond to low pollution, while high values in maps C and D correspond to high pollution.

By calculating ILD values only with the nitrophytic species of Table 1, another map was plotted (Figure 1C), which was expected to reflect the total nitrogen deposition in the area. This map showed that most of the municipality is subject to high N pollution.

However, it is well known that in the Mediterranean area dust can play an important role since alkaline dust contamination raises bark pH and causes bark hypertrophication (Gilbert 1976). Moreover, dust impregnation, independently of dust chemistry, also makes the bark drier, (Loppi & Pirintsos 2000), and since nitrophytic species are also xerophytic, such interactions should be taken into account. Furthermore, also synergistic effects of light must be considered since most nitrophytic species are also photophytic and a higher light influx leads to drier conditions.

To overcome such problems and obtain a reliable map of NH_3 emissions, only strictly nitrophytic species, ie those with an indicator value for eutrophication from 4 to 5, were selected form the list of Table 1. This led to the calculation of ILD values based on four species only: *Candelariella reflexa*, *Physcia dubia*, *Physcia vitii* and *Physconia grisea*.This last map (Figure 1D) showed two peaks, supposed to reflect higher ammonia concentrations in the air. No direct atmospheric NH_3 measurement was available, but in the vicinity of these two peaks, two big sheep and pig stock farms are present, and the latter was so big that it was coupled with a methane production plant.

Conclusions

From this worked example from central Italy, some general conclusions can be drawn:

∉ monitoring of air quality must consider the 'noise' caused by nitrophytic species;

∉ monitoring of nitrogen deposition is feasible using nitrophytic species in a broad sense;

∉ monitoring of ammonia pollution is feasible using strictly nitrophytic species;

∉ indicator nitrophytic species should be selected at a regional scale.

Furthermore, there are some issues to be considered in large-scale and/or long-term monitoring programmes:

∉ mapping of ammonia pollution requires a very high sampling density;

∉ caution is required in interpreting the results of recolonization studies.

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Session 1 Discussion

Group 1. A changing lichen flora

Participants: Stefano Loppi (chair), Val Cooper, Laetitia Davranche, Peter James, Randolphe Kricke, William Purvis, Norbert Stapper, Ray Woods

Questions

- How do we separate out effects of different pollutants?
- Need to consider other non-pollutant effects (eg grazing, shading).
- Role of other factors in causing variation in species composition and diversity eg bark texture.
- Impact of climate change and interaction with N pollution?
- How can we use Ellenberg values for lichens?
- What changes in lichen flora are acceptable?
- Colonisation and establishment – inability to cultivate in laboratory conditions.

How do we separate out effects of different pollutants?

Pollutants include NOx, NHx, SOx, heavy metals, Ozone.

Measurement of factors such as bark pH will contribute to our knowledge of ongoing changes, providing additional information where destructive sampling is used to detect changes in accumulated substances.

Use of acid barked trees to detect shifts from acidophytes to nitrophytes.

Explore the possibility of twigs as an early warning system.

Use of multivariate analysis and robust statistics to elucidate factors affecting lichen distribution, for example separating climate change from N pollution.

Use of a robust sampling strategy such as BioAssess (Scheidegger and others, this volume).

How can we use Ellenberg values for lichens?

The Italian lichen database already includes much of the Ellenberg approach in identifying factors associated with lichen distribution across climatic conditions of Italy.

Final selection of regional indicator species is important for detecting shifts in species distribution due to changes in environmental conditions.

Create synthetic indices and use results to reassess indicator species.

What changes in lichen flora are acceptable?

Changes in distribution are inevitable but interpreting the direction of changes and their causes contributes to avoiding undesirable changes.

Colonisation and establishment – inability to cultivate in laboratory conditions. The importance of studies under controlled lab conditions (for example artificial substrates) cannot be underestimated as these contribute towards our understanding of lichen populations in the field.

Lichens as sentinels – detection of changes using a range of techniques now available including physiology, biomarkers, genetic variation etc. This type of research needs collaboration and large scale funding such as a joint European Project in the VIth Framework.

3. The pollution environment

Air quality and ecosystem impacts: the policy perspective

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Summary

Defra's ongoing programme of research to understand the impacts of air pollutants on natural ecosystems is outlined including progress towards defining critical levels and critical loads of known pollutants. Protocols contributing to the decline of emissions of pollutants are outlined and ecosystems at risk from eutrophication due to nitrogen oxides, ammonia and ground level ozone are highlighted as requiring more stringent emission targets.

Introduction

Air pollution can lead to adverse effects on human health, materials and ecosystems. There have been considerable improvements in air quality over the last 30 years. However, Defra recognises that there is still more to be done and the impacts of air pollution on ecosystems are a continuing driver of national and international agreements to reduce emissions.

The effects of sulphur dioxide and acid rain on the environment are well known. The most striking examples are the acidification of lakes and streams in upland areas of the UK and the widespread changes in lichen diversity. Other priority air pollutants are nitrogen oxides, ammonia and ground-level ozone. When present in high concentrations, above the 'critical level', these pollutants may lead to direct toxic effects on sensitive vegetation. In addition, when deposited from the atmosphere onto the land surface, air pollutants can contribute to acidification of soils and waters, and terrestrial eutrophication of sensitive plant communities. These problems can bring significant consequences for ecosystems, particularly in upland areas, and sites of conservation importance.

Ongoing programmes of research

Defra invests in a considerable ongoing programme of research to understand the impacts of air pollutants on the environment, together with a large monitoring programme to measure concentrations and deposition of air pollutants. The National Air Quality Archive (website) presents research reports and monitoring data.

Reducing the impacts of air pollution on sensitive ecosystems is one of the principal drivers of air quality policy on both the national and international level. Two important concepts have been developed to guide policy:

- "critical levels" are concentrations of pollution in air which, when exceeded, may lead to adverse effects on plants or ecosystems.
- "critical loads" are the amount of deposited pollutant which, when exceeded, may lead to adverse effects on ecosystems. Critical loads for acidity and nutrient nitrogen are set to protect against acidification and nitrogen over-enrichment.

The UK's National Focal Centre for Critical Loads (website) recently updated the national maps of habitat types and their critical loads, to reflect developments in the scientific research within the UK and Europe. By overlaying UK maps of concentration or deposition onto maps of critical levels and loads, it is possible to identify areas where the level or load is exceeded and thus the ecosystem is at risk.

The Air Quality Strategy for England, Scotland, Wales and Northern Ireland (AQS 2000) was published by the Government and the devolved administrations in January 2000, establishing the framework for improving ambient air quality. The AQS objectives focus on achieving critical levels, as high concentrations of pollutants are dominated by local or regional sources within the UK, and can therefore largely be achieved through implementation of measures within the UK and on a local/national level.

The AQS focuses mainly on reducing the health effects of air pollution, but also includes two objectives for protection of vegetation and ecosystems based on nitrogen oxides and sulphur dioxide critical levels in air (NO_x concentration not to exceed 30 $\mu g/m^3$ as annual mean; SO_2 concentration not to exceed critical level of 20 $\sigma g/m^3$ as annual and winter mean). These objectives, that do not apply in exclusion zones (zones close to agglomerations, roads and large industrial plant), have been achieved, and Defra is now looking at ways to tighten the objectives in future, particularly to protect designated conservation areas.

At the same time as seeking to reduce ambient concentrations of pollutants to below critical levels, a principal aim of air quality policy is to reduce the deposition pollutants to below harmful levels that cause acidification and eutrophication - ie below the critical loads. The long-range transport of air pollutants means that a proportion of UK emissions is exported from the country, and deposition within the UK is heavily influenced by non-UK emissions. In recognition of the transboundary nature of air pollution, two international frameworks have been established to agree international emission targets: the UNECE Convention on Long Range Transboundary Air Pollution **(**website**)** and the EU's Acidification Strategy and the more recent Clean Air For Europe Programme (website).

As a result of commitments under UNECE Protocols and European Directives, emissions of nitrogen oxides have declined by 40**%** and of sulphur dioxide by 70**%** since 1990. The Government has recently committed to further emission reductions, to be achieved by 2010, of NO_x, SO_2, VOCs (precursors to ground-level ozone) and ammonia under the UNECE Protocol to Abate Acidification, Eutrophication and Ground-level Ozone (the Gothenburg Protocol) and the European National Emissions Ceilings Directive. These agreements are the first to contain national emission targets for ammonia, an air pollutant mainly emitted from livestock manure.

The Government commissioned the National Expert Group on Transboundary Air Pollution (NEGTAP) to investigate the impacts of air pollution on UK ecosystems and the prospects for the future. NEGTAP's report (2001), entitled 'Transboundary Air Pollution: Acidification, Eutrophication and Ground-level Ozone in the UK' concluded that some improvements are being observed. For example, the UK Acid Waters Monitoring Network (website) is showing signs of improved chemical quality of upland acidified lakes and streams, thanks to substantial cuts in sulphur dioxide (from large reductions in emissions from power generation and industry). Also, peak concentrations of ozone have declined in response to reductions of nitrogen oxides and VOCs in the UK and Europe. However, as a result of growing emissions of these precursors outside Europe, background levels of ozone are continuing to rise. Defra and a range of research organisations are examining the potential implications of this growing background, on human health and vegetation. NEGTAP also highlighted the continuing threat from ammonia. As emissions of nitrogen oxides have been cut, ammonia has become the dominant air pollutant contributing to eutrophication in Europe. Defra has published 'Ammonia in the UK' (website), a summary of its research on ammonia, its emissions, impacts and potential options for emission abatement.

Ecosystems at risk

Recent estimates by Defra's National Focal Centre for Critical Loads estimate that approximately 60% of the area of UK ecosystems are at risk from eutrophication, and a similar area for acidification. There is now emerging evidence for changes in the plant species across semi-natural areas of the UK, and deposition of nitrogen oxides and ammonia is likely to be part of the problem. For example, the New Atlas of British and Irish Flora (Preston and others 2002) found a decline in species typical of habitats with low nutrient levels, and a corresponding increase in species requiring high nutrient levels.

It is recognised that further emission reductions will be needed to curb the impacts of air pollution on ecosystems, particularly eutrophication and ground-level ozone. The UK is preparing for international negotiations to review the UNECE Gothenburg Protocol and the EU National Emission Ceilings Directive in 2004/5. It is expected that reducing the impacts on semi-natural ecosystems will be major drivers towards setting more stringent emission targets.

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Air Pollution Assessment by the Statutory Nature Conservation Agencies

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Summary

Atmospheric nitrogen pollution is a threat to the conservation of wild habitats and species in the UK. Deposition of nitrogen exceeds the critical loads over large areas of sensitive habitats and recent national surveys have provided evidence of an increase in nitrophilic species in some sensitive terrestrial habitats. New legislation is challenging the conservation agencies, and the pollution regulators, to re-visit their assessment of air pollution impacts on ecosystems. Bio-monitoring tools, including lichens as indicators of air quality, are likely to have an important role in the evaluation of air pollution impacts on nature conservation, both nationally and on individual sites.

Introduction

The National Expert Group on Transboundary Air Pollution (NEGTAP 2001) reported that atmospheric nitrogen deposition poses a major threat to semi-natural ecosystems in the UK. Large areas of sensitive ecosystems exceed the nutrient nitrogen critical load. This includes a significant proportion of nationally and internationally important nature conservation sites.

Two recent major surveys, the Countryside Survey 2000 (Haines-Young and others 2000) and the New Plant Atlas for British and Irish Flora (Preston and others 2002), have demonstrated a significant shift towards more nitrophilic plant species in some communities.

Recent legislation, such as the Integrated Pollution Prevention and Control Directive and the Habitats Directive, places new obligations on the pollution regulators and the nature conservation agencies as statutory consultees. They require an assessment of air pollution impacts on individual sites from new and existing regulated sources, and this is challenging the way we judge the impacts of nitrogen on ecosystems.

N pollutant issues

NEGTAP provides a detailed review of the emissions, distribution and impacts of the different atmospheric nitrogenous pollutants (NEGTAP 2001). The greatest threat to nature conservation, in terms of the SSSI/European site series and the wider countryside, is from total nitrogen deposition. However, in addition, statutory nature conservation sites may be at risk from elevated concentrations of oxides of nitrogen if they are situated in close vicinity to major roads, industry or urban areas. Ammonia is also an air pollutant that is increasingly at the forefront of air pollution issues for nature conservation. Phytotoxic concentrations of ammonia are experienced close to large agricultural livestock units, or areas where manures or slurries are spread (Chadwick this volume). More significantly in terms of the overall risk to nature conservation is the deposition of ammonia, or ammonium. This is both a local issue and a long range issue since reduced nitrogen is now the dominant form of total nitrogen deposition in the UK.

Projected reduction of oxidised nitrogen compounds (NOx) and reduced nitrogen (NHx) emissions show that in 2010 (the date by which the UK commitments under the Gothenburg Protocol and the National Emissions Ceilings Directive are required to be met) large areas of protected UK habitats will still be receiving unacceptable inputs of nitrogen compounds.

Air pollution impacts on statutory nature conservation sites

The UK has over 800 European sites, ie Special Areas of Conservation, designated under the Habitats Directive and Special Protection Areas designated under the Birds Directive, and over 6000 Sites of Special Scientific Interest. The Habitats Directive requires the UK Government and other Member States to prevent harm to, and avoid deterioration, of European sites. In addition, the Government has set a target to achieve favourable condition on 95% of Sites of Special Scientific Interest (SSSI) in England, by area, by 2010.

Furthermore, the Conservation (Natural Habitats &C) Regulations 1994 (the 'Habitats Regulations'), which transpose the Habitats Directive into UK law, introduces strict site safeguard measures concerning 'plans or projects' which are likely to have a significant effect on a European site. Subject to reasons of over riding public interest, a plan or project should only be authorised if it can be concluded that it will not have an adverse effect on the integrity of a European site.

Air pollution impacts are one of the many threats to the condition and integrity of designated nature conservation sites. The conservation agencies, as advisers to the Government, its devolved administrations, and the pollution regulators, have a vital role to play in advising on pollution reductions necessary to protect the natural environment and in particular, designated nature conservation sites. This may take the form of advising on the impacts of an installation in the vicinity of a statutory site or, more strategically, advising on the impacts of air pollution at the national scale. In this context, the conservation agencies use the critical loads and critical levels approach to assess broad scale risk to statutory sites.

The last few decades have seen significant reductions in the emissions of sulphur dioxide and oxides of nitrogen, in particular from industrial sources. However, nitrogenous emissions are now dominated by diffuse, non-regulated, sources of oxidised nitrogen (ie traffic) and reduced nitrogen (ie intensive agriculture). As a result, it is becoming increasingly costly or more difficult to achieve further reductions of nitrogen emissions to ensure that critical loads and levels are not exceeded. In order to advocate the continued reduction in emissions, a better understanding of the risk and impacts on statutory sites.

The conservation agencies therefore will need to better understand the condition of designated sites in the context of air pollution impacts. This information will be required to allow the conservation agencies to continue advocating the need for further and more targeted cuts in emissions of air pollutants. It is also required in order to better inform conservation agency staff with regards to the threats of air pollution and how this might interrelate with conservation objectives, monitoring and management of sites, and to inform our advice on air pollution related casework.

However, there is currently no systematic monitoring or assessment of air pollution impacts on species or habitats at individual sites. Whilst there are monitoring networks of air and deposition chemistry throughout the UK and extensive and wide ranging biological monitoring on statutory sites, there is very little assessment of the former evaluation in the context of the latter.

These shortcomings were recognised by NEGTAP and led to one of its key recommendations "site-specific assessments are urgently needed to ensure that the protection of sites of high biodiversity under the Habitats Directive takes into account the impacts of local and regional air pollution".

NEGTAP also identified the need for research to "develop improved methods of ecological risk assessment for air pollution to be applied at international or local scales".

Current initiatives

In order to understand better the impact of nitrogenous compounds on protected sites we need to develop robust monitoring methods. These need to be used to monitor the impacts of nitrogen pollutants and to help relate observed changes, at the sites, to the causal factor.

The conservation agencies are currently funding work to describe the range of, and the potential use of, bio-monitoring methods for assessing the effects of nitrogen pollution on sites. Lichens may have a role to play in the future assessment of the health of internationally important wildlife sites from air pollution effects, in addition to being species of innate conservation importance.

Coupled with the development of guidance for risk assessment of air pollution impacts and an Air Pollution Information System, this bio-monitoring should complete a package of tools for the conservation agencies to use in evaluating impacts on sites. It is then essential that the conservation agencies better articulate the risks from atmospheric nitrogen pollutant to the nature conservation resource.

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Exposure of ecosystems to atmospheric ammonia in the UK and the development of practical bioindicator methods.

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Summary

This short paper summarises our knowledge on the distribution of ammonia in the UK, highlighting the uncertainties that remain in quantifying deposition at individual sites, due to the existence of substantial local variability. It then considers the application of bioindicator methods to provide a complementary approach to estimating N deposition, NH_3 concentrations and/or effects of N deposition. While the focus in the paper is on NH_3, the principles are relevant for other components of N deposition, although it is noted that the responses of bioindicators to nitrogen oxides (NO_x) and nitrates (NO_3^-) may differ and are less well established.

Introduction

Atmospheric ammonia is recognized as being a major component of nitrogen and acidifying deposition in the UK (NEGTAP 2001, DEFRA 2002). For many years its distribution was highly uncertain, due to lack of measurements and uncertainty in the modelling of its behaviour. However, as a result of substantial activity over the last decade there is now a much-improved understanding of the emissions, concentrations, fluxes and impacts of ammonia in the UK (for example Sutton and others 1993, 1998, 2001; Fowler and others 1998, Dragosits and others 2002, Pitcairn and others 2002). The picture that emerges shows ammonia to be a pollutant with two broadly contrasting scales of impact: firstly, the exposure to NH_3 concentrations and dry deposition of NH_3 nitrogen may have substantial effects on ecosystems, with a high spatial variability related to the magnitude of the farm emissions occuring in the rural environment (Dragosits and others 2002). Secondly, the NH_3 that is not locally deposited is dispersed more widely in the atmosphere, with a large fraction being converted, by reaction with atmospheric acid sulphates and nitrate, into ammonium containing aerosol. This aerosol may be transported in the atmosphere many 1000s of km, with most of it being removed as wet deposition, largely in hill areas. As a consequence of these two effects, the nitrogen deposition from ammonia appears to be having significant effects on sensitive ecosystems in both source regions and remote hill areas.

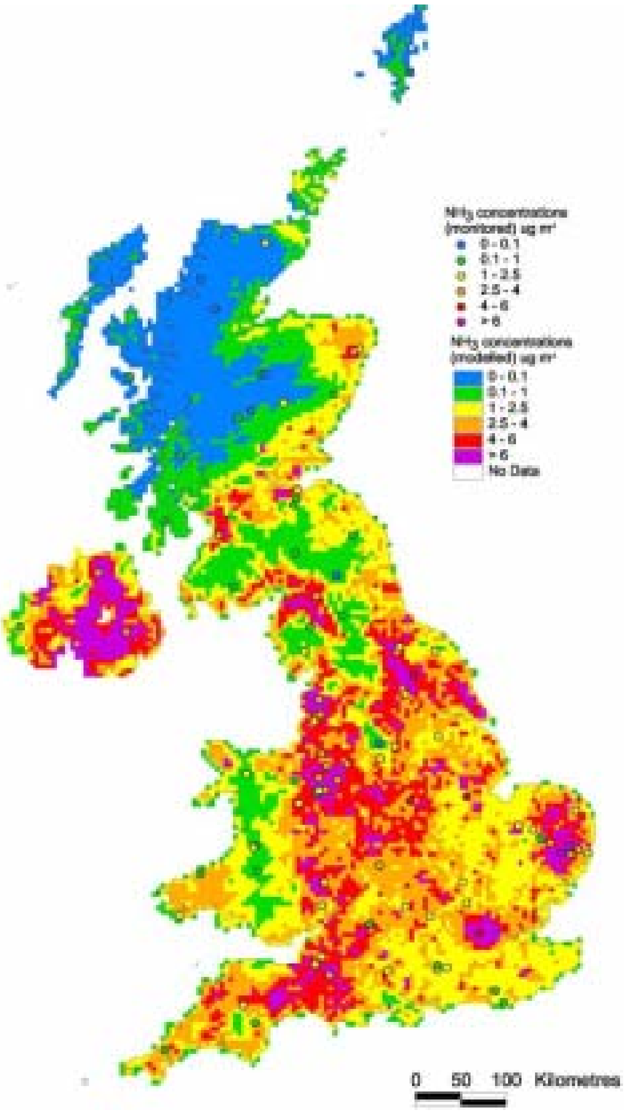

Figure 1: Distribution of NH_3 concentrations across the UK (Sutton and others 2001). The coloured circles show measured concentrations from the National Ammonia Monitoring Network, while the gridded background represents estimates from the FRAME atmospheric dispersion model.

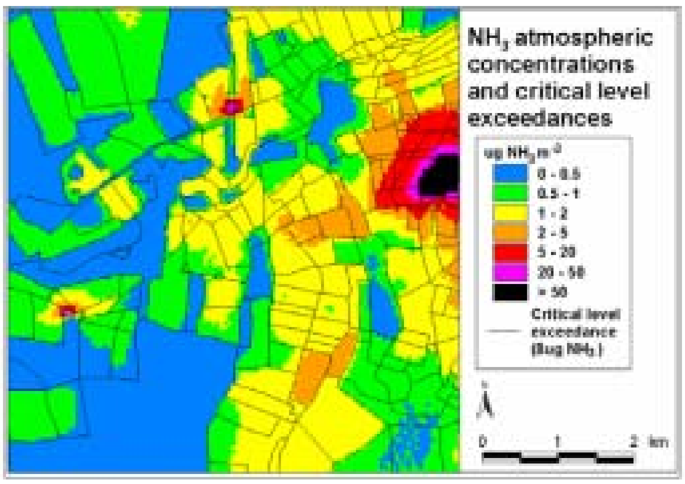

Figure 2: Modelled NH_3 concentrations across an example 5 km gridsquare 'Ambridge', an unspecified location in central England (Dragosits and others 2002). The currently accepted critical level of 8 σg m^{-3} (annual mean) for NH_3 is shown as a black ring around each of three livestock farms.

Distribution of ammonia across the UK

The spatial distribution of ammonia in the UK was assessed initially using models, with subsequent measurements allowing evaluation of the model estimates. The modelling approach currently used incorporates an inventory of all the NH_3 emissions from different sources at 5 km resolution, including livestock, fertilizers and crops, vehicles fitted with catalytic converters, sewage, pets, wild animals and a host of other minor sources (Dragosits and others 1998, Misselbrook and others 2000, Sutton and others 2000). These emissions are dispersed in the atmosphere and allowed to react with SO_2 and NO_x emissions in the FRAME (Fine Resolution Atmospheric Multi-pollutant Exchange) model (for example Fournier and others 2002, Sutton and others 2001). FRAME includes a detailed vertical resolution in the atmosphere, allowing ground level NH_3 concentrations to be estimated, as well as aerosol ammonium (NH_4^+) concentrations, plus wet and dry deposition. Obviously, such a modelling approach generates theoretical estimates based on many assumptions. The UK National Ammonia Monitoring Network (NAMN) was established in 1996 to provide robust measurements on the spatial distribution and long-term trends in ammonia and this provides a key resource to evaluate the model estimates (Sutton and others 2001, CEH 2002).

An example of the output of FRAME and the NAMN is shown in Figure 1. This compares the mean gaseous NH_3 concentrations across the UK from the measurements and modelling. It demonstrates the high spatial variability across the country, with NH_3 concentrations in the range ~0.05 to 15 σg m^{-3}. By comparison, since aerosol NH_4^+ is a secondary product formed slowly in the atmosphere, it has a much lower spatial variability, with sites within 50 km having similar concentrations (Sutton and others 2001). It is important to recognize that Figure 1 still hides much of the spatial variability, which can be substantial even over distances as small as 10 m. For example in an example 5 km gridsquare with a mean concentration of around 2 σg m^{-3}, concentrations may vary between 0.5 to 50 σg m^{-3}. This is illustrated in Figure 2, which provides a modelled case study of variability for 'Ambridge', an unspecified location in central England.

The example of 'Ambridge' shows how substantial increases in NH_3 concentrations occur near three livestock farms, with the largest concentrations around a poultry farm housing >200,000 birds. It also shows the substantial changes in NH_3 concentrations across the landscape due to field activities (fertilization, grazing, manure spreading). The lowest concentrations occur in the centre of large woodland areas, such as in the SW corner of the map. The distribution of N deposition from this NH_3 is even more complex (Dragosits and others 2002), due to the fact that deposition rates depend on vegetation roughness and land management. The largest N deposition (and hence impact) occurs on woodland and moorland edges adjacent to agricultural sources.

In the context of needing to know the exposure and impacts of atmospheric N at individual statutory nature conservation sites, these results demonstrate the importance of local site-based measurements of NH_3 concentrations. In upland areas, a similar spatial variability in N wet deposition is expected, and here site-based wet deposition estimates become important in estimating total N deposition reliably.

Bioindicators for ammonia and nitrogen deposition

A detailed review of methods (Pitcairn and others 2003) identified three broad types of bioindicator for atmospheric nitrogen:

1. **Biochemical methods**, which sample a component of a species or species-group that occurs in the habitat. These include both parameters reflecting an *accumulation of nitrogen* in the system and *chemical/physiological responses* to nitrogen exposure.
2. **Species composition methods**, which characterize the presence and extent of occurrence of certain species, each of which have been previously categorized as having different nitrogen preferences.
3. **Transplant methods**, where either locally native species or standardized plants are exposed to a range of nitrogen deposition/concentrations and their responses assessed.

Methods in each of these categories may be variously suited to indicating atmospheric deposition, concentrations or environmental effects, while the application of these methods in time provides the basis for biomonitoring for nitrogen, typically over a period of several years or decades.

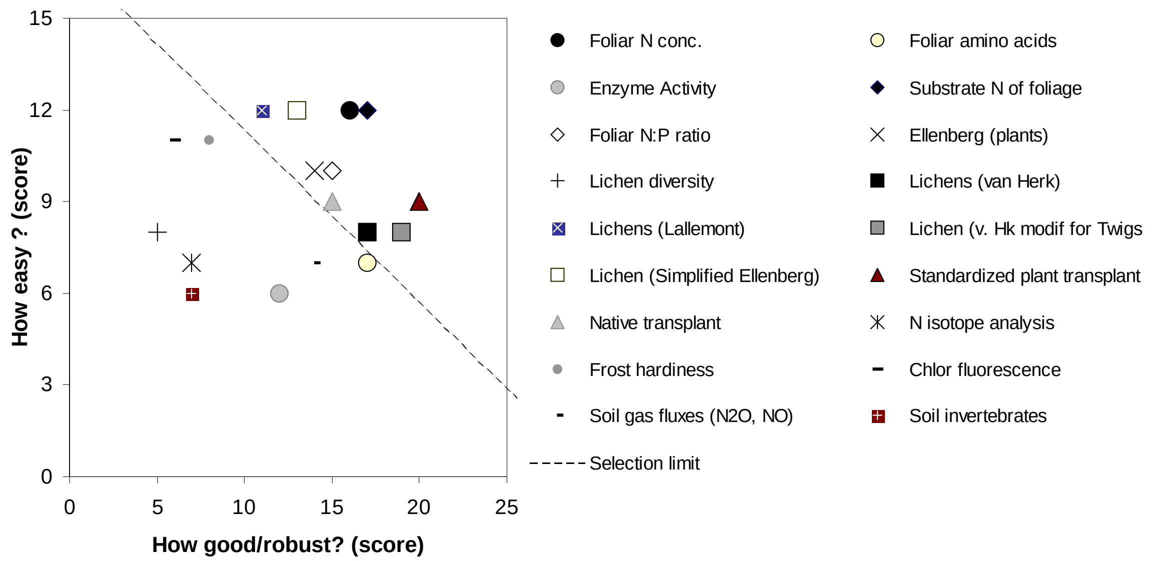

Figure 3: Empirical assessment for the general application of different N bioindicator methods (from Pitcairn and others 2003). The highest scoring (best) methods are shown to the top-right of the figure.

Two approaches were used to assess the applicability of a wide range of bioindicator methods for use in site-assessments by the Conservation Agencies (Pitcairn and others 2003). Firstly, methods were rated for general practical application, using an empirical scoring system, trading-off issues for each method of 'how easy?' *versus* 'how good/robust?'. Secondly, based on a broad view of the most robust methods, a 'decision tree' approach was used to select methods suited to different contexts and questions. Figure 3 provides a view on the general empirical scoring of the methods, with the diagonal line indicating that a selection limit would justify a more expensive method where it is proportionately more robust. The limits of such an assessment mean that it should not be over interpreted. However, it provides a means to help identify a short list of recommended methods. The scoring also highlights the limitation of biochemical or species response based bioindicators, which were not specific to nitrogen (eg frost hardiness, chlorophyll fluorescence and lichen diversity). Similarly, ecosystem responses broadly specific to nitrogen (for example soil emissions of N_2O, enzyme responses), tended to be less suited to indicating N deposition than N accumulation based approaches (foliar N, foliar soluble N, amino acids). One reason for this is that the biological response is usually a function of both the N accumulation (more directly measured by accumulation based methods) and other factors, such as moisture, light, soil type etc.

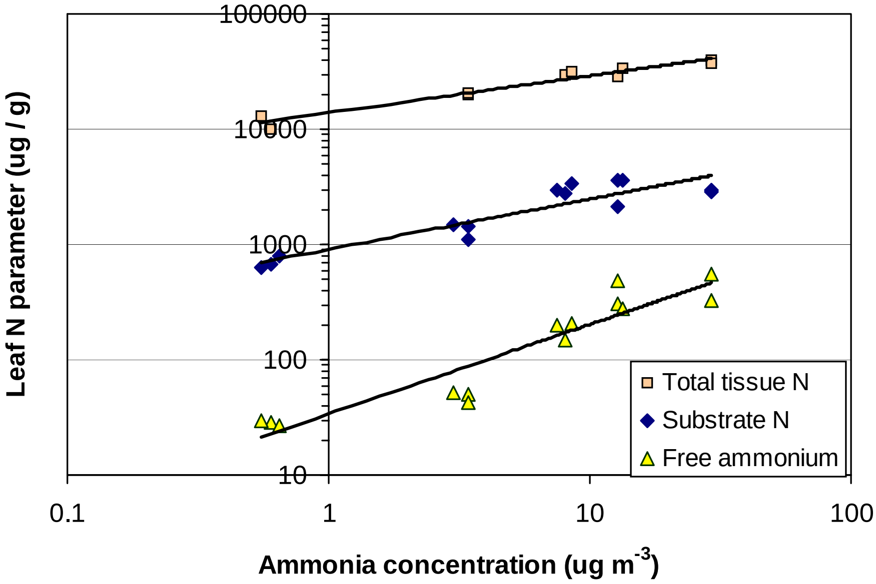

Figure 4: Comparison of three N bioindicator methods (total foliar nitrogen, substrate N and foliar NH_4^+) for the moss *Rhytidiadelphus triquetrus* along a transect downwind of a poultry farm (Pitcairn and others 2003). The largest relative response is seen for foliar NH_4^+, consistent with it representing the smallest nitrogen pool of the three measures.

Biochemical bioindicator methods for nitrogen

Probably the best-studied biochemical bioindicator for nitrogen is the total N content of dried foliage. This has been shown to respond to nitrogen deposition, particularly in *Calluna* and bryophyte species (Pitcairn and others 1995), while there is evidence that tissue N content also reflects the N habitat preference of the species concerned for a wide range of other species. Uncertainty in the literature regarding foliar N concentration appears to largely reflect the variability between species, time of year and protocols for sampling, as well as uncertainty in atmospheric deposition estimates. For this reason, practical application of the approach is best focused on target species groups (for example pleurocarpous mosses) using carefully standardized sampling protocols.

While foliar amino acids have been recognized for some time as providing good bioindicators for N accumulation in response to N deposition, these differ widely between species making it difficult to generalize. In contrast, newly recognized parameters include ‘substrate N’ and total foliar ammonium (eg Riedo and others 2002, Hill and others 2001). The former may be approximated by measurement of soluble N, and reflects the surplus N available for growth, while the latter represents mainly the recycling pool within the plant metabolism for synthesis of a wide range of nitrogen compounds. In principle, the smaller the pool size and more direct the link to atmospheric N deposition, the larger the N response, and this is demonstrated by measurements comparing these different parameters. Adjacent to a poultry farm, where NH_3 provided the basis for an increase in total N deposition, total N increased by a factor of 3, substrate N by a factor of 5 and foliar NH_4^+ by a factor of 20 (Figure 4). Although the data in

Figure 4 are for the moss *Rhytidiadelphus triquetrus* and there were differences in mean values between species, the available data demonstrated a single overall response for pleurocarpous mosses for each of substrate N and foliar NH_4^+, with different species occurring at different distances from the farm.

Species composition based bioindicator methods for nitrogen

The most well-established approach to indicating the availability of nitrogen using species surveys is the allocation of Ellenberg nitrogen scores for higher plants (Ellenberg 1988) and bryophytes (Siebel 1993). In this approach, each species is allocated a preference score for nitrogen (1 = oligotrophic preference; 9 = eutrophic preference), so that the average Ellenberg score for a site may be calculated from the full list of species present. A modification is to utilize data on the relative species abundance and calculate, for exsample a cover-weighted Ellenberg score. This approach is well suited to long-term monitoring assessments or small scale transects near point sources, but, because of the many other factors affecting species presence, particular caution is required in its interpretation, especially between habitats and on regional scales.

Epiphytic lichens are well known to be sensitive to air pollution, with a long history of research into the effects of SO_2. More recently, it has become clear that lichens are equally sensitive to ammonia in the atmosphere (Herk 1999). While part of the response may be due to the increased availability of nitrogen, it appears that the main effect is the basic action of NH_3 in increasing bark pH. Overall lichen diversity does not provide a good indicator of NH_3 effects, as species favouring clean naturally acidic bark (“acidophytes”) tend to be replaced at high NH_3 levels with those favouring higher bark pH and nitrogen availability (“nitrophytes”). Van Herk (1999, 2001) has shown clear relationships for the lichen flora on oak trunks for the Netherlands, and recent studies in the UK (Wolseley & James 2002, Pitcairn and others 2003) have shown some success applying the van Herk (1999) scoring system.

In particular, the UK results demonstrate importance of comparing the more usual trunk epiphytes with lichens growing on twigs (Wolseley & Pryor 1999). This is illustrated in Figure 5, which summarizes the changes in the trunk and twig lichen flora through a mixed-woodland downwind of a poultry farm (Pitcairn and others 2003). The results from the chicken farm show how nitrophyte species (NIW) increase with NH_3 concentration, while acidophyte species (AIW) decrease with NH_3 concentration, with the loss of acidophytes occurring at lower concentrations than the increase in nitrophytes. The overall extent of nitrophily of the lichen flora is shown in Figure 5c, by AIW minus NIW. It is important to note that while the trunk acidiophyte community survives up to around 8 $\sigma g\ m^{-3}$, the twig acidophytes are completely lost above 2 $\sigma g\ NH_3\ m^{-3}$. This is associated with a higher bark pH of the twigs, indicating that acidophytes are unable to colonise the twigs. Overall, it can be seen that a decline in the natural acidophyte lichen flora (both twigs and trunks), occurs at mean concentrations less than 2 $\sigma g\ m^{-3}$, which is substantially less than the current UNECE critical level of an annual mean of 8 $\sigma g\ m^{-3}$.

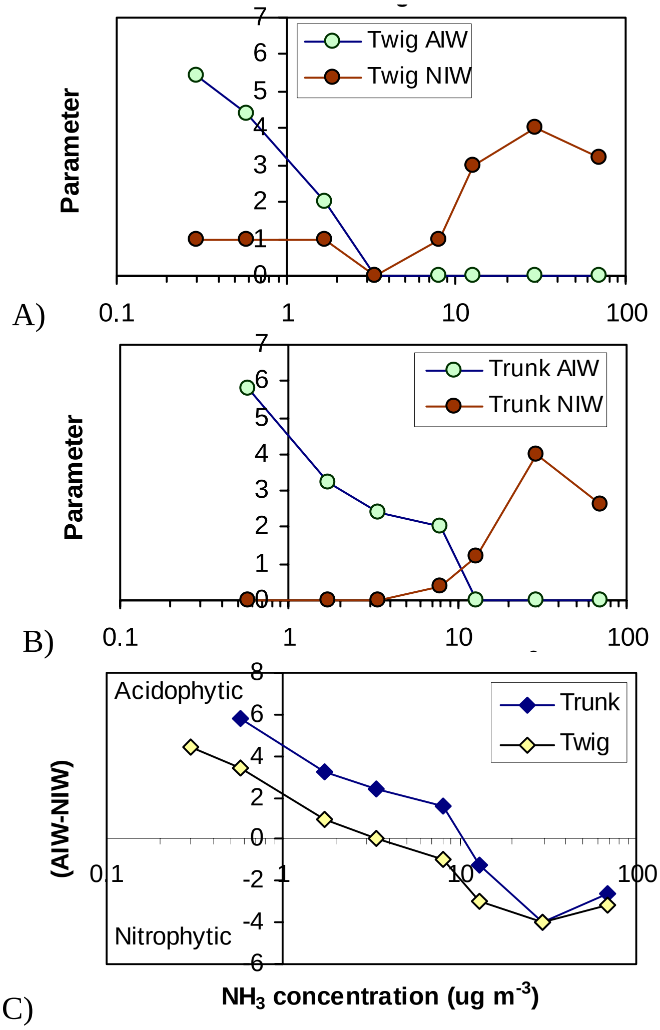

Figure 5: Occurrence of lichens on the trunks and twigs of a mixed-woodland adjacent to a poultry farm in Scotland (Pitcairn and others 2003). A) Lichens on trunks were scored using the van Herk (2001) approach, while B) lichens on twigs were scored using a modified sampling procedure of Wolseley and Pryor (1999), with in both cases the AIW index representing 'acidophytic' species (which prefer naturally acidic bark) and the NIW index representing 'nitrophytic' species (which prefer nitrogen enriched more basic bark resulting from enhanced NH_3 concentrations). C) Subtraction of AIW-NIW provides an overall index of whether the flora is acidophyte or nitrophyte dominated.

Transplant based bioindicator methods for nitrogen

Transplant based bioindicators provide a specific approach that may be useful for the demonstration of nitrogen effects in certain critical situations. The two basic approaches are a) use of standardized model plants or b) transplantation of naturally occurring species between locations. In the former, *Lolium* has successfully used as a biomonitor for nitrogen deposition (Sommer & Jensen 1991, Sutton and others 1993). Applying this approach along the same poultry farm transect mentioned above showed a two-fold increase in above-ground biomass immediately adjacent to the farm, compared with plants set out 300 m distant (Pitcairn and others 2003). This provides a dramatic demonstration of NH_3 effects for stakeholders. Shading is a possible interaction, leading to reduced growth and higher tissue N contents. However, these effects were found to be largely cancelled out by measurement of the total above-ground N inventory of the plants, substantially improving the correlation with

NH_3 concentration (Figure 6). A key advantage of this approach is that results may be obtained using standard material over periods of several weeks.

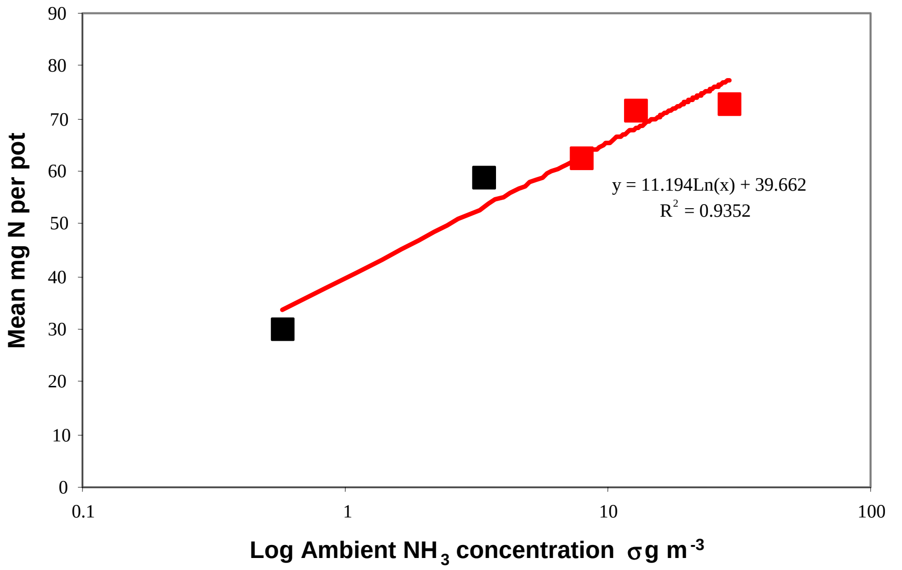

Figure 6: Grass biomonitors for nitrogen: The total N inventory in shoots of *Lolium perenne* showed a clear relationship to NH_3 exposure following 6 weeks exposure downwind of a poultry farm (Pitcairn and others 2003).

The use of reciprocal native transplants requires longer periods (for example one year) and significant resources, but is well suited for epiphytic bryophytes that are easily transplanted. This approach was applied by Mitchell and others (2003), who transplanted three bryophyte species between high and low N deposition sites. They showed both increases and decreases in foliar N content on transplanting to clean and polluted sites, respectively. Importantly, they also showed matching changes in growth rates. Hence in addition to showing the negative effects of nitrogen on sensitive species, a key advantage of this method is that it can provide a demonstration of recovery following a reduction in N deposition. This method may therefore be helpful to demonstrate to stakeholders the benefits of proposed investment in N pollution abatement.

Discussion: Further development of Bioindicators and Biomonitoring for Nitrogen

The high spatial variability in NH_3 concentrations and deposition provides a challenge to assess the impacts of ammonia on a site-by-site basis. This is particularly relevant for the Conservation Agencies, who are charged with protecting statutory nature conservation sites, such as Sites of Special Scientific Interest (SSSIs) and Special Areas of Conservation (SACs). While physical monitoring will usually provide the cheapest (and most direct) means to estimated NH_3 concentrations (eg Sutton and others 2001), bioindicators and biomonitoring can play an important role in providing site based evidence of NH_3 concentrations, N deposition and ecological effects. For example, estimating N deposition is complex, particularly in situations with strong local gradients near sources or at woodland edges. In these contexts, particularly accumulation based biomonitoring methods may provide an alternative estimate of deposition to a site, for comparison for example with a site-based critical load. The challenge here must be to improve the relationships between deposition and foliar N responses, focusing on specific species groups (for example pleurocarpous mosses) in order to be able to improve the predictive ability of the approach.

Fundamentally, although an increase in foliar N levels may suggest an increase in likelihood of species change, the measurement of actual species change is a much stronger parameter to demonstrate a loss of 'favourable condition' of an SSSI or SAC. The difficulty in the use of Ellenberg approaches for higher plants is that there are so many other factors that affect plant communities. Therefore, whereas Ellenberg changes may be observed, care is needed to investigate whether the cause of change is really increased nitrogen availability from atmospheric deposition. In this respect the N accumulation-based bioindicators and species-based bioindicators are complementary and the strongest evidence of change would be provided by a combination of both approaches.

Lichens provide a specific example relevant to the assessment of atmospheric ammonia exposure and effects. By considering epiphytes growing on specific tree species, the complicating effects of soil differences are largely avoided. At the same time, the differences between trunks and twigs is very informative, since the latter are more sensitive to ammonia and may respond more quickly, with trunk communities reflecting to some extent the pollution climate over previous years and decades. A key point is that lichen species composition appears to respond to the effect on bark pH due to NH_3. Hence the effects of ammonium aerosol or wet deposition (which due to some nitrification may have an acidifying effect) may be opposite to that for NH_3. Similarly, the effects of oxidized nitrogen on lichens are at present highly uncertain. Despite the limitation regarding other forms of nitrogen, the evidence for NH_3 effects on lichens is now extremely strong and has been used for biomonitoring for a number of years (van Herk 1999).

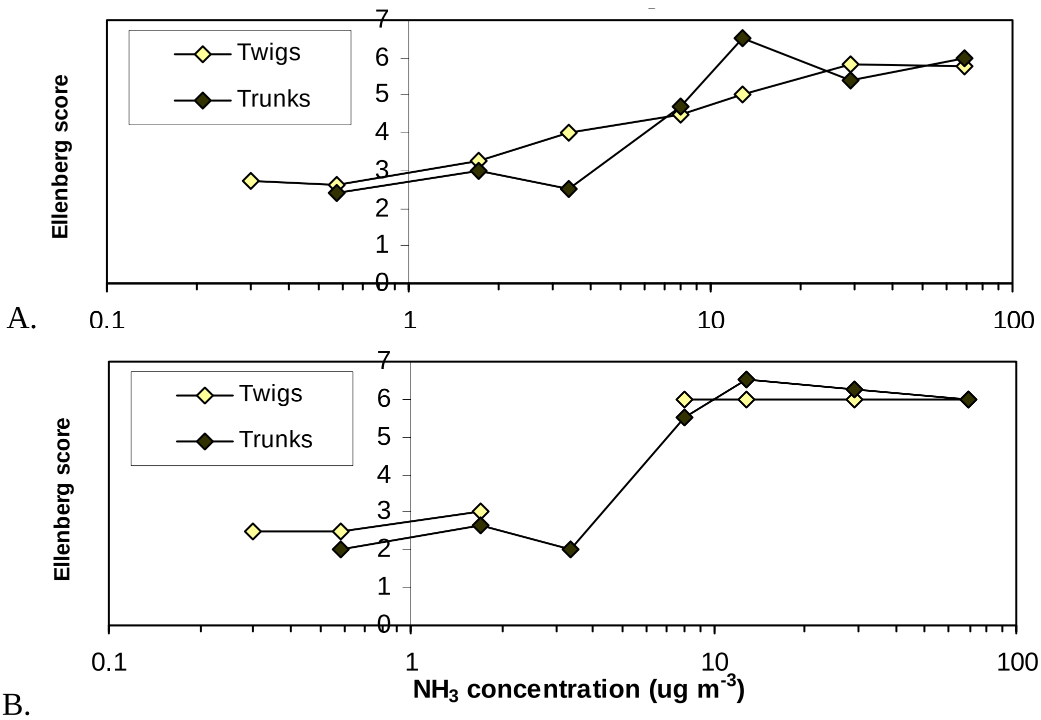

Figure 7: Application of the 'Ellenberg' scores for lichen flora (Wirth 1992) to a transect through a deciduous woodland adjacent to a poultry farm in southern Scotland (Figure 5). A) overall site Ellenberg index based on presence of all species identified; B) simplified Ellenberg index based on only easily identifiable foliose and fruticose lichen species.

A key feature of biomonitoring, however, is the need for practical methods that can be implemented cheaply and robustly for conservation purposes. Further work is necessary here to improve and standardize protocols as well as to simplify methods. The example of the lichen response to NH_3 provides a good example of this. Using nitrophyte and acidophyte indicators defined by van Herk (1999, 2001) produces results which are extremely sensitive to low levels of NH_3 (Figure 5), but is biased towards continental species, is time consuming and includes crustose species which require an experienced lichenologist to identify. The Ellenberg scoring system has been applied to lichens by Wirth (1992), and can be applied more simply based on presence/absence of species scored as nitrophytes. This method requires identification of all lichen species present. A further simplification focuses on macrolichens of the foliose and fruticose habit. The attraction of this kind of approach is that it becomes amenable to the non-specialist, raising awareness of the effects of NH_3 on lichens and allowing quick initial assessment of sites. This approach is illustrated in Figure 7, which compares the lichen scoring for the transect at the poultry farm shown in Figure 5. The full Ellenberg scoring system shows a clear response to NH_3, and is also shown approximately by the "simplified Ellenberg" scheme. Obviously, the simple approach has limitations (eg there is a gap where macrolichen species occur at one site in the wood). However, this approach demonstrates the potential to widen-out the bioindicators debate from the preserve of specialists to a practical engagement with people interested in the environmental effects of excess atmospheric nitrogen.

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Trends in Nitrogen use in Agriculture

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Summary

Nitrogen is a key macro-nutrient essential for plant growth. Large quantities of inorganic N fertilisers are used by farmers to ensure adequate crop yields and economic return. However, the N cycle is inherently leaky and surplus N in the soil, not used by crops, is at risk of loss to the environment. Livestock systems are particularly prone to N losses. This is especially so for ammonia (NH_3). Once NH_3 has been emitted it can be transported in the atmosphere and deposited in either gaseous or aqueous forms onto sensitive ecosystems and cause shifts in plant community structures. Lichen communities can also be adversely affected. Excess N remaining in the soil can also be lost to the atmosphere as nitrous oxide (N_2O), a greenhouse gas, and to water as nitrate.

This paper discusses trends in fertiliser N use and livestock numbers over the past 25 years in order to provide background information with which to assess changes in lichen species abundance. We also discuss why so much N is used in agriculture. There is growing concern about the fate of excess N in the environment, particularly at the European level. Therefore, legislation is being put into place to control N emissions. There are several livestock, fertiliser and manure management options that are being considered as alternatives to maintain income but reduce the risk of N transfers to air and water. These are also discussed.

Trends in fertiliser use and animal numbers

In the UK, N use increased considerably after World War II when production was the main goal. N fertiliser use increased between 1975 and the early 1980's, when it peaked at c. 1.8 million tonnes of N. Since then, N fertiliser use has decreased gradually to a present figure of *c*. 1.2 million tonnes. At the same time livestock numbers have fluctuated. For example, pig and beef cattle numbers have fallen gradually over this period, whilst the number of poultry and horses have increased. The number of dairy cows has fallen from 3.9 million in 1975 to 2.9 million in 2002. Despite the decline in the number of dairy cows, milk production has not decreased as the average milk yield per cow has increased from *c*. 4700 to 6800 litres per recording year. In order to achieve these greater milk yields, the genetic merit of cattle has been altered and they need to consume greater quantities of protein. Therefore, the quantity of N excreted by the dairy herd has not decreased by as much as would be expected despite the decrease in animal numbers, ie from *c*. 289,000 to *c*. 215,000 tonnes.

The present estimate of N excreted by UK agricultural livestock (cattle, pigs, poultry, goats and horses) is 950,000 tonnes. Approximately 50% of this N is deposited in fields by grazing and outdoor animals. However, c. 450,000 tonnes of this N is collected in animal houses and from yards and is applied to soils via animal manures each year. The present challenge is for farmers to use this form of N effectively within their nutrient management plans.

Consequences of excess N use

As a result of increased fertiliser use, imported feedstuffs for livestock and intensification on limited land areas, many farms are operating at nutrient surpluses, particularly for N and P. Since the N cycle is leaky, any N not used by growing crops or assimilated by livestock, is at risk of loss to the environment. The major N emissions to the atmosphere are NH_3, N_2O, di-nitrogen (N_2) and nitric oxide (NOx). Di-nitrogen is environmentally benign. However, NH_3 emission followed by transport in the atmosphere can result in deposition to fragile ecosystems which become enriched with N. The effect of this N deposition can be soil acidification and loss of flora adapted to low soil nutrient conditions in favour of more competitive grass species. Nitrous oxide is a greenhouse gas and agriculture is now thought to be responsible for 50% of the total UK emission. Losses of nitrate, nitrite and ammonium to watercourses by leaching and runoff cause eutrophication and poor water quality for Cyprinids and Salmonids.

Only a small proportion of applied N ends up in agricultural product. For example, for a beef system operating on a clay loam soil, 300 kg N can be applied as N fertiliser in one year. Model calculations (Scholefield and others 1991) using the NCYCLE model, show that only 10% of the applied N ends up in the product (ie in meat). Up to 40 kg can be lost as NH_3, 50 kg lost through denitrification (ie $N_2O + N_2$) and 60 kg leached as nitrate.

Major research programmes have been directed at quantifying emissions of N species to air and water and developing agricultural practices that reduce such losses in the UK. Several European Directives are now in place to bring about changes in agricultural management to reduce the impacts of N on the environment. For example, the protocol to Abate Acidification, Eutrophication and Ground-Level Ozone (Gothenburg Protocol) of the UNECE Convention on Long-Range Transboundary Air Pollution was signed in 1999. This Protocol sets annual emission ceilings on four pollutants, including NH_3, which are to be met by 2010. The UK annual target is 297,000 t of NH_3.

Why is so much N used in agriculture?

There are several reasons why so much N is used in agriculture. Some of these are listed and discussed briefly below.

- Crops are responsive to N – ie there is a good relationship between N fertiliser rate and crop yield. However, there is an optimum application rate after which the economic returns in agricultural product become small.
- N use is inefficient – crop uptake of fertiliser N ranges from <65% - >95%, only about 60% of the plant N is harvested (in above ground herbage), rumen capture is only *c*. 30% efficient. Hence, farmers often apply more than what is really required to ensure adequate productivity and safeguard economic return.
- Nutrients (eg N) in manures are not always integrated wither inorganic fertilisers in the nutrient management of the farm. ie farmers lack confidence in using manures as a source of N and hence apply manure plus mineral fertilisers (Smith and others 2001). This leads to surplus N in the soil.
- The inorganic N content of manures is readily available, hence this form of N acts like inorganic fertiliser N. However, up to 90% of the N content of a manure can be in an organic form (Chadwick and others 2000). This is not immediately available to crops

and requires breakdown (mineralisation) by soil micro-oranisms to forms that are plant available. The rate of N release from organic forms is difficult to predict (Chadwick and others 2000). Hence, farmers cannot account for this in their nutrient management plan for the farm and excess inorganic N fertiliser is applied.

∉ The N cycle is inherently leaky. Losses of N can occur from soil even when only small quantities are applied. Hence farmers need to take these losses into account and can be tempted to apply more than the crop requires.

How can N emissions be reduced?

∉ Nitrogen fertilisers can be used tactically, ie either the soil mineral N content is modelled or measured and the N fertiliser application based on the quantity required to increase the soil N content to the level required by the crop. This method of N management has been tested on commercial farms and resulted in 28% less fertiliser N use. The consequences of the reduced N applications was a 38% decrease in nitrate leaching whilst maintaining productivity (Brown and others 1997).

∉ The UK Government has funded a large number of projects to assess the magnitude of the NH_3 sources from agriculture (Defra 2002). The construction of inventories has shown where abatement practices may be most effective. Table 1 illustrates the sources of NH_3 in the UK in 2000. This table shows yards as a source, although they are not yet included in the official inventory.

Table 1. Sources of NH_3 emissions from UK agricultural and non-agricultural sources (2000)

Source	NH_3 losses, (t * 1000)	% of total
Housing	92.5	26.5
Yards	24.8	7.1
Manure stores	15.2	4.3
Manure spreading	102.9	25.5
Fertilisers	32.3	9.3
Outdoor animals	26.7	7.6
Non-agricultural sources	54.9	15.7
Total	349.4	100

∉ Recent research suggests that NH_3 emissions can be reduced from cattle housed on increasing amounts of straw. A doubling of straw can reduce emissions by 50% (see Figure 1).

∉ Collection yards are now thought to be a significant source of NH_3 emissions (Misselbrook and others 2001). Effective hosing and scraping of these yards reduces emissions.

∉ The use of fixed and floating slurry store covers can also reduce emissions considerably, by up to 80%, and reduce rainfall entry to the store. However, fixed covers are expensive, so crust development on dairy slurry stores may be a cheaper alternative.

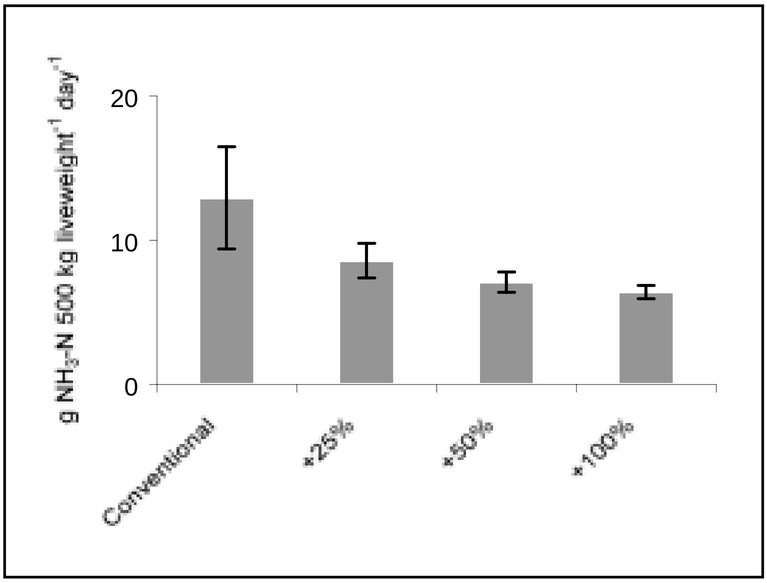

Figure 1. Ammonia emission rates from cattle housed on increasing amounts of straw.

- The spreading of slurries by alternative techniques, eg shallow injection, trailing shoe or trailing hose is known to reduce emissions compared to surface spreading (Figure 2). Shallow injection and trailing shoe are particularly efficient at reducing ammonia emissions on grassland.
- Incorporation of slurries and solid manure on arable land by ploughing (or similar cultivation techniques) soon after spreading is another effective method of reducing emissions.

The predicted 'natural' reduction in fertiliser use and in animal numbers may reduce the total NH_3 emission close to the annual target value of 297,000 tonnes. However, this alone may not be sufficient and some of the livestock and manure management options discussed above may also be required to achieve the necessary reduction.

Trailing hose

Deposits slurry in discrete bands on the surface of soil and vegetation. Typically reduces ammonia emissions on grassland by *c*. 30% compared to surface spreading.

Trailing shoe

The ‘shoe’ parts the sward and the slurry is deposited in bands on the soil surface. The grass canopy then closes back over the slurry reducing air flow. Typically reduces ammonia emissions on grassland *c*. 60% compared to surface spreading (depending on sward height).

Shallow injection

Tines cut narrow ‘V’ shaped slots *c*. 6 cm deep in the soil and slurry is placed in the slots. Typically reduces ammonia emissions on grassland by *c*. 75% compared to surface spreading.

Figure 2. Slurry spreading techniques to reduce ammonia volatilisation.

Conclusions

N fertiliser use and animal numbers are reducing, yet excess N is still applied to agricultural land. There are options to reduce N transfers to water and air. Providing farmers with confidence in the use of manures as sources of nutrients will encourage them to integrate manure and fertiliser use, and this is key to greater efficiency of N on farms. There are also real cost savings to be made through the optimal use of manure N, P and K. Other specific measures can be used to reduce emissions of NH_3 from various stages of manure management, but these are likely to be a more expensive option.

Acknowledgements

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Session 2 Discussion

Group 2. The pollution environment

Participants: Kok van Herk (chair), Linda Davies, Alastair Headley, Barbara Hilton, Mark Seaward, Lucy Sheppard, Mark Sutton (raconteur), Alison Vipond, Amanda Waterfield, Clare Whitfield.

Questions

- What should the NH_3 thresholds be for the UK?
- Relative importance of NH_3 and NOx in terms of impacts on lichens?
- How far does NH_3 travel before it is deposited?
- What can be done for sites that already exceed critical loads?
- What is the way forward in developing lichen monitoring schemes for atmospheric N?
- What is the right balance between a national policy to reduce UK ammonia emissions and local policies to protect specific sites?

What should be the NH_3 threshold for the UK?

Inevitably different thresholds for different situations. Should there be differentiation for the kind of land? for example in a circle of a farm (20 µg m-3), or in agricultural areas (10 µg) or natural areas (for example 5 µg).

Thresholds must depend on defining what is to be preserved, so need to know the assemblage to be protected, for example protection of *Bryoria* would require very big emission reductions that could only realistically be achieved for specific nature conservation sites through local mitigation strategies. Need an optimal level of N, but optimal for what?

What is the critical level ? It was agreed that 8 µg is far too high for areas such as Scotland.

How far does NH_3 travel before it is deposited?

Important to realize that the decrease in NH_3 concentrations away from sources is NOT primarily due to the loss due to deposition, but due to dispersal and dilution. While around 5-40% of the ammonia emitted is re-deposited back to the same 5 km grid-square, conversely around 30-40% of the NH_3 emission is exported from the UK, largely in the form of its reaction product ammonium aerosol.

A key concern about ammonia is its widespread distribution across the country in the rural environment: –as agriculture is right across the country, so emissions are very widely dispersed. Only areas with little animal agriculture have low ammonia levels.

Relative importance of NH_3 and NOx in terms of their impacts on lichens?

NH_3 kills acidophytes but there was general uncertainty in the group about the extent and nature of impacts of NOx. The effects of NH_3 are very clear, conversely, the NOx effects are much less clear. In recent years, further confusion has been added to this in that NH_3 is now emitted by catalytic converters and at roadsides contributing a large fraction of the N captured by vegetation. NH_3 is by far the most important but NOx also important due to NO_3 in precipitation (long range). It is debatable as to whether there is a significant local effect of NOx.

What can be done for sites that already exceed critical loads?

How can we set national targets for air quality, for lichens rather than human health. Can't go back to pristine state so can we set targets for what we are aiming to achieve for lichen diversity, for example could some eutrophication be good to increase diversity?

What is the right balance between a national policy to reduce UK ammonia emissions and local policies to protect specific sites?

This will depend on the balance of costs. Some local landscape measures may be cheaper than technical abatement.

What is the way forward in developing lichen monitoring schemes for atmospheric N?

How to market lichen monitoring – difficult to market – has substantial needs and costs. Although one view is that lichen monitoring cannot be done cheaply without trained lichenologists, another favoured the development of simple pollution scale for lichens and NH_3. Constraints: it may be possible to do this for a rising or stable pollution environment but very difficult to do on a falling pollution load.

Sampling methods discussed and complete lists versus selected species. Former can be applied for other purposes than for which it was originally collected (for example climate change). Dutch map based on 6000 sites for each of 10 trees. It took 5 years work for one lichenologist. (ie 60,000 trees)!

Be aware of the suitability of the measuring point – results may be biased if only focusing on where red-listed species occur, - we need to establish a sensible random sampling protocol in order to establish a comprehensive monitoring system, in the same way as establishing a weather recording system.

4. Selecting and monitoring species and communities

Global ecological patterns of nitrophytic lichens

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Abstract

Nitrophytic lichens are strongly unequally distributed among taxonomic groups. Most belong to the *Physciaceae* or the *Teloschistaceae* and, inversely, most species belonging to these families are nitrophytes.

The wide (often cosmopolitan) distribution of many species of the *Physciaceae* and *Teloschistaceae* make them potentially useful monitoring organisms for hypertrophication over wide distances. However, many species behave differently in different regions and on different substrata, where they have different thresholds regarding hypertrophication, in relation to the nutrient availability of the unpolluted substratum.

Introduction

Nitrophytic lichens are strongly unequally distributed among taxonomic groups. Most belong to the *Physciaceae* or the *Teloschistaceae* and, inversely, most species belonging to these families are nitrophytes, albeit not all to the same extent.

There is also a strong geographical dimension to this. *Physciaceae* are the dominant lichens on hypertrophicated bark in the tropics (for example *Pyxine* and *Dirinaria* on Coconut palms in coastal city parks), as well as on wayside trees in subtropical and temperate regions (for example dominance of *Physcia*s and *Phaeophyscia*s). *Teloschistaceae* are the dominant lichens on rock in subtropical (for example *Caloplaca*s on limestone), temperate (for example *Xanthoria*s on brick walls) and arctic areas (for example *Xanthoria elegans* and its many morphotypes described from Antarctica).

The wide (often cosmopolitan) distribution of many species of the *Physciaceae* and *Teloschistaceae* make them potentially useful monitoring organisms for hypertrophication over large areas.

Observations

The knowledge about the actual distribution of many lichen taxa is severely hampered by imperfect and unequal sampling, availability of literature or herbarium material (see Tables 1-3, based on expert judgements by various specialists). No reliable global distribution maps are available to date, not even of well-known and easily recognizable species such as *Physcia caesia* and *Xanthoria elegans.*

Table 1. Regional importance of the major nitrophytic lichen groups by major geographical areas

	Physciaceae	*Teloschistaceae*
Africa	high	medium
Antarctica	low	high
Asia (temperate)	high	medium
Asia (tropical)	medium	low
Australasia	medium	medium
Europe	high	high
North America	high	high
Latin America	high	medium

Table 2. Knowledge of the major nitrophytic lichen groups by major geographical areas (excellent = monographs available, consistently treated among floras; very good = revisions and floras available for most areas and most groups; good = revisions and floras available for some areas and many groups; fair = revisions available for only a few groups, floras very limited; poor = very few revisions available, floras (if any) very restricted)

	Physciaceae	*Teloschistaceae*
Africa	fair	poor
Antarctica	good	very good
Asia (temperate)	good	fair
Asia (tropical)	fair	poor
Australasia	good	fair
Europe	very good	very good
North America	very good	good
Latin America	fair	poor

Table 3. Availability of collections of the major nitrophytic lichen groups by major geographical areas (excellent = present as major components of most herbaria; good = present as major components of only a few herbaria; fair = rarely collected well; poor = very poorly collected)

	Physciaceae	*Teloschistaceae*
Africa	good	fair
Antarctica	good	excellent
Asia (temperate)	good	good
Asia (tropical)	fair	fair
Australasia	good	good
Europe	excellent	excellent
North America	excellent	good
Latin America	fair	poor

Therefore, another approach is taken in this paper. Based on literature reports and personal observations, diagrams showing the ecological preferences of three species at different latitudes (on the Northern hemisphere) are presented. A characteristic behaviour of nitrophytic lichens is that they tend to be rather catholic as to the actual substratum. Exposed, eutrophicated rocks can bear a lichen flora that is nearly identical to that of trees in the neighborhood. This phenomenon, often referred to as a 'special' case of, for example atlantic regions or extremly cold environments, is general rather than specific and can be as easily observed in lowland tropics as in the high mountains (not in the arctic in the absence of trees, but here saxicolous taxa manage to appear on the soil and on branches of dwarf shrubs).

However, there are marked differences between the various species. Of the three species presented here, only *Xanthoria parietina* is roughly equally common on tree bark as on rock. *Xanthoria elegans* occurs only on rock, while *Physcia caesia* occurs widely and predominantly on rock, and only appears on tree bark in hypertrophicated situations.

Examples of patterns

Many species behave differently in different regions and on different substrata, where they have different thresholds regarding hypertrophication, in relation to the nutrient availability of the unpolluted substratum. This can be easily observed from variation in *Xanthoria parietina* (Figure 1a), which is absent from real tropical or arctic habitats. On the equator, it only grows in subtropical areas at c. 2000 m alt. (Swinscow & Krog 1988), but on both tree bark and rock. In boreal areas it only grows on nutrient-enriched substrates like *Populus* bark (receiving balsam effluent) and on rock only at bird-perches. In temperate regions, it grows on unpolluted rock and stone (for example vertical brick walls) and bark of trees with a high pH. In temperate areas, it can thus be used as an indicator of eutrophication when it occurs on acid bark. In subtropical regions, it grows naturally on virtually all substrata, varying from limestone to siliceous rock outcrops and on all kinds of broadleaf trees and shrubs. Here, only its occurrence on highly acidic bark, like pine trees, is indicative of eutrophication. To make things more complicated, there is also a distinct effect of the proximity to the coast, which may however be explained by an actual eutrophication effect of nutrients deposited by salt spray.

a) b)

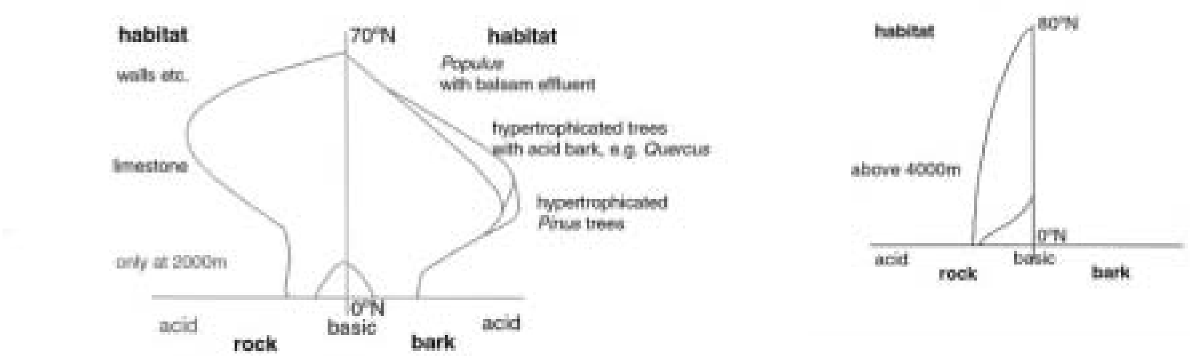

Figure 1. Ecological amplitude of a) *Xanthoria parietina* and b) *Xanthoria elegans*

Despite the frequent use of the term bipolar distribution (Galloway & Aptroot 1995, Murtagh and others 2002), nearly all putative examples of this distribution pattern (for example *Xanthoria elegans*) prove upon closer examination to be essentially cosmopolitan with differential restrictions within oroclimatic zones. *Xanthoria elegans*, for example, occurs everywhere in the arctic to subtropical zones in both hemispheres, but is restricted to higher altitudes (4000 m, orotemperate) in the tropics, where it is, however, known from all three tropical continents. Figure 1b shows its continuous occurrence on the Northern hemisphere from the equator up to 80º North, demonstrating a shift in its pH requirements with latitude.

The third species, *Physcia caesia,* exemplifies both a strongly asymmetrical spatial ecological pattern and a temporal change in habitat requirements. Figure 2 shows that it too, like the preceding species, occurs on rock from the equator up to nearly 80º North, also changing its pH requirements. In addition, however, it can be found on tree bark in temperate regions. On this substratum it is rare under natural trophic situations, but the species becomes a common element of the epiphytic lichen flora in hypertropicated situations. An example is its occurrence in the Netherlands: in the 18th century it was known from only one tree although it was known to be a common saxicolous species. In the second half of the 20th century its occurrence on trees expanded, for instance its frequency increased by a factor five between 1979 and 2001 from 6.5 to 33% in the Province of Utrecht (van Herk and others 2002). Now, it is one of the most common epiphytic lichens of exposed trees influenced by local (for example dogs) or regional (animal husbandry) ammonia pollution. The wide distribution of this species, especially in the N hemisphere, suggests that this species has potential use as an indicator of hypertrophication over large areas.

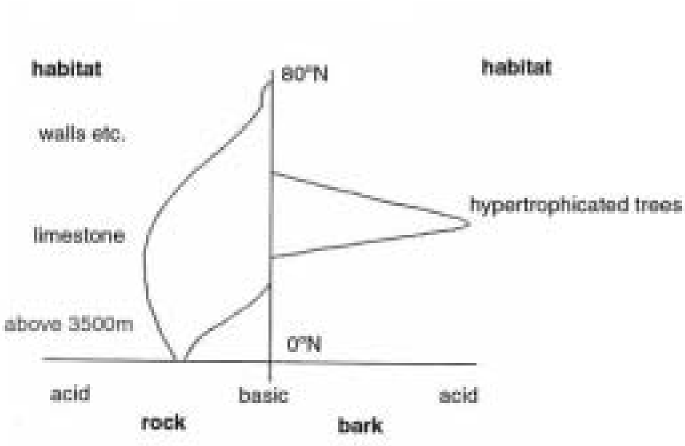

Figure 2. Ecological amplitude of *Physcia caesia*

Application and framework for future research

Despite the unequal distribution of taxonomic and distributional knowledge of species occupying nitrophytic habitats, the occurrence of cosmopolitan nitrophytic species suggests that these can be used over wide areas in different oroclimatic zones to detect changes in atmospheric conditions.

In order to be an effective tool, however, detailed ecological and distributional data should be collected over large areas to provide a framework to evaluate shifts in conditions. Many nitrophytic macrolichens are well-known and widespread, and more attention should be paid to the natural occurences of these weedy species. Strangely enough, right now there is more detailed information about the autecology of, for example, *Xanthoria elegans* in Antarctica than in the rest of the world.

Once abundant information becomes available, the question WHY species behave differently in different (oro)climatic zones may be addressed.

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Lichen indicator species of environmental conditions in Liguria (N-Italy)

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Summary

Recent studies in the field of lichen biomonitoring suggested that analysis of lichen vegetation provides heterogeneous information about the state of the environment. Separation of pollution response from the "noise" given by other influential environmental factors is a recurring problem. However, in relatively small and homogeneous areas, it is possible to discriminate between the effects of each environmental variable, obtaining a set of indicator species and indices, which are useful for the assessment of environmental quality.

The results of several recent studies in Liguria (N-Italy), dealing with the effects of climate, air pollution, substrate eutrophication and habitat fragmentation on lichen diversity are presented. Sets of lichen indicator species strictly related to particular climatic conditions were used to define homogeneous bioclimatic regions, to ensure comparability of results obtained in different areas. The occurrence of crustose pioneer species and nitrophytic species was used to identify areas with active recolonization and areas subject to nitrogen pollution respectively.

Introduction

Lichen diversity is a complex variable depending on several factors. During recent decades, several methods have been proposed for assessing environmental quality (principally air pollution) on the basis of epiphytic lichen diversity (for recent reviews see Kricke & Loppi 2002). The problems associated with the interpretation of lichen diversity data have also been investigated recently by several authors (Wirth 1995; McCune and others 1997; Loppi and others 2002*a*; *b*). In fact, as biological data depend on several factors other than air pollution, biologists often find it difficult to discriminate between the effects of pollution and those of climate, substratum ecology and other anthropogenic types of interference (Seaward 1995; Nimis and others 2000).

How to separate a pollution signal from a background of other environmental factors is an important and recurring problem in lichen biomonitoring (Brunialti & Giordani 2003). An overview of some approaches used in Liguria (NW-Italy) to interpret lichen diversity will be presented in this paper. Sets of lichen indicator species strictly related to particular climatic conditions were used to define homogeneous bioclimatic regions, to ensure comparability of results obtained in different areas. The occurrence of crustose pioneer species and nitrophytic species was used to identify areas with active recolonization and areas subject to nitrogen pollution respectively.

Materials and methods

Datasets

Analysis was performed on the datasets of two biomonitoring surveys carried out recently in Liguria. In 2000 a biomonitoring survey of Liguria region (NW-Italy) was carried out to detect the effects of atmospheric pollution on epiphytic lichen vegetation (Giordani and others 2002). In 2001 a detailed survey of the metropolitan area of the capital, Genova, was planned on the basis of the results of the previous monitoring.

The guidelines for the Index of Lichen Biodiversity (ILB) method (Nimis 1999) were followed in both cases for the selection of the sampling stations and for the relevé procedures, which are reported in details by Giordani, and others (2002). Sixty-nine sampling stations at the intersections of a 9x9 Km grid, and 44 stations at the intersections of a 3 x 3 km grid were selected for the regional and the metropolitan surveys respectively. Information on the ecological requirement of Ligurian epiphytic lichens was obtained from the Italian floristic database ITALIC (Nimis 2000).

Spot and linear emission sources in Liguria

According to a recent air-monitoring report (Liguria Regional Council 1999), 78% of the total SO_2 emissions in the region are produced by three large thermoelectric power plants (Savona, La Spezia and Genova), which produce about 84,000 tons/year. Other spot sources contribute about 11,400 tons/year, and a further 16,000 tons/year are produced by road traffic and heating systems. The thermoelectric power plants are also the main source of NO_x emissions (55% of the total: about 46,000 tons/year). About 9,200 tons/year (11%) are produced by other smaller plants and 28,000 tons/year (28%) are due to road traffic. Further, agriculture areas in the western part of the region contribute more than 2,000 tons/year of NH_3.

Results and discussion

Lichen indicator species for bioclimatic zones

Most lichens depend on relative humidity for their water so that humid areas have a higher lichen diversity (Rundel 1988). Taking into account the main macroclimatic factors, such as rainfall, humidity and altitude, Nimis (2000) recently created a phytoclimatic map of Italy. According to this approach, there are at least three different bioclimatic regions in Liguria, characterized by humid Mediterranean, humid sub-Mediterranean and dry sub-Mediterranean conditions respectively. The extent to which climatic variables might modify epiphytic lichen communities and hence affect the interpretation of LB scores is particularly important for evaluating the results obtained by lichen biomonitoring networks active recently in Italy and Europe. The distribution of some lichen species which are strictly limited to areas with humid climate probably corresponds to the boundaries between the three bioclimatic regions (Brunialti & Giordani 2003). As an example, the distribution of *Parmotrema chinense* in Liguria is shown in Figure 1, defining the humid sub-mediterranean region.

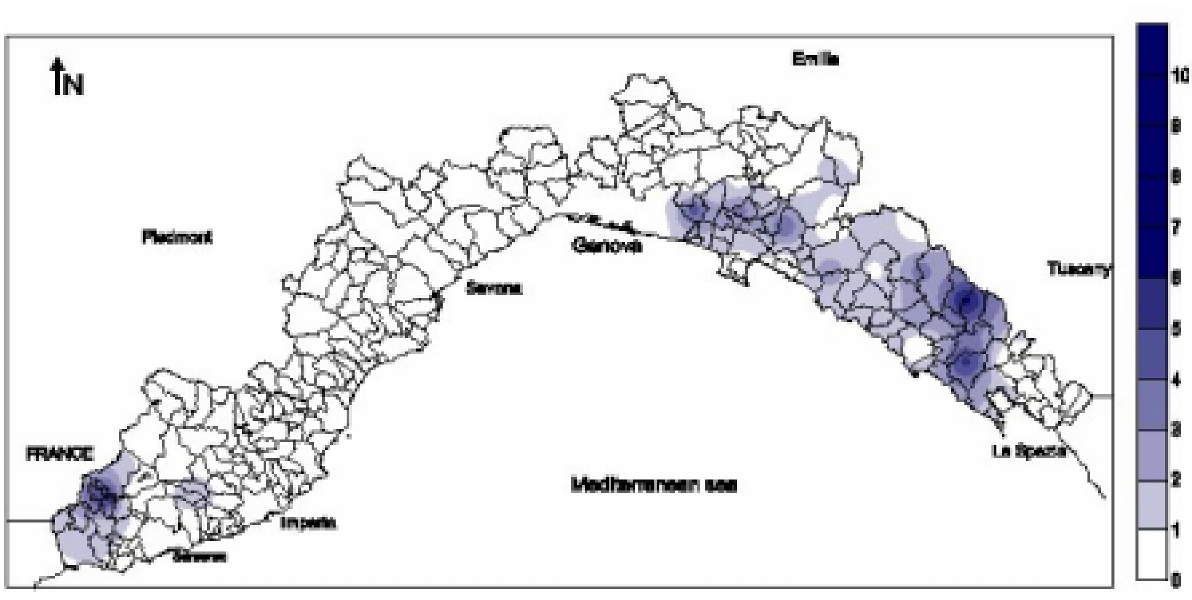

Figure 1. Distribution of *Parmotrema chinense* in Liguria showing frequency within the sampling grid.

Lichen indicator species for eutrophication: applicability in Tyrrhenian side of Italy

Many authors (van Dobben & de Bakker 1996, van Herk 1999, 2001) have recently observed an increase of nitrophilous species in western Europe, as a consequence of a decreasing of SO_2 and of high levels of nitrogen compounds (namely NH_3 and NO_x), due to intensively cattle husbandry and road traffic.

Ecological indicator values for eutrophication (Nimis 2000) are used to detect possible ecological gradients for this parameter in an area. The ecological index specifies, for each lichen species, a range on a 5-class ordinal scale, from ‘no eutrophication’ (1) to ‘very high eutrophication’ (5). In Figure 2 a comparison between the distribution of the values of the eutrophication index of all the Ligurian epiphytic species and of the epiphytic lichens in the industrial area of Savona are shown. Evident differences may be due to hypertrophic conditions of the substrata in the latter area.

Van Herk (1999) proposed the use of a list of strictly nitrophilous species (NIW) to monitor the effects of ammonia pollution.

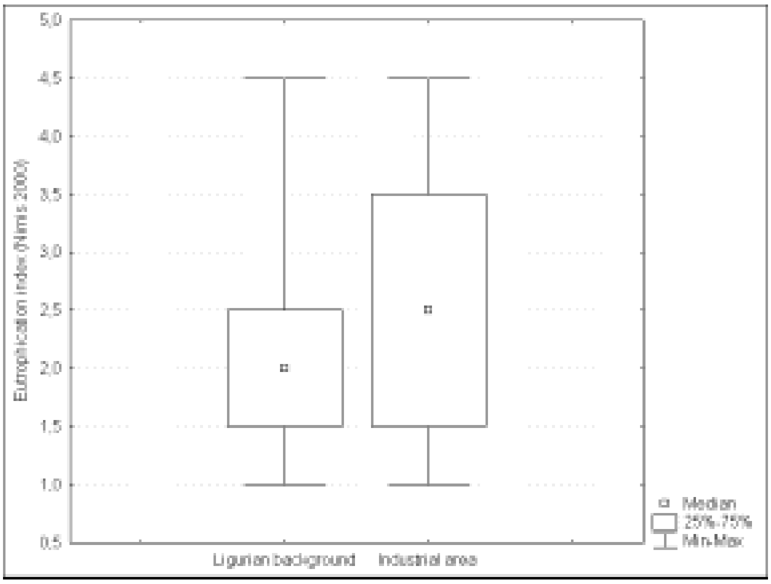

Figure 2. Differences between median N requirement (according to Nimis 2000) in the Ligurian epiphytic lichens and in the species collected in an industrial area of the Savona province.

To extend this kind of approach in other bioclimatic areas and to allow direct comparison of the results, it is essential to develop appropriate lists of nitrophilous species that take into account the natural background conditions and lichen flora of each region, as proposed for the interpretation of bioindicators (Loppi and others 2002a, b). In Table 1 a list of nitrophytic species for Liguria is presented obtained on the basis of the ITALIC database (Nimis 2000).

Table 1: List of nitrophytes for Liguria. Only the species with eutrophication index > 3 (Nimis 2000) have been taken into account. Species with broad ecological requirement have been excluded.

Eutrophication index 4-5
Caloplaca cerina
Candelaria concolor
Candelariella reflexa
Hyperphyscia adglutinata

Index 3 - 5
Phaeophyscia chloantha
Phaeophyscia hirsuta
Phaeophyscia orbicularis

Index 3 to 4
Physcia adscendens
Caloplaca herbidella
Caloplaca luteoalba
Caloplaca ulcerosa
Diplotomma alboatrum
Lecanora umbrina
Phaeophyscia cemohorskyi
Physcia aipolia
Physcia biziana v. *biziana*
Physcia dubia
Physcia tenella
Physcia dimidiata
Physconia distorta
Physconia grisea subsp. *grisea*
Physconia enteroxantha
Rinodina colobina
Xanthoria fallax
Xanthoria fulva
Xanthoria parietina

It is noteworthy that *Xanthoria parietina*, a common species with a worldwide distribution is quite rare on subacid substrata in the humid areas of Liguria (Aptroot, this volume), as are other species of the *Xanthorion* communities. (The *Parmelion* being the predominant vegetation). For this reason, changes in the frequency of this species may in future be a valid tool for monitoring ammonia pollution in Italian Tyrrhenian regions.

On the other hand, in dry conditions and on basic substrata nitrophytes are the 'natural potential vegetation' (Figure 3). In this case, it is difficult to interpret whether high frequencies of these species are due to ammonia pollution. Furthermore, these species are not only nitrophytic, but also 'eliophytic' (requiring high light intensity) thus it is not always possible to distinguish which ecological factor most affects their distribution.

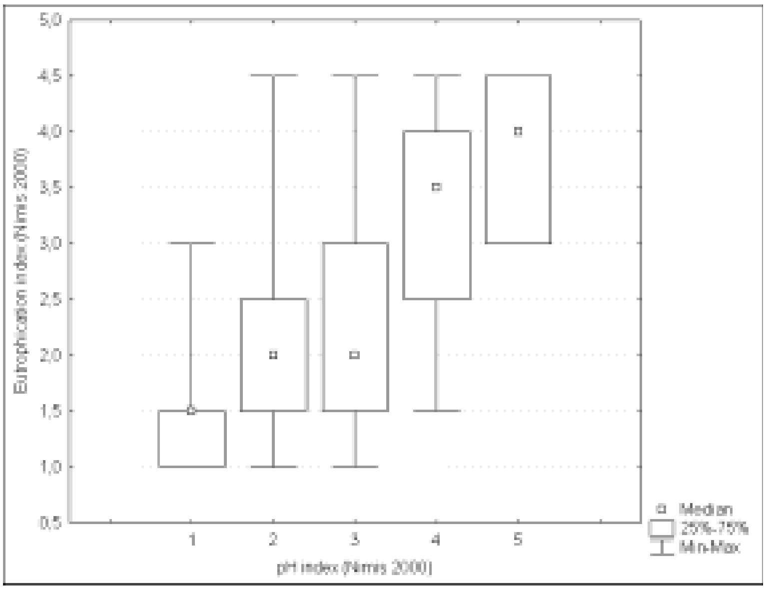

Figure 3. Relationship between eutrophication index and pH index of the substratum (Nimis 2000) in the Ligurian epiphytic lichens.

Genova: a case study.

Maps of the Index of Lichen Biodiversity (ILB) (a), the percentage frequency of crustose species (b) and the percentage frequency of nitrophytes in the relevés (c) in the town of Genova are shown in Figure 4. As a consequence of a decreasing SO_2 level, lichen diversity is increasing in the survey area, where the former lichen desert was restricted to the industrial zone in the western part of the city. The high frequency of crustose lichens, representing the pioneering phase of colonization on trees may indicate a significant improvement of air quality. On the other hand, both the concentrations of nitrogen compounds and particulate matter still remain on high and very high levels and are probably indicated by the map in (c). In this case, the distribution of lichens is the result of complex responses to the main variables involved, so that lichen diversity no longer represents a single pollution gradient. Further work is necessary to establish the response of lichen species to the separate variables involved.

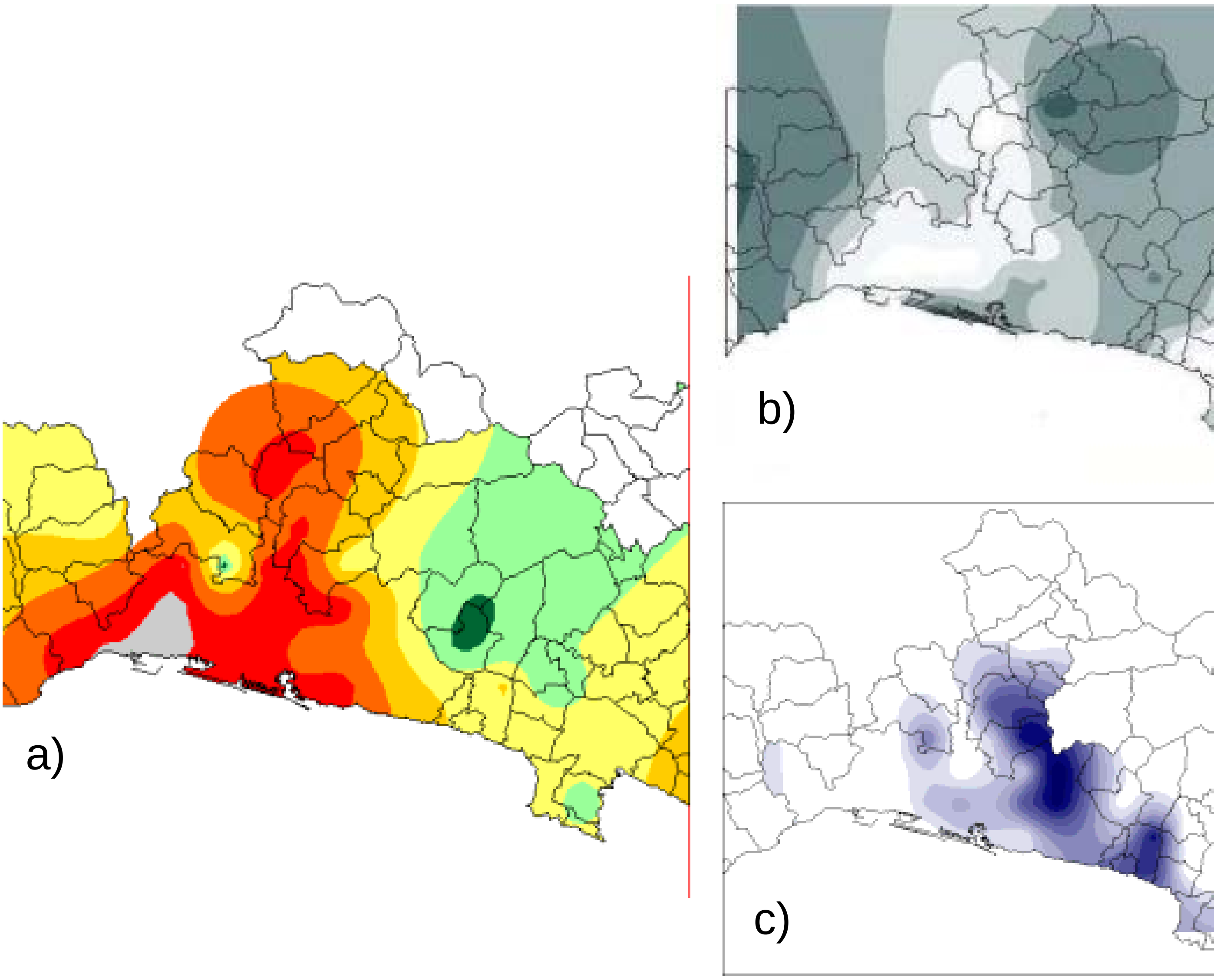

Figure 4. Maps of 50km section of Genova town and adjacent regions in Liguria (NW Italy) showing a) Index of Liche
light green = 51-60; light yellow = 41-50; deep yellow = 26-40; orange = 11-25; red = 1-10; grey = 0, b) percentage fre
percentage frequency of nitrophytes in the relevés.

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Responses of *Cladonia portentosa* growing on an ombrotrophic bog, Whim Moss, to a range of atmospheric ammonia concentrations.

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Summary

This paper describes a novel field fumigation system for treating ombrotrophic bog vegetation with ammonia. Early effects, visible damage, bleaching, the spread of *Pleurococcoid* algae and depression of chlorophyll fluorescence (Fv/Fm) are described for *Cladonia portentosa* in response to different levels of NH_3 exposure and issues of interest are flagged.

Introduction

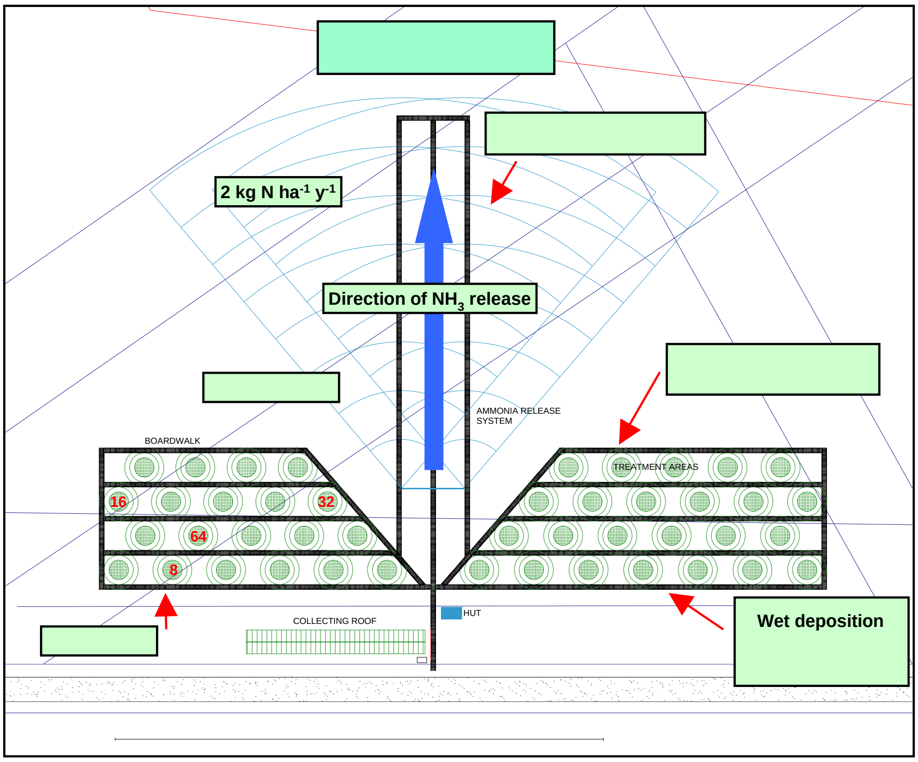
Lichen distribution is strongly influenced by atmospheric pollution and species presence, absence and vitality are used to map air pollutant concentrations (van Herk this volume). Nitrophytic epiphytes that grow on neutral or eutrophicated bark are common on trees surrounding large intensive animal units which are sources of reduced N. It is generally accepted that the presence of these epiphytes reflects the neutralising effect of ammonia (NH_3) on bark pH. But what are the direct effects of NH_3 on the performance of other lichen groups under field conditions and do environmental factors condition responses?

As part of a large scale field experiment to determine the impact of different N forms on ombrotrophic bog vegetation, CEH Edinburgh are treating mat forming lichens eg *Cladonia portentosa* with NH_3 and wet NH_4^+ and NO_3^- *in situ*. The facility is unique in coupling realistic treatments with natural variation in deposition scenarios, *.ie.* wet treatments track rainfall events and gas exposure is dependent on wind direction

Site, materials and methods

Whim Moss represents a transition between a lowland raised bog and a blanket bog, 280 m asl. receiving a mean annual rainfall of 900 mm. The Moss is located 10 km south of CEH Edinburgh (NT 206535), but most importantly has low background N and S inputs, < 8 kg N, and 4 kg S ha^{-1} y^{-1}. The vegetation is unmanaged, NVC M19a – (Rodwell 1991). The surface undulates gently over 3 to 6 m of acid peat, pH 3.3-3.9 (H_2O). *C. portentosa* grows in the open or under degenerate *Calluna* branches. The experimental plots which cover almost 0.5 ha are accessed using a 1 km network of boardwalks (Figure 1).

The automated ammonia release system is controlled by a Campbell 23X data logger which records timing, duration and environmental conditions prevailing during NH_3 release. The system is activated when windspeed exceeds 2.5 m s^{-1} and direction (sonic anemometer) is in the sector 180 to 215^0. Gaseous NH_3, from a pressurised anhydrous NH_3 cylinder, flows *via* a mass flow controller through a stainless steel pipe at 3.3 g min^{-1} into a fan unit where it is mixed with air. The diluted NH_3 is released at one metre from the ground through a series of 6 mm holes evenly distributed along and around a 10 m pipe. NH_3 concentrations are quantified in the downwind sector along a transect at 1, 2, 4, 6, 8, 12, 16, 32, 60 m and an upwind control position. A combination of diffusion tubes and ALPHA (Adapted Low-cost Passive High Absorption) samplers (see Sutton this volume) were installed at 2 heights (0.1 and 0.5 m above the vegetation), These are left in position for one month periods, exchanged for regenerated badges and the NH_3 concentration measured (Tang and others 2001). These samplers are impregnated with citric acid to absorb the NH_3 and provide the mean concentration (amount absorbed/exposure period) for each month. These monthly concentration means can be significantly modified by wind speed and closeness to the ground vegetation. Peak NH_3 concentrations are not quantified by this method.



Visual records of the lichen appearance are made routinely, together with digitised photographic records of marked plots. Visible damage was quantified by counting numbers of damaged and healthy *C. portentosa* clumps in February 2003. Measurements of, chlorophyll fluorescence (Fv/Fm) were made monthly for 3 months after visual damage (bleaching) was first observed in November 2002. Twenty measurements per position were made on **wet** apices using a Hansatech HANDY PEA, dark adapting 15 minutes, and using an actinic light intensity of 2500 mmol s^{-1}. Despite being a non-invasive technique, the actual operation of attaching the dark adaptation clips could physically damage the apices restricting the number of repeat measurements.

The experiment is funded to the end of March 2004. Additional funding is being sought to maintain the site and treatments for at least another three years. The longer the experiment runs the greater the opportunity for testing whether relatively small N inputs (8-16 kg N $ha^{-1}yr^{-1}$, ie ambient in many areas) are affecting biodiversity and ecosystem function.

Results and discussion

Table 1 Mean hours of ammonia release per month

Summer		Winter	
Month	**NH_3 fumigation 2002**	**Month**	**NH_3 fumigation 2002/3**
May	93	October	12
June	97	November	75
July	19	December	46
August	34	January	39
September	35	February	50

The ammonia release system began at the end of April 2002 and has operated continuously since then, with a downtime of <2 %. Release by the very nature of wind direction has been intermittent, and monthly release hours have varied up to eight-fold between 12 and 97 hours per month (Table 1). The unpredictable nature of the release provides a very realistic simulation of the situation downwind of an intensive animal unit but confounds our ability to identify the drivers of lichen responses to NH_3. For instance in June following a period of sustained release, (averaging 4hrs for 35 days) visible damage (pink apices) were observed. A similar phenomenon was observed in a separate open-top chamber experiment (Leith and others 2002) where *C. portentosa,* taken from the same bog had been receiving 90 µg m^{-3} NH_3 continuously. Neither concentration nor hours of exposure appear to explain the phenomenon but the occurrence coincided with a week of hot sunny days. A return to the cooler overcast conditions that prevailed that summer saw the effect disappear, not to return. This suggests the pink colouring, which is common on epiphytes growing near local NH_3 sources, may be related to the impact of such conditions on the ability of the phycobiont to assimilate and detoxify the NH_3. This has implications for Critical Load exceedance as it suggests that under hot sunny conditions lichens may be more at risk from gaseous pollutants if there is sufficient moisture to facilitate deposition and keep the photosynthetic machinery partially active.

Table 2 Mean of the monthly NH_3 concentrations (µg m^{-3}) at 0.1 m above the vegetation to the end of January for each distance. Fv/Fm values for bleached (B) and none bleached (U) apices of *C. portentosa* measured along the transect and the proportion of damaged clumps in each category as of February 2003. (SE on the Fv/Fm values=<0.03, SD =<0.14 irrespective of the month).

Distance in m	NH_3 µg m^{-3}		Dec Fv/Fm	Jan Fv/Fm	Feb Fv/Fm	% damaged clumps
1	42.7	B	0.29	0.56	0.33	86
		U	0.65	0.68	0.58	
2	97.1	B	0.49	0.48	0.29	100
		U	0.58			
4	92.4	B	0.34	0.44	0.19	100
		U				
6	94.2	B	0.34	0.42	0.17	100
		U				
8m	101.9	B	0.39	0.52	0.37	67

Distance in m	NH_3 µg m^{-3}		Dec Fv/Fm	Jan Fv/Fm	Feb Fv/Fm	% damaged clumps
		U	0.57	0.54	0.37	
12m	68.3	B	0.67	0.53	0.33	100
		U	0.61	0.50	0.57	
16m	46.7	B		0.54		63
		U	0.50	0.64	0.53	
32m	17.2	B				0
		U	0.45	0.62	0.55	
60m	5.2	B				0
		U	0.73	0.69	0.59	
Ambient	0.3	U	0.67	0.63	0.63	0

In November 2002 a different damage symptom, bleaching, was observed between 1 and 8 metres from the NH_3 source. In addition the bases of some apices were dark green due to colonisation by *Pleurococcoid* algae. These damage symptoms occurred on clumps that had been exposed to high NH_3 concentrations, > 40µg m^{-3}, typical of those found close to large intensive animal units (Fowler and others 1998). The bleaching continues, suggesting that mat forming lichens would be highly sensitive to the positioning of new large intensive animal units. By the end of February the apices were totally white.

The damage score represents a cumulative treatment effect but the mean NH_3 concentration is an average (May to January) with no information about peak concentrations. The apparent discrepancy between the level of damage and mean concentration may reflect a combination of peak concentrations and dose. We are in the process of calculating an NH_3 deposition velocity and canopy resistance (Flechard & Fowler 1998) for this type of vegetation in order to calculate the N dose concomitant with the onset of bleaching. Not all the clumps of lichen at any one position were equally affected (Table 2). The reason for this difference in sensitivity is not clear; it may be genetic or it may be environmental. Moisture differences influence NH_3 deposition and the ability to detoxify the NH_3 and an ameliorating effect from neighbouring vegetation is possible. Differences in peak concentrations may occur as a result of differential mixing of the NH_3 in the atmosphere but only continuous monitoring will resolve this. The progression of damage appeared to speed up in February after several days of subzero temperatures. However, it is difficult to separate the influence of temperature from the accumulating dose and exposure concentrations. If we are to understand mechanisms and set sensible Critical Levels and Loads we will need to test the potential interactions of moisture / NH_3 concentration / accumulated dose and temperature under controlled conditions.

Fluorescence measurements of photosystem II Fv/Fm, showed a reduced ratio from approximately 0.65 to 0.35 (Table 2) in bleached apices, so that for December and February Fv/Fm close to the source was < 50% of ambient. The January values for bleached apices indicated some recovery. Uptake of NH_3 may have been restricted by snow, which covered the lichens until shortly before measurement. If this were so it suggests the lichen was still capable of recovery under conditions of reduced NH_3 exposure.

These results clearly demonstrate the sensitivity of mat forming lichens to NH_3 and we will be following the lichens growing at around 10 µg m^{-3} NH_3 to see if and when these develop similar symptoms. The responses of *Cladonia portentosa* to intermittent NH_3 exposure imply a significant interactive, modifying role for climate. The results also suggest the effects of NH_3 are mediated through both cumulated dose and peak concentrations.

Implications

The increasing amount of damage away from the source with time indicates these lichens are sensitive to N dose as well as peak concentrations. Detrimental effects at background levels may be difficult to detect in the short-term.

Further work

1. Studies are required to validate whether the bleaching observed is specific to ammonia or just a general stress response. This is important if the symptom is to be evaluated as a biomonitor for NH_3.
2. Define the relationship between NH_3 concentration and N deposition for lichen species.

Acknowledgements

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Using lichen communities to assess changes in sites of known ammonia concentrations

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Summary

Lichen data was collected on *Quercus* trunks and twigs from trees in 2 climatically different sites in Norfolk and Devon, where ammonia recording stations were situated, together with bark pH of both substrates. Using DAFOR frequency values of 1-5, the contribution of species to each lichen community at the alliance level was calculated for trunks and twigs at each station. The results showed that increases in the Xanthorion alliance were correlated with increasing NH_3 concentration and bark pH and with replacement of lichen communities including the Usneion and Parmelion. This was more marked on the newly available surfaces of twigs than on trunks where relict communities still occurred.

Background to pollution monitoring

The traditional view of lichens as sensitive to pollution was highlighted in the 20th century across Europe in regions of industrial and domestic pollution due mainly to SO_2. In Britain sensitive and tolerant species were selected to map pollution levels (Hawksworth and Rose 1970), and on the continent lichen diversity was used in unit areas (IAP, VDI (Kricke and Loppi 2002)). Falling SO_2 levels across Europe led to recovery of lichens in urban areas where previous levels of SO_2 had eliminated lichens or allowed the few SO_2 tolerant species to increase their distribution. Species recovery did not follow the expected pattern (Hawksworth & McManus 1989, Gilbert 1992). An increase in species characteristic of high nutrients became apparent in both rural and urban areas accompanied by a loss of sensitive acidophyte species such as *Usnea*. In this situation it was no longer possible to use lichen diversity or simple indicator species to indicate an increase in pollution, partly because it was apparent that some lichens were encouraged by increased N and others were depressed. In order to determine the effect of one aspect of N pollution, ammonia, in the vicinity of pig farming units in Holland van Herk (1999) used frequency of species on *Quercus* trunks in the vicinity of ammonia monitoring stations. Analysis of the results from 100 sites of 10 trees detected a number of indicator species that were highly correlated with NH_3 concentrations and a number of acidophytes which were highly sensitive to increased NH_3.

During a short survey by the Natural History Museum for DEFRA (Wolseley & James 2002) two of the authors were asked to review the van Herk methods and test their application in the UK climate in sites where NH_3 concentrations were known. Two sites were chosen across the E-W climatic gradient at Thetford in Norfolk (0∀49'E, 52∀27'N) in a continental climate where rainfall is c. 500 mm a year and at North Wyke, Devon (3∀54'W, 50∀27'N) with an oceanic climate where rainfall is c. 1000 mm a year.

Species were recorded on trunks and twigs (Wolseley & Pryor 1999) of *Quercus* spp. in stations adjacent to NH_3 monitoring sites on trunks using a cover-abundance scale (van Herk

1999) and on twigs using a DAFOR frequency scale of 1-5 (D = dominant, A = abundant, F = frequent, O = occasional, R = rare). Species identified as nitrophytes or acidophytes by van Herk were used to calculate indices of nitrophytes (NIW) and acidophytes (AIW) (van Herk 1999). Results showed that indicator species in Holland differed from those in England apart from a few widespread species that contributed to the NIW and AIW, due to climatic and historical differences. In addition the strong climatic gradient and differences in former pollution levels at Thetford and North Wyke showed very different results on the trunks, resulting in higher diversity on trunks at North Wyke than at Thetford, while the twig flora was more similar (Wolseley & James 2002a & b).

However the interesting difference between the 2 sites was the contribution made by species falling outside the nitrophyte or acidophyte categories defined by van Herk. Both sites had ancient trees that showed continuity of land use but the lichen flora was very different, and in particular North Wyke included small populations of rare and/or endangered species such as *Teloschistes flavicans, Lobaria pulmonaria, Opegrapha lyncea (*Wolseley & James 2002 a & b). At Thetford, sites with low levels of ammonia had greater cover of foliose species of the Parmelion with species that were not included in the AIW scale. The classification of lichen communities into associations and alliances allows the detection of shifts from one community to another irrespective of the geographical distribution of their component species. Epiphytic communities of lichens were described by James and others (1977) and factors affecting epiphytic diversity described in the same paper. This paper constructs changes in the lichen communities at alliance level in stations across an ammonia and land use gradient in both sites. The data is based on frequency of species components of the epiphytic alliances on *Quercus* trunks and twigs at each site

At North Wyke, NH_3 concentrations have been measured on a monthly basis since 1996 at one site upwind of an experimental livestock farm of the Institute of Grassland and Environmental Research, as part of the UK National Ammonia Monitoring Network, using the DELTA active sampling denuder system (Sutton and others 2001). In order to estimate NH_3 concentrations at different locations around the farm, the SCAIL model (Simple Calculation of Ammonia Impact Limits) (Theobald and Sutton 2002) was applied. This is a simple empirical model of NH_3 dispersion, which used information on the measured concentration from the NAMN site (1.7 µg m^{-3}), plus information on the location of lichen sampling points to different NH_3 sources at the farm to generate the site specific local enhancement of NH_3 concentrations. In order to avoid bias in the estimates, the contribution of the nearby farm to the NH_3 monitoring data (1.0 µg $m-^{3}$) was subtracted from the measured value to estimate a landscape background NH_3 concentration of 0.7 µg m^{-3}. The estimated overall NH_3 concentrations for the sites at North Wyke ranged between 0.89 and 2.64 µg m^{-3}. At Thetford, measured NH_3 concentrations were available directly from monthly measurements at 25 sites as part of the GANE LANAS project for a period of 9 months. The measurements were made with ALPHA high-sensitivity passive samplers (Tang and others 2001).

Results

Background information on stations, NH_3 concentrations, lichen diversity and bark pH at North Wyke and for Thetford are shown in Table 1

Table 1

North Wyke

Station no.	Modelled NH_3	pH trunk	pH twig	Trunk div	Twig div.
1	4.47	4.9	5.3	18	4.45
2	4.71	5.1	5.9	18.2	5
3	3.39	4.9	5.5	15.7	9.3
4	3.54	4.9	5.4	20.7	11.5
5	2.96	4.25	-	18.6	12.3
6	3.97	-	-	16.5	-
7	3.02	4.6	5.9	18	11.2

Thetford

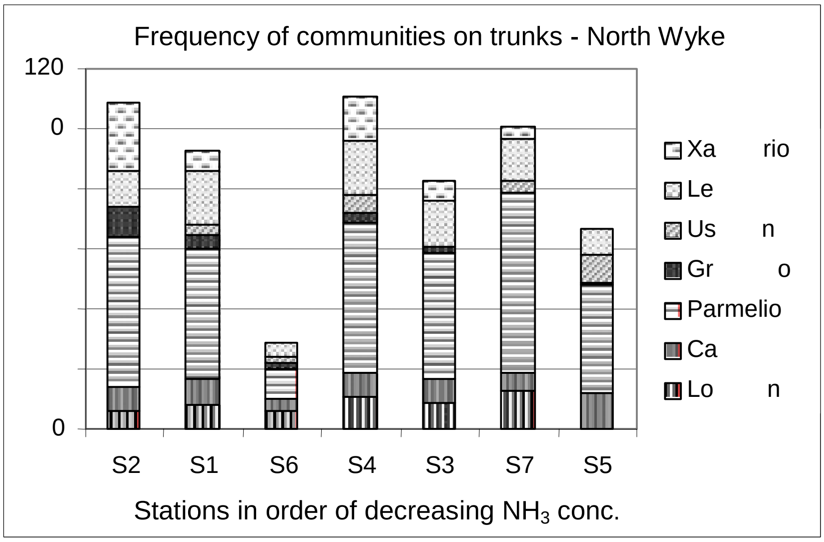
Station no.	NH_3 $\mu g/m^3$	pH trunk	pH twig	Trunk div.	Twig div.
1	2.88	4.1	5.1	7.4	5
2	2.9	6.3	5.7	10	2.7
3	1.9	4.2	5.3	8.5	6.6
4	2.2	3.5	5.3	7.7	7.7
6	2.8	5.1	6.4	7	3
8	9.2	5.7	6.2	5.3	0.5
9	4.54	5.1	6.4	10.3	2
10	2.6	4.4	5.4	7.2	3.5
11	1.4	4	5.5	10	6.1
12	2.05	3.9	5.1	7.3	8

Lichen frequency on twigs at North Wyke is lowest in the vicinity of the farmyard and slurry tank and highest at the station on the edge of *Molinia* moor where ammonia concentrations were lowest. Lichen frequency on trunks showed little correspondence to ammonia levels, frequency being high in all stations except site 6, a single tree where twigs were not accessible. This tree also supports rare species of the Lobarion and the Calicion at low frequency (Wolseley & James 2002).

At Thetford lichen frequency on twigs follows a similar pattern with low levels in the vicinity of high ammonia levels.. Exceptions occurred at stations 10-12. Diversity and frequency on trunks shows a slightly better correlation to ammonia levels than for North Wyke.

Distribution of lichen communities on twigs and trunks in 7 stations at North Wyke is shown in figure 1 and 10 stations at Thetford in Figure 2.

1a.



1b.

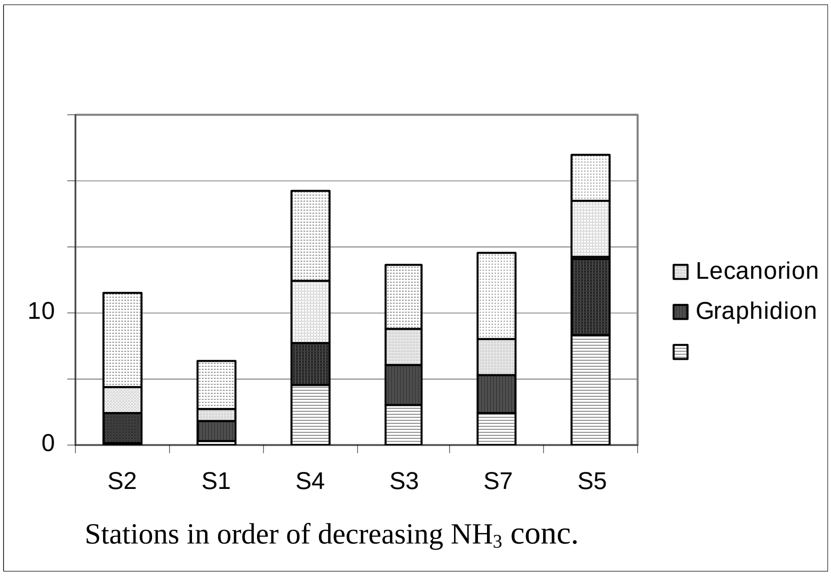
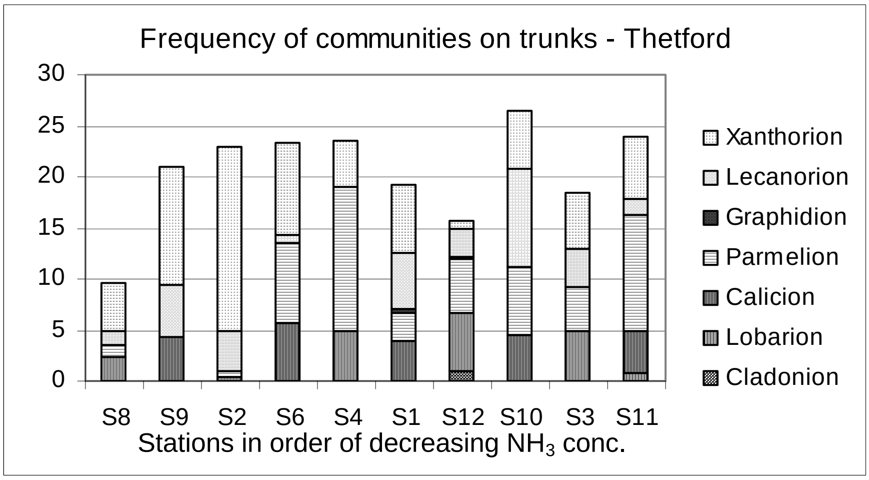


Figure 1. Lichen communities sampled on *Quercus* trunks (a) and twigs (b) at stations in the vicinity of North Wyke, Devon, arranged by modelled NH_3 concentration. Station numbers correspond with those in Table 1.

2a.



2b.

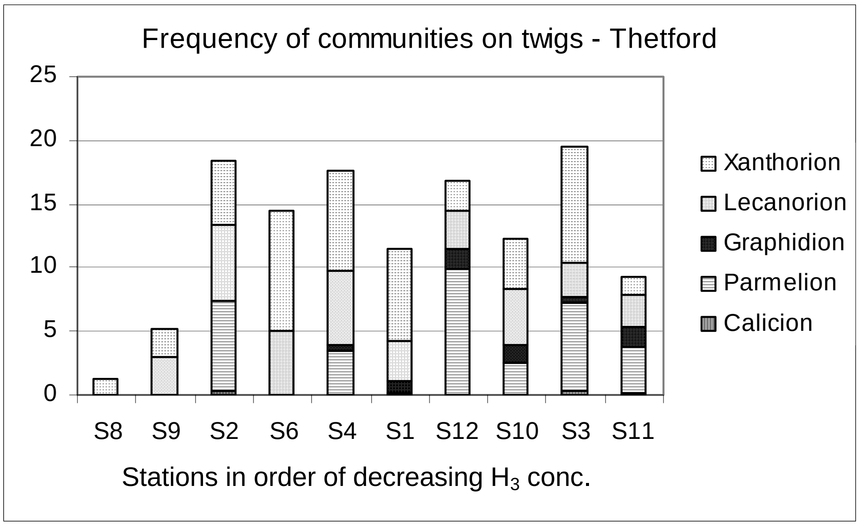


Figure 2. Lichen communities occurring at different stations on *Quercus* trunks (a) and twigs (b) in the Thetford area, Norfolk, arranged according to NH_3 concentration. Station numbers correspond with those in Table 1.

Both histograms demonstrate changes in the lichen communites . At North Wyke species of the Xanthorion are present on all twigs, lowest frequency being recorded at site 5 and highest frequency at station 2 in the vicinity of the slurry tank and farmyard. In these sites the Parmelion is either absent or rare, and increases in frequency with distance from the farmyard and estimated lower levels of ammonia.

The absence of the Usneion on twigs in all sites contrasts with its presence on trunks in all stations except that adjacent to the farmyard and slurry tank. The Xanthorion is absent from trunks at the cleanest stations with NH_3 concentration <1µg/m3 but present at low frequency, except adjacent to the slurry tank. The importance of the Parmelion in its contribution to the total flora of the trunks is shown in all stations except 6 where the Lobarion, a bryo-lichen community supported both *Lobaria* and *Teloschistes* at low frequency. Although the

Lobarion and Calicion were represented at other stations the rarer species were not found. At North Wyke the Lecanorion and the Graphidion are represented in all stations.

At Thetford the dominance of the Xanthorion at all stations on twigs was apparent although at high ammonia concentrations frequency was low and other communities absent (S8) or at low frequency (S9). The Parmelion was present at all stations on the trunks but on the twigs it was absent from stations where NH_3 concentrations were > 3 µg/m3 and bark pH >5.7. In both sites bark pH of twigs was higher than that of trunks, but on the trunks the stations with higher frequency of the Parmelion were associated with NH_3 < 3 µg m-3 and bark pH < 5.2. The Xanthorion is present on the trunks at all stations and the Graphidion present at low frequency in all but the most polluted stations. The Calicion is present at low diversity in most stations consisting mainly of *Lepraria* species which are pollution-tolerant acidophytes, the most conspicuous absences being the Usneion, and the Graphidion is only present in stations 1 and 12 at low frequency where pH values were 4.1 and 3.9 and NH_3 concentrations <2.08 and 2.05 µg m^{-3} respectively.

Discussion

Epiphytic communities of lichens contain species that occur in specific habitats and conditions (James and others 1977). Components of these communities may shift by geographic region and may also contain species tolerant of and sensitive to pollution. Shifts from one epiphytic community to another may be part of a normal succession that occurs on substrates of different ages, or part of a response to changes in conditions, as is occurring in both urban and rural areas across Europe. An important factor in the assessment of shifts in condition is the recognition of characteristic species of each community and of species that belong in other communities. The shift in alliances is associated with either environmental conditions or with succession and ecological continuity, while species composition is best described at the association level (Fig 3.). For example the Xanthorion is associated with high nutrients but may be found in a species-rich association the *Teloschistetum flavicantis* and a species-poor the *Physcietum ascendentis*. Both communities were represented at North Wyke.

Although pollution from SO_2 resulted in a lowering of epiphytic diversity the effects of N pollution in all its forms may be different due to low levels of nitrogen being a growth stimulant. One of the effects of ammonia is to cause an increase in bark pH. This is especially obvious on acid-barked trees where loss of acidophyte species of the Usneion and the Parmelion is associated with an increase in nitrophytes of the Xanthorion. This community, although rapidly invasive on new substrates, cannot rapidly invade an already closed community such as occurs on the trunks of old *Quercus* at North Wyke, whereas at Thetford in a more continental climate that also was affected by former SO_2 pollution the Xanthorion has almost replaced the Parmelion in stations 2, 8, 9 with NH_3 concentrations > 2.8 and bark pH >5.5.

The importance of bark pH in effecting changes in lichen communities cannot be overestimated, although as yet we do not understand the mechanism by which this works in combination with nitrogen products (van Herk 2001, and this volume). The high pH values of the twigs at both sites suggests that this may affect the succession of communities over time.

Conclusions

Recording species on trunks and twigs using a simple frequency scale allows both the detection of shifts in communities, and more particularly the monitoring of rare or local components of that community. The loss of acidophyte species of the Usneion and Parmelion is directly correlated with an increase in the Xanthorion, and this shift has taken place already on the twigs and is not yet apparent on the trunks at North Wyke, whereas at Thetford the Xanthorion is frequent on trunks and twigs. In this situation the twigs provide an early warning system about changes which are slower to affect established communities on trunks.

The recording of all communities present allows the recorder to check that all components are present. In both sites the most polluted stations have a reduced number of communities present, and reduced species within the communities (other paper). Monitoring of shifts in species components will allow correlation with a range of environmental factors including pollutants, climate change and management.

The loss of rare or local species is directly correlated with the loss of the community that supports these species. Although climatic conditions at North Wyke are sufficient to support *Lobaria* and *Teloschistes*, these associations are species-poor and restricted to one hedgerow tree at station 6. The presence of components on other trees at this site suggests that these species were more widespread in the past. Although there is a general increase in the Xanthorion on trunks at North Wyke the absence of species of the Xanthorion at station 6 suggests that changes at this station may be due to other environmental factors than nitrogen.

The classification of species in communities allows species replacement in different geographical regions so that northern or western species may replace components of the same community. This has an obvious advantage for the development of practical methods of assessing NH_3 impacts.

Age substrate (yrs)	Industrial pollution (SO_2 and related) pH <3.5	Upland, high rainfall pH 3.5-4.5	Ecological continuity unpolluted air pH 4.5-5.5	Unimproved pasture and parkland pH 5-6	Intensive agriculture pH 6+
1-10	**Lecanorion** *Bacidietum chlorococcae* ↓	**Graphidion** *Arthopyrenietum punctiformis* ↓	**Graphidion** *Arthopyrenietum punctiformis* ↓	**Graphidion** *Arthopyrenietum punctiformis* ↓	**Lecanorion** *Bacidietum chlorococcae* ↓
5-20	Species poor **Pseudevernion** + *Parmeliopsis ambigua* ↓ ↑	**Pseudevernion fufuraceae** ↓	**Usneion barbatae** ↓	**Lecanorion** *Lecanoretum subfuscae* ↓ ↑	**Xanthorion** *Bullietum punctiformis* ↓ ↑
15-50 (-100)	**Lecanorion** *Lecanoretum pityreae* + *Lecanora conizaeoides* ↓	**Parmelion laevigatae** ↓	**Parmelion perlatae** ↓	**Xanthorion** *Parm. elegantulae* or *Teloschistetum flavicantis* ↓	**Xanthorion** *Physcietum ascendentis* ↓
100 +	**Calicion** *Leprarietum incanae*	**Cladonion coniocreae**	**Lobarion pulmonariae**	**Lobarion** well-lit facies + *L. amplissima*	**Cladonion coniocreae** *Bacidia delicata*

Figure 3. Succession on exposed side of trees in clean and polluted environments associated with bark pH (following James and others 1977) Alliance shown in bold figures, association and characteristic species components in italics.

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Session 3 Discussion

Selecting and monitoring species and communities

Participants: Christoph Scheidegger (chair), Andre Aptroot, Damien Cuny, Chris Ellis, Isabelle Franzen-Reuter, Paulo Giordani, Peter James, Chantal van Haluwyn, Pat Wolseley (raconteur)

Questions raised in session 3:

- How to accommodate stakeholders interests.
- Should we be monitoring species or communities?
- What methods should we be using to monitor changes?
- What are the constraints?
- How to monitor the effects of N on protected sites.
- How to raise awareness of these issues

The group established the necessity of designing methods with appropriate resolution for stakeholders interests and for validating the results.

Species or communities?

Community-based approach inherently useful across larger areas where shifts in species components may occur or species may become locally rare, but urgent need to update phyto-sociological studies of lichens in order to detect shifts between communities due to changes in environmental conditions.

Considered that protection is needed at the community level in order to protect component species of conservation interest. Also important to monitor communities in order to detect changes in frequency and distribution that may be occurring as a result of changes in environmental parameters.

Should methodology be open or targetted? If all species recorded as in van Herk (same vol.) tree survey work in Holland, the whole data set is available for interpretation. However this is very expensive and time consuming and produces much redundant data. However it allows a return to the site data in order to ask other questions. Targetted methodology may not address the problem of disappearing populations. More work needed on lichen ecophysiology in response to stress.

Constraints

- Not enough experts available.
- In order to address the problem of threatened species we have to agree on a minimal set of methods to obtain RAPID biodiversity assessments.
- In areas where identification is difficult it may be necessary to use morphospecies as a proxy in order to assess lichen diversity.

Consideration of N impacts on protected sites.

The group recognised:

a) the need to establish buffer zones around protected sites;

b) the preparation of habitat and site inventories;

c) the requirement to establish and maintain favourable conditions for noteable species in sites of conservation importance.

Raising awareness

Difficulty of raising awareness for minor groups like lichens, so a very targetted approach was needed. Co-operation with other specialists and integration with other projects is very important in order to raise the profile of lichens. For instance to co-operate towards the VIth European Framework.

The urban biodiversity message was producing interesting results which could be exploited by creative PR. Most of all needed to engage the media with the subject and the message!

5. Conserving lichen communities and species diversity

Wayside trees: their conservation importance

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Introduction

Until relatively recently the importance of wayside trees for epiphytic lichens has been overlooked, possibly because they are often found in areas not visited by lichenologists. A recent review of the conservation importance and management of lichens on wayside trees is given by Coppins in Fletcher (2001).

Wayside trees are found widely in the rural landscape, along trackways, along roadsides, in fields and in hedgerows. Many are relicts of 18th and 19th Century landscaped parklands and may survive as exotic plantings such as tree avenues. However, they are often found outside of protected sites such as Sites of Special Scientific Interest, and unlike their urban counterparts are not usually covered by Tree Preservation Orders.

A surprising number of rare lichens are associated with wayside trees. Those listed as Critically Endangered, Endangered or Vulnerable in the British Lichen Society's Conservation Evaluation of British Lichens (Woods and Coppins 2003) are listed in table 1.

Table 1. Red Listed lichen species occurring on wayside trees

Anaptychia ciliaris VU	*Fuscopannaria ignobilis** VU
Bacidia incompta VU	*Leptogium saturninum** VU
Caloplaca flavorubescens EN	*Physcia tribacioides* VU
Caloplaca luteoalba VU	*Parmelina quercina* VU
Caloplaca virescens EN	*Punctelia submontana* CR
Catapyrenium psoromoides CR	*Teloschistes flavicans* VU
Collema fragrans EN	*Thelenella modesta** CR
Cryptolechia carneolutea VU	

VU = vulnerable EN = endangered CR = critical
* Scotland only

Many of the above species are associated with the well-lit trunks of old basic-barked trees with *Fraxinus* and *Acer pseudoplatanus* particularly important. Five species, *Bacidia incompta*, *Caloplaca luteoalba*, *C. virescens*, *Collema fragrans* and *Cryptolechia carneolutea* were formerly more frequent on old *Ulmus*, but have undergone a drastic decline as a result of Dutch Elm disease.

Anecdotal information suggests the epiphytic interest of wayside trees is seriously threatened by a number of factors including:

- The natural loss and lack of replacement of old trees.
- The felling of trees for a variety of reasons including the widening of roads, the loss of hedgerows and increasing use of large farm machinery.
- Atmospheric pollution; including agricultural pollution through artificial fertilisers, sprays and slurry.
- Dutch elm disease.
- Encroachment of ivy on the trunks of trees

Wayside trees in Dorset: a case study

Background

Dorset is a small, largely rural, county situated on the south coast in central southern England. Atmospheric pollution has, until now, only seriously affected the far east of the county. SO_2 pollution remains low at <2.00ppb for most rural areas and 2.1-4.0 ppb for urban areas (DCC 1999), the average SO_2 concentration was in the range 1-2 ppb (Review Group on Acid Rain 1997). During this period, NO_2 levels are 6-8 ppb in east Dorset and 4-6 ppb in the west. However, most ‘acidifying’ deposition is now attributable to dry deposition of ammonia. Ammonia (NH_3) is produced from many local sources, notably slurry lagoons, and is thought still to be increasing whereas NO_2 and SO_2 are decreasing (Hill and Edwards 2003).

Agricultural land occupies approximately 74% of the land surface, occupying between 193,000 and 198,000 ha in the last decade (DCC 2000). Of this 51% is arable land with the remainder grassland or rough grazing. Cattle (30.5%) and sheep (32.6%) account for most of the grazing animals in permanent pastures and short term leys. The livestock farming systems are fairly intensive, but not in the same way as in eastern England or on the Continent. Animals are grazed outdoors much of the year except for dairy cattle that are wintered indoors. The grazing pastures are improved by applications of solid animal manure and by granular fertilisers. However, spreading of liquid slurry is widespread in the winter on fields harvested for maize in the autumn.

Wayside trees in the countryside

Wayside trees are a prominent feature of the agricultural landscape in parts of the county, particularly in the chalk valleys in the centre of the county and in clay vales in the north and west. They are found in a wide variety of situations such as along hedgerows, isolated in pastures, along tracks and roadsides, around farmsteads and in churchyards. Many are relicts of former exotic plantings originating in the 18th and 19th Centuries and take the form of groups or avenues of *Aesculus, Tilia, Acer pseudopolatanus*.

The lichen flora

During vegetation surveys in the west and centre county from 1994-2001 it become clear that a lichen flora of significant conservation importance is present, but not in any way protected.

Table 2. Red Listed and Near Threatened lichen species present in Dorset

*Anaptychia ciliaris** VU	*Physcia tribacioides** VU
Bacidia incompta VU	*Parmelina quercina* VU
Caloplaca flavorubescens EN	*Physcia clementei** NT
Caloplaca virescens EN	*Teloschistes flavicans* VU
*Catapyrenium psoromoides** CR	*Wadeana dendrographa* NT
Cryptolechia carneolutea VU	

VU = vulnerable EN = endangered CR = critical
* populations of national significance

Observations suggest that the best developed flora is associated with well-lit, naturally enriched, old, basic-barked trees. In permanent pastures with extensive grazing regimes the *Parmelietum revolutae* is the dominant community with acidophytes such as *Evernia prunastri* and *Hypogymnia physodes* frequently occurring. Trees in improved pasture with a more intensive grazing regime support the *Physcietum ascendentis* association. While nitrophytes such as *Amandinea punctata*, *Diploicia canescens*, *Phaeophyscia orbicularis* and *Xanthoria parietina* are frequent they rarely dominate. The association as described by James and others (1977) is largely restricted to more enriched sites. More sensitive species such *Anaptychia ciliaris*, *Parmotrema chinense*, *Rinodina roboris* are found in a species-rich community somewhat intermediate between the *Parmelietum* and the *Physcietum*. In some ways it resembles a community described by Rose (1988) from southern Europe which may include *Anaptychia ciliaris*, *Collema* spp., *Physcia* spp., *Physconia* spp. and the mosses *Leptodon smithii* and *Leucodon sciuroides*. This needs further phytosociological work. Whatever the communities involved, observations suggest the lichen flora has developed along with the type of agriculture, but as yet has not been adversely affected by it.

A survey of 67 mature to aged trees by B. Edwards and Vince Giavarini found a total of 133 lichen and 28 bryophyte species to be present.

Table 3. Summary of lichen and bryophytes of the 51 trees surveyed in Dorset

Tree species	Mean girth (m)	Total lichens	Total bryophytes	Average no. epiphytes
Ash	3.11 (1.55-5.50)	101	24	35 (13-54)
Oak	2.78 (1.55-5.15)	93	12	28 (21-41)
Sycamore	2.45 (1.40-3.85)	85	17	28 (19-40)
All trees	**2.88 (1.40-5.50)**	**133**	**28**	**31 (13-54)**

Table 4 Percentage occurrence of forty most recorded epiphytes from the trees surveyed in Dorset

Xanthoria parietina	59
Diploicia canescens	54
Lecanora chlarotera	54
Hyperphyscia adglutinata	51
Pertusaria hymenea	49
Ramalina farinacea	48
Phaeophyscia orbicularis	46
Pertusaria albescens	45
Schismatomma decolorans	42
Flavoparmelia caperata	41
Rinodina roboris	40
Cliostomum griffithii	39
Parmotrema chinense	39
Physcia tenella	39
Opegrapha atra	38
Amandinea punctata	37
Lecanora expallens	37
Ochrolechia subviridis	37
Lecidella elaeochroma	36
Punctelia borreri	36

Melanelia fuliginosa ssp. glabratula	36
Ramalina fastigiata	36
Physcia adscendens	35
Lepraria incana agg.	33
Candelaria concolor	32
Physconia grisea	31
Parmelia sulcata	28
Enterographa crassa	27
Normandina pulchella	27
Ramalina canariensis	27
Puntelia subrudecta	26
Pyrrhospora quernea	26
Evernia prunastri	24
Physconia distorta	24
Acrocordia gemmata	23
Anaptychia ciliaris	23
Lecania cyrtella	20
Arthonia impolita	19
Caloplaca obscurella	19
Caloplaca citrina	18

From those trees surveyed so far a pattern is beginning to emerge of species that appear to be sensitive to agricultural pollution and those that are relatively tolerant of or increase with a rise in nitrogen levels. Table 5 lists these as sensitive and tolerant species respectively. However, there is another smaller but interesting group of 'middling species' that are tolerant of certain levels of Ammonia, but are not tolerant of hypertrophication. Anecdotal evidence suggests that these species are increasing their range in Dorset and in other counties. These species could potentially be used as biomonitors to indicate changes in Ammonia levels.

Table 5. Species indicating sensitivity or tolerance to Nitrogen compounds in Dorset

Sensitive species	**Middling species**	**Tolerant species**
Anaptychia ciliaris	*Candelaria concolor*	*Amandinea punctata*
Bacidia rubella	*Flavoparmelia soredians*	*Diploicia canescens*
Caloplaca cerina	*Lecanora barkmaneana*	*Hyperphyscia adglutinata*
Caloplaca obscurella	*Physcia tribacia*	*Phaeophyscia orbicularis**
Caloplaca ulcerosa	*Punctelia borreri*	*Physcia adscendens**
Physcia clementei		*Physcia tenella**
Physcia tribacioides		*Physconia grisea*
Physconia distorta		*Ramalina farinacea*
Physconia perisidiosa		*Xanthoria parietina**

* = species considered to be nitrophytes (van Herk 1999).

43% of the trees surveyed were situated within improved pastures (Table 6.). These grasslands were grazed mainly by dairy cattle, but sheep and horses were present in some fields. It is significant that despite the improved state of the fields and the continuation of a fairly intensive agricultural system a very varied flora survives. The reasons for this appear to be:

∉ The relatively small size of individual farms.

∉ The lack of large scale indoor livestock farming resulting in little need for large slurry storage systems.

∉ The continued use of solid manure and granular fertilisers limiting the impact of 'splashing' direct onto trees.

∉ The relatively limited use of large scale liquid slurry spreading.

Table 6. Habitat of the 51 trees surveyed in Dorset

Habitat	**No. of trees**
Improved pasture	32
Semi-improved pasture	13
Amenity/Urban	9
Roadside	8
Arable	2
Farmstead	2
Unimproved pasture	1

The future

Despite the apparent good health of the general lichen flora changes in agricultural practices in recent years pose a serious threat to wayside trees and their epiphytic flora. These threats include:

∉ The change from small scale livestock farming to large scale intensive systems.

∉ An increase in the growing of maize and subsequent increase in the winter application of liquid slurry.

∉ The decrease in animal numbers resulting in a change to a more intensive arable regime.

Protection of the wayside tree flora

Several recent initiatives could potentially aid the protection of wayside trees and their associated flora. Agri-environment schemes such as Environmentally Sensitive Areas (ESA) and Countryside Stewardship (CS) are now in place over large areas of the county. Where these schemes overlap with wayside tree 'hotspots' they could be used to protect and enhance conditions by:

∉ Limiting the application of liquid fertiliser and slurry.

∉ Encouraging the traditional pastoral farming systems that formerly prevailed in these areas.

∉ Encourage the planting of replacement trees.

Much of Dorset falls within a Nitrogen Sensitive Area. This should help reduce the amount of manure and fertiliser use. However there is also a restriction on the time in which manure and fertiliser can be applied. Potentially this could have the effect of a lot of manure being spread over a very short period of time. This 'short sharp burst', particularly in winter months when most lichens are growing, could have a detrimental effect at a local level.

Research and awareness

Research on the effects of nitrogen on lichens across Britain is urgently needed.

While casual observations suggest that the present lichen community has developed in association with, and because of, a certain type of agriculture, what is the threshold when the lichen flora starts to deteriorate? This is essential if we are to conserve the current diverse flora. This could be done in conjunction with agri-environment schemes, and raising the awareness of the importance of the wayside trees flora.

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The impact of a changing pollution climate on the conservation of lichen--rich sites in England and the UK

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Summary

This study reviews the evidence for the impact of the changing pollution climate and in particular nitrogen compounds on Sites of Special Scientific Interest (SSSIs) which have been notified for their lichen interest, using examples primarily from England. It also considers the implications for reaching or maintaining the government's public service agreement of '95% by area of SSSIs being in favourable or unfavourable recovering condition by 2010'.

Sites of major lichen interest and their distribution

In England there are about 130 SSSIs where lichens are the reason that the site has been notified or are a principal component of the interest; a further 109 sites mention lichens in their citation. These cover a wide range of habitats (Table 1) and range in size from 0.89ha. (Stokenham, Devon) to 28948 ha. (The New Forest, Hampshire)

Table 1 Sites where lichens are either the reason or a major reason for selection

Habitat	No of sites	Percentage
Parkland/woodpasture	24	18.5
Woodland	59	45
Heathland & grassland	13	10
Lowland rock including sarsen stones	8	6
Upland rock	4	3
Montane	4	3
Rivers	2	1.5
Maritime cliffs & heath	15	11.5
Maritime dune & shingle	5	3.5

Although these sites are distributed throughout England (figures 1 and 2) many of the most species-rich lichen communities of the UK are associated with oceanic conditions of high rainfall and leached bark in the south and west of Britain, acidic vegetation on nutrient-poor uplands or with maritime rocks and heathlands along the extensive coastline. The contrast between east and west is also marked in that sites in the east are sparser due to the destructive effects of SO_2 where only acid-tolerant species survived, despite the continuing presence of nitrogen in many forms (James & Davies 2003).

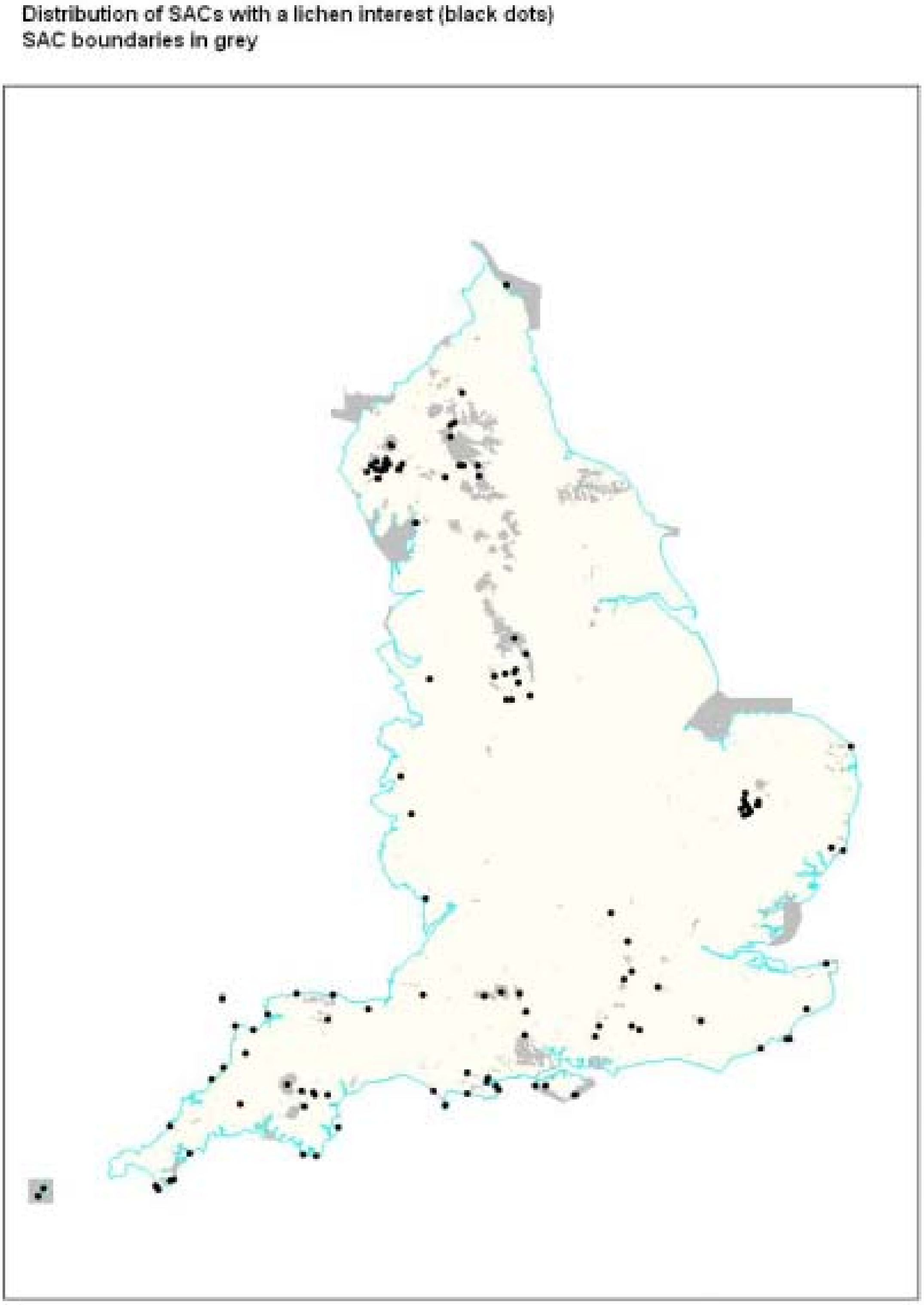
Distribution of SACs with a lichen interest (black dots)
SAC boundaries in grey

Figure 1

Distribution of SSSIs with a lichen interest (black dots)
SSSI boundaries in grey

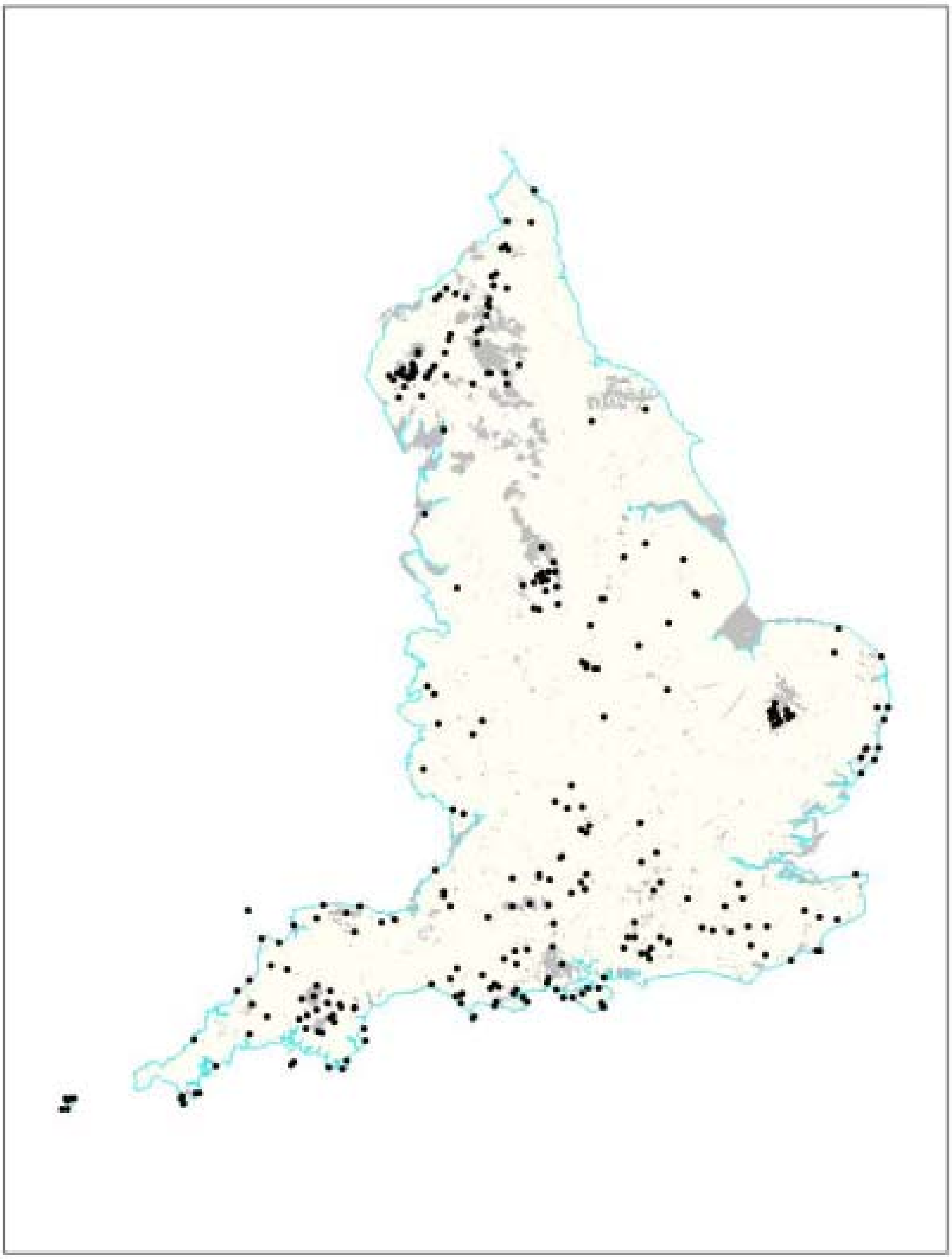

Figure 2

In western areas lichen-rich woodlands and parklands, such as the New Forest, Hampshire, Nettlecombe, Somerset and Bocconoc, Cornwall, are often ancient sites containing a high number of indicators of long Ecological Continuity (Rose 1992, Coppins & Coppins 2002). Many of these species are also sensitive to changes in environmental and atmospheric conditions of increasing SO_2 or nitrogen deposition. In the past the high rainfall and prevailing westerly winds protected western areas from the influence of SO_2 deposition from industrial and urban areas further east, but the increase in nitrogen deposition in rural and urban areas has led to the loss of acidiophytes including SO_2 tolerant species and a steady increase in nitrophytic species (Wolseley and others in this volume). This switch was recognised by foresters and ecologists alike (Woodin & Farmer 1993) so that it is now of great concern that the apparent increase in nitrogen is accompanied by a marked loss of sensitive acidophyte species and an increase in nitrophytes across the country in a great range of lichen habitats from epiphytic to terricolous and saxicolous.

The pollution climate

The pollutants concerned are diverse and have many sources so that it is difficult to attribute changes in lichen communities to the broad spectrum of N pollutants and their sources. NH_3 is deposited locally around the source from intensive livestock rearing or arable land (Sutton and others,this volume) whereas NH_4 may be carried hundreds of kilometres thus affecting acidophytic lichens at a great distance from source (van Herk 2003, this volume). NO_2 is produced in ever increasing amounts by road traffic but there appears to be very little evidence of its impact on lichen communities (van Herk, this volume). Recent research has shown that the greatest increase in NOx is from shipping (*Acid News* 2003), which may affect many of our important coastal sites. There is a strong correlation between ammonia deposition, substrate pH and loss of acidiophytic lichens in rural areas. However the rapid increase in nitrophytes in urban areas is not correlated with a single factor, there being many contributors to urban atmospheric pollutants including both N compounds, particles, and a shift to a warmer climate. Work in Italy suggests that dust plays an important role in the increase in nitrophytes (Loppi, this volume) and that global warming has a part to play in encouraging nitrophytes (Aptroot, this volume).

Whilst there is plenty of evidence for a changing pollution climate and its impact on lichen communities in urban areas as well as in the wider countryside, there is little direct evidence for impacts of nitrogen deposition on SSSIs. Recent work at Epping Forest has shown remarkable changes during a recording period between 1784 -2003. Nitrophytes and old woodland species present in 1784 were subsequently lost and replaced by acidophytes tolerant of high levels of SO_2. At the present time despite the loss of most of the epiphytic and terrestrial acidophytes the diversity has increased considerably due to the many nitrophytes that have colonised during the last decade (James & Davies 2003). However this effect is not restricted to areas of former SO_2 deposition and it appears that rural habitats across the country are showing a shift in species composition from acidophytes to nitrophytes. Recent work on oak twigs at Tycanol SSSI in Pembrokeshire has shown that between 1995 - 2003 there has been a sharp drop in acidophytes and the appearance of nitrophyte species on the boundaries of agricultural land (Larsen & Wolseley, in prep.). This is associated with an increase in bark pH and in accumulated nitrogen within the lichens. An investigation of lichens on twigs and trunks at IGER North Wyke, Devon also showed that although oak trunks continued to support a few sensitive species including *Lobaria pulmonaria* and *Teloschistes flavicans* that acidophytes on the twigs were reduced in the proximity of the farmyard and slurry tanks and nitrophytes increased (Wolseley & James

2002b, 2002c, also this volume). In areas where SO_2 deposition has been a dominant factor in the past the results are rather different. At Thetford in Norfolk the lichen flora of ancient trunks remained depauperate whereas younger branches and twigs supported a considerable lichen flora.

Evidence for change on SSSIs and from threatened lichens

Papers in this volume include many examples of changes in lichen communities where nitrogen is implicated. However it is often difficult to link these directly to increasing nitrogen deposition due to several factors; a) field surveys only provide correlative evidence, b) the gradient of climates across the UK makes it difficult to extrapolate between sites in the east and west, or north and south, c) there is still very little atmospheric monitoring in areas of high nature conservation. Furthermore there are additional factors such as climate change or loss of a lichen-rich substrate such as was formerly provided by elms prior to Dutch elm disease.

The effect of atmospheric nitrogen within different habitats also varies with environmental conditions. Whereas lichens on ancient trunks in woodlands may be somewhat protected by a buffer zone of trees, heathlands are exposed to nitrogen deposition and the consequent loss of nitrogen-sensitive species over extensive areas. The coasts of the UK are one of the richest habitats for lichens and contain many species associated with increased nutrients from both the sea spray and from sea bird colonies. This natural form of nitrogen deposition has over time resulted in the development of lichen communities containing rare and restricted species such as occur adjacent to sea bird colonies on Lundy and Skomer (James and others 1996, Wolseley and others 1996). Changes in the type of nitrogen deposition may have deleterious effects. Some examples of factors associated with nitrogen and loss of species in a range of habitats across Britain follow.

Woodlands

The importance of woodlands and parklands for lichen communities is reflected in the number of sites that are important for lichens (Table 1). In addition the use of lichens to assess the ecological continuity of ancient woodlands is well known (Rose 1992). However many of these communities are sensitive to both increasing acidification and nitrophication.

A UK wide project undertaken from the Natural History Museum between 1986 -1990 used fixed quadrats to assess changes in cover of species of *Lobaria* and the Lobarion in nature conservation sites. Sites were selected close to atmospheric monitoring stations, but in conditions of falling SO_2 values the evidence for thallus loss being associated with acidification was only strong in areas where acid deposition remained high (eg the Pennines). In other areas the deterioration of *Lobaria* species continued despite amelioration of acid deposition. Many of these sites are in open wood pasture and parkland adjacent to agricultural improvement where atmospheric nitrogen is increasing. Other factors that may affect loss of indicators of ecological continuity include loss of veteran trees and alteration of the habitat. (Looney & James 1989; Wolseley & James 2000).

There is plenty of evidence that the acidophyte components of the lichen flora are declining: The decline of *Bryoria* species on the continent can be linked to increased long range ammonium deposition (van Herk 2003, this volume). Species such as *Usnea florida* are decreasing rapidly in south-west Britain (B. Benfield & P. James, *pers. comm.*), *Ramalina* *fraxinea* is another species that has declined in a similar but as yet unexplained manner. The decline of *Anaptychia ciliaris* and *Pleurosticta acetabulum* since the 1970s can be linked with the spread of Dutch elm disease and the loss of mature elms. These species survived initially on other phorophytes eg *Fraxinus* and *Acer pseudoplatanus,* but appear to be relict and unable to colonise new substrates in the current pollution environment. These species still continue to be lost from sites eg Fritton Common, Norfolk & Sotterley Park in Suffolk although the trees remain.

Heathlands and chalk grasslands

Evidence for the decline in vegetation of characteristically nutrient-poor heathlands in association with increased N deposition came originally from the Netherlands (Woodin & Farmer 1993). In East Anglia heathlands have also shown a marked decline in acidophytic vegetation and a corresponding increase in grass cover which Pitcairn and others (1991) suggest is partly due to increased N. However at the same time the loss of close grazing by rabbits (due to myxomatosis and haemorrhagic disease) has led to an increase in coarse vegetation and tree cover These factors lead to rapid changes in microclimate and soil conditions independently of the toxic affects of atmospheric pollutants. In the Breckland of East Anglia the decline of Fulgensietum communities on Lakenheath Warren SSSI began in about 1986-7 and have been documented (Hitch & Lambley 1996, Gilbert 2001, 2003). This community occurs on very shallow calcareous soils low in nutrients. Its decline is associated with several other changes occurring in this environment such as the cessation of sheep grazing, invasion of pine seedlings, and increased bryophyte cover. However Gilbert (2003) considered that the underlying cause of the decline was most likely to be increased nitrogen deposition resulting in a change in the competitive balance between species. This shift in the community has resulted in the complete loss of the Biodiversity Action Species *Buellia asterella,* which is now extinct in Britain. This community also largely disappeared from three other Breckland SSSIs in East Anglia (Deadmans Grave, Thetford Heath and Little Heath Barnham) at about the same time.

In the west of Britain monitoring of a population of *Fulgensia* at Stackpole Warren National Nature Reserve in Pembrokeshire showed a strong decline in the population of *Fulgensia* associated with an increase in bryophytes and grasses *Dactylis glomerata* and *Brachypodium pinnatum.* This site was adjacent to intensively stocked grassland with a strong smell of livestock during a winter survey of the site (Wolseley & James 2000a). Ammonia concentrations were monitored at Stackpole as part as part of the National Ammonia Monitoring Network (Sutton sand others 1998, 2001) during October 1996 - November 1997. The mean monthly concentration was 2.6 (range 0.42 - 7.2) µg m-3, with peak concentrations coinciding with autumn and spring when manure spreading is most common. Although these concentrations are clean by comparison with Dutch conditions, they are larger than the concentrations shown to effect twig acidophytes in Scotland (Sutton and others, this volume), and background concentrations in the UK are around 0.05 - 0.3 µg m-3. This suggests that the loss at Stackpole is entirely consistent with increased NH_3 emissions. The evidence that van Herk (this volume 2003) presents on the effects of long range ammonium suggests that there may well be unassessed effects on upland heaths in high rainfall areas.

Rock surfaces

Rock surfaces support a large component of the saxicolous lichen flora in montane and maritime regions of Britain including many rare and restricted species of habitats that have often remained unaffected by agricultural improvements. As yet there is little evidence for changes in these communities, but the evidence for long range dispersal of NH_4 suggests that these sites should be monitored in order to detect changes in populations of sensitive species. However in lowland areas the story is very different where surface nutrient is often available, resulting in increased algal growth (*Desmococcus* and other species) on the surface of the lichen which in effect shades out and kills the underlying photobiont causing the death of the lichen. A survey of sarsen stones on Fyfield Down SSSI, Wiltshire identified considerable damage to lichen communities. It was observed that these changes followed a change in surrounding livestock grazing which increased the number of stock fourfold for shorter periods of time (O'Dare & Coppins 1994). A recent survey of Stonehenge, Wiltshire where grazing has been locally excluded, showed little evidence of invasive nitrophytes and no deterioration in long established lichen communities (Giavarini & James 2003, Gilbert 2003b), suggesting that there was little effect from surrounding land management.

Teloschistes flavicans was formerly more widespread in both saxicolous and epiphytic habitats but is now mainly restricted to islands off the west coast of Britain and maritime sites in the west of England and Wales (Gilbert & Purvis 1996). Although this species was considered to be sensitive to SO_2 its continuing loss in mainland sites and its healthy status on island sites such as Bardsey, Lundy, Skomer and Ramsey where there is no agricultural influence suggests that changes in N deposition may be a contributing factor (Fletcher 2001, James and others 1996, Wolseley and others 1996).

Freshwater habitats

There is a considerable body of evidence from other groups of organisms to show the effects of excess nitrogen on freshwater systems but there is as yet little research into how nutrient run-off from agricultural land can affect lichen communities. Monitoring of populations of the aquatic Biodiversity Action Species *Collema dichotomum* has shown that that this species is associated with salmon or trout waters (eg River Eden SSSI, Cumbria) where surrounding land use is unimproved and water quality high (O'Dare & Coppins 1995).

The future

There is a Public Service Agreement target for '95% by area of SSSIs throughout England, Wales and Scotland to be in favourable or unfavourable recovering condition by 2010'. In a changing pollution climate this is a major challenge with respect to lichen sites. If we are indeed seeing a trend to increasing nitrophyte-dominated lichen communities, even in the high rainfall areas of the south and west, it is difficult to see how this can be reversed except by the general lowering of ammonia and ammonium emissions over the next ten and more years. It is unlikely that, even if achievable, it would match the dramatic decline of sulphur dioxide levels since the 1970s because of the far greater number and diversity of sources. It might be possible to combat some of the effects of nitrogen on heathland and calcareous grassland lichen communities through techniques of nutrient-removal such as turf-stripping and changes in grazing regimes, but it is difficult to see how this could be achieved with respect to epiphytic and saxicolous communities. The planning system provides a mechanism to prevent or control new ammonium emission sites, such as livestock units close to SSSIs, but cannot be effective in relation to longer range ammonia emissions because of the wide range and type of sources and trans-boundary issues. Encouraging a change in land use involving lower fertiliser inputs and stocking rates via various environmental incentive schemes may also be a management strategy where effects are more localised.

Despite the observed shifts in lichen communities the complex mixture of N pollutants makes it difficult to obtain the evidence for the effects of nitrogen on lichen communities. A recent review outlines methods available for detecting changes in lichen communities due to N deposition (Sutton and others 2004) but emphasises that further critical research is required in order to; a) establish baselines for assessing changes, b) develop monitoring and surveillance procedures, c) establish monitoring of atmospheric nitrogen components in both rural and urban areas, d) estimate rates of accumulation and thresholds of the lichen species. The focus on rare species and their distribution within a site provides important information on the quality of the site but it requires specialist knowledge to set up and to assess changes in species composition or population change with time. Although permanent quadrats have been used to monitor BAP and other species the assessment of factors influencing a whole population requires other methods such as that developed for *Teloschistes* (Wolseley & James 2000). Simpler methods to monitor changes in lichens on recently available substrata such as lichens on twigs may provide an early warning system for ongoing environmental changes in SSSIs and surrounding areas (Wolseley & Pryor 1999, Sutton and others this volume). This method is at present being tested in the vicinity of ammonia monitoring stations and conservation sites across the UK. Results from this survey should allow a preliminary evaluation of climatic and geographical variation in response of lichens on twigs and trunks to a major N compound - ammonia. This can be used as a basis for further work on other N compounds and their affect on sensitive species.

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Session 4 Discussion

Conserving lichen communities and species diversity

Participants: Bryan Edwards (chair), Ann Allen, Jenny Duckworth (raconteur), Vince Giavarini, Peter Lambley, Laurens Sparrius, Gill Stevens.

Questions arising from Session 4

- What does 'favourable condition' mean for lichens?
- Consideration of N impacts on protected sites.
- Role of buffer zones and how to delimit?
- Raising profiles amongst site managers.
- Developing key messages to inform the legislative process.
- Need for funding for research (both long and short term).

Favourable condition for lichens. Recognised the need for a simple way of assessing favourable condition - so that an ordinary conservationist can assess if there are any problems, for example ivy on trees or slurry in fields. Neil Sanderson has had a go developing favourable condition assessment for National Trust on parkland (pers. com.). An equivalent proxy system used for woodlands in the Netherlands.

N impacts on protected sites. Nitrogen impacts cause a shift in lichen communities and this shift needs monitoring. Recognition of the need to update lichen phytosociology since James and others 1977 and to incorporate recent literature. Lichen phytosociology should be 'up there' with the NVC - something that should be pursued by the Conservation Committee of the British Lichen Society. The need to revisit and revise the BLS habitat inventories was also recognised.

Discussion on whether there is a danger of attention shifting too much towards nitrophiles following previous messages from `lichen deserts' and `nettle' equivalents?

Impact of N on local/county wildlife sites is important as well as on internationally important ones and provides potential to work with wildlife trusts?

How wide should buffer strips be? Need more research but seems that it's not possible to generalise. Small fragmented sites are particularly vulnerable due to edge effect. Whole farm initiatives, for example Tir Gofal would benefit wayside trees. Guards around trees not a good thing because of shade producing weedy vegetation such as nettles.

Raising lichen profiles and developing key messages to inform the legislative process: Need to target agri-environment schemes and local farm advisors, and find ways of popularising wayside trees among farmers and the public.

To improve public awareness there is a need to get the press and media involved – for example Countryfile slot on nitrogen and epiphytes, also target getting articles into Farmers Weekly, British Wildlife. The BLS education workshop in Aug 03 aims to come up with simple projects for young people - a simple assessment of N impacts could be suitable Consider having a wayside tree workshop involving FWAG, WTs, Local Authorities, DEFRA etc.

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Top left: Using a home-made moth trap.
Peter Wakely/English Nature 17,396
Middle left: Co_2 experiment at Roudsea Wood and Mosses NNR, Lancashire.
Peter Wakely/English Nature 21,792
Bottom left: Radio tracking a hare on Pawlett Hams, Somerset.
Paul Glendell/English Nature 23,020
Main: Identifying moths caught in a moth trap at Ham Wall NNR, Somerset.
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