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Centre for Ecology & Hydrology

Centre for
Ecology & Hydrology
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CENTRE FOR ECOLOGY & HYDROLOGY

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Executive Summary

BA Emmett[1]

Centre for Ecology and Hydrology

1. This report describes the work carried out by the UKREATE (UK Research on the Eutrophication and Acidification of Terrestrial Ecosystems) consortium in the latest phase of the NERC-DEFRA Terrestrial Umbrella (2004 – 2007). Key findings are reported against individual Work Packages and Tasks as detailed in the tender specification.

Work Package 1:

2. *Task 1 (CEH Bangor):* We have continued to provide advice and support to the UK National Focal Centre (UKNFC) on Critical Loads Mapping currently at CEH Monkswood. This has involved responding to queries as requests come in from the CCE. All UKREATE members have been available for specific queries and updates as required and activity has included attendance at the CCE update meeting in Slovenia including a keynote talk.
3. *Task 2 (U York):* A more sophisticated approach to mapping empirical critical loads of nitrogen, using differentiated values within the range recommended for application within UN/ECE based on ecosystem characteristics, has been developed. This will improve the quality of information that can be provided for national and local mapping, as well as for site-specific applications. The approach that has been developed, after active consultation with potential users, provides a decision support matrix based on a combination of data from the National Vegetation Classification (NVC), site-specific information where appropriate, and endorsement theory to assess the strength of evidence supporting a particular recommended critical load value. The decision support matrix has been developed for heathlands and will shortly be sent to user groups for further comment and consultation.

Work Package 2

4. *Task 3 (U Sheffield):*Experimental work has continued at Wardlow Hay Cop in the Peak District to identify (i) if floristic and soil changes be reversed in grasslands that have exceeded critical N loads, (ii) how long will recovery take without active interventions, (iii) the effects of the accumulation of a soil N 'reservoir' on ecosystem functioning and cation and nitrate leaching and (iv) if losses of cations from nitrogen saturated grassland soils have long-term irreversible effects. Results indicate that whilst inorganic-N levels in soils decline quickly, losses of base cations are likely to have a more long lasting and significant effect with negative consquences for the recovery of populations of base-demanding species and in particular their flowering.
5. *Task 4 (MMU):* Continuation of the Ruabon experimental site has indicated surprisingly good recovery of both *Calluna* growth and reduction in inorganic-N levels in the soil following reductions in N treatments. This contrasted with continued negative re-growth at a lowland heathland site at the Budworth site suggesting timing of recovery will be variable across different heathland types. A regional survey of nitrogen indicators in *Calluna* moorland from sites in Scotland, Wales and the Peak District has investigated a range of plant and soil N and P measurements and biodiversity. The exchangeable N in surface litter layers was closely related to N deposition and the consequence of this elevated pool of available nitrogen may be to reduce the diversity of the bryophyte community within the *Calluna* canopy and may be a useful monitoring indicator.

6. *Task 5 (Imperial College London)*: Nitrogen addition to experimental plots at Thursley Common NNR continues to have large and significant effects on the growth, canopy development and phenology of *Calluna.* Nine years after the cessation of nutrient additions, there is still clear evidence of former N treatments on the *Calluna* canopy, as well as effects on flowering and bud burst. This confirms that ecosystem recovery from eutrophication is a slow process, even when initial nutrient loadings are relatively low, and highlights the need for long term experiments to monitor these responses. Experimental addition of N and P to P-limited heathland mesocosms has also provided the first evidence to emerge which suggests that elevated N deposition can have both direct and indirect detrimental effects on P-limited systems, with implications for the role of P-limitation as a potential modifier of critical loads. A survey of 33 lowland heaths in southern and eastern England has provided information on the extent of variation in soil and plant nutrient status, and the relationships between these. Results indicated that more than half of the sample sites had either NP co-limitation, or P limitation. Data also show that soil N and P contents increase with stand age, highlighting the importance of controlling for such factors in wide scale surveys of this nature.
7. *Task 6 (ADAS)*: N and grazing treatments have continued at the Pwllpeiran site to evaluate the interaction with of nitrogen deposition with the intensity of sheep grazing together with the relative importance of reduced versus oxidised nitrogen on species change. Phosphorus additions were applied to one set of plots to test the hypothesis that N impacts would be greater in non-P limited systems. No measurements were made this year with final vegetation composition measurements due to be made in 2007. A synthesis of the findings from this study were presented at the Biogeomon 5th International Symposium on Ecosystem Behaviour in Santa Cruz, California, June 2006.
8. *Task 7 (CEH Edinburgh)*: The Whim bog experiment has continued to examine the effects of wet versus dry and reduced versus oxidised N deposition in a N-limited bog community. Results have indicated that effects of ammonia gas from point sources on plant species are concentration mediated and NH_3 sensitive ombrotrophic bog species will not be protected from irreversible damage from high NH_3 concentrations by either the monthly or annual Critical Levels proposed by Van der Eerden *et al.* (1991). Other key findings include the conclusion that the use of N tissue status of mosses as a bioindicator for N impacts has some caveats namely extended exposures to high N deposition or [NH_3] can lead to low N concentrations as cells become damaged and leaky. Amino acid levels may be more indicative of negative N impacts, but on a routine basis these are costly to analyse and require controls and a range of N doses for interpretation. Other findings include the conclusion that loss of individual plant species appears to require the interaction between high N doses and a stress, i.e. the accumulation of N on its own does not appear to be toxic. Also, N deposition at ~ 2 x the Critical Load for moorland inhibits the regeneration of *Calluna* from old wood but not the establishment of *Calluna* seedlings. Previous treatment with either ammonia or wet ammonium has not inhibited colonization by mosses.
9. *Task 8 (CEH Bangor)*:The timing of recovery following reductions in N deposition and implications of climate change for responses to N deposition are being investigated in two experiments. One is located in a relatively non-polluted upland heathland site at Clocaenog, North Wales which has been running for 8 years. The second is a new experiment in a upland moorland site in a heavily polluted site at Peaknase, in the Peak District. An experimental approach is used which uses retractable curtains to: (i) experimentally reduce N deposition, or (ii) create repeated summer drought or (iii) produce whole ecosystem warming. Changes in water quality, species composition and soil organic matter turnover are being assessed . The equipment for the Peaknase

experiment has now been successfully moved from Mid Wales to a site in the Peak District although this was delayed. Baseline measurements have been made and treatments started in early July 2006. The second field site in Clocaenog with 8 year long climate change manipulations on a heathland in less polluted conditions has been continued and results have highlighted the importance of repeated summer drought in mobilising both carbon and nitrogen from the soil store. Results from this site continue to be presented at a range of national and international conferences and future work will integrate the results to inform both model development and critical load assessments.

10. *Task 9 (MLURI):* Four montane areas in Scotland with a high concentration of archive data points of vegetation composition and contrasting in terms of land-use intensity and pollutant inputs and contained a range of plant communities have been identified and re-sampled. These were the Cairngorms, Southern Uplands, the Isle of Mull and the Orkney and Shetland islands. In 2005/6 a total of 95 sample points were re-located and re-recorded on Mull (34) and in the Southern Uplands (61). An initial, basic analysis of the data collected so far shows that a shift in the vegetation composition can be detected across all the habitats sampled, but that there is some evidence that the shift may be greatest in snowbed communities and *Racomitrium* heath.
11. *Task 10 (U York):* A botanical re-survey of a sample of a network of calcareous grassland sites that were established in the early 1990s by Prof. John Rodwell and colleagues of Lancaster University (with support from Defra) to monitor long-term impacts of air pollution and climate change, has been undertaken. Bryophyte species composition was surveyed at 16 of the sites in the summer of 2005. While significant changes in frequency of individual species were found, there was no significant association with modeled nitrogen deposition. Further survey work focusing on higher plants, with associated soil analysis, at a wider range of sites is planned for the summer of 2006.
12. *Task 11 (U York):* Two models have been developed that allow prediction of seasonal variations in nitrate concentrations in river water for unmanaged or minimally managed catchments across the UK. These will allow prediction of long term changes in nitrate leaching in response to changes in N deposition. Winter maximum and summer minimum nitrate concentrations have been shown to be related primarily to deposition flux of oxidized N species, and are apparently not significantly influenced by precipitation amount. However, annual precipitation has been shown to be a key driving variable in the prediction of high flow and low flow calcium and alkalinity concentrations across the UK, in accordance with a hypothesis based upon intuitive expectation. Work on the effects of road-salting upon the disruption of N cycling in upland soils has continued with results that effects on N cycling are sustained and substantial. Preliminary results on assessment of N cycling at the Hob Moor Local Nature Reserve in York indicate that N deposition has resulted in soil N accumulation, low soil C:N ratios, and mobile ammonium and nitrate in the soils even during periods of active plant growth.
13. *Task 12 (CEH Lancaster):* Nitrate concentration data for three lakes in the Lake District has been continued extending the available information on trends to the period 1955 to 2005. After removing a few outliers from the 1950s data, the upward trends in all three lakes are highly significant ($p < 0.01$) however, the increases have not been steady, and there has been little change over the past 20 years. Sampling of streamwaters in the catchments of the lakes showed (a) that nitrate concentrations are not significantly affected by the small areas of improved grassland in the valley bottoms, and (b) that brown podzolic soils release less nitrate than rankers. The combined results for streams and soils indicate that nitrate leaching is not directly related to C:N ratio, as has been found for forest soils, and proposed for UK moorland soils. A N budget modelling based on inputs of N from long-term atmospheric deposition, and the nitrogen and carbon pools

of the two main soil types (brown and ranker) is being developed. Preliminary results indicate that the N pools in the soils can be accounted for in terms of deposition and leaching, primarily of nitrate and dissolved organic nitrogen.

14. *Task 13 CEH Lancaster and Bangor*: The conventional ^{15}N pool dilution technique has been redeveloped for meaningful application in the often anaerobic environments of peaty wetland soils which act as an important transition zone between terrestrial and freshwaters systems. ^{15}N pool dilution studies have been made at a wetland catena (Plynlimon, Wales) on three occasions in the past 18 months. The data are currently being processed and early analyses suggest that there is significant difference in the gross soil nitrogen fluxes between ombrotrophic, rheotrophic and mieneratrophic wetland soils which indicate they will vary in their ability to moderate transfer of N species into freshwater systems. It was noted that seasonal differences were less pronounced than intersite variability. A full data analysis will be made once all four data sets are intact.
15. *Task 14 (Forest Research):* Key findings from the EC and UNECE-ICP (Forests) Intensive Forest Health Monitoring (Level II) Network, which were established in 1994-5 include the confirmation of recovery of soil solution pH from high pollution loading at the Ladybower site in the English Midlands and a downward trend in soil solution sulphate concentrations at most sites. The steady increase in nitrate in soil solution at Tummel, that was observed between 1996 and 2003, was reversed in 2004 and 2005, with annual mean concentrations in 2004 half the value observed in 2003 at both 10 and 50 cm depth. The soil solution nitrogen dynamics at this site cannot be interpreted by dynamic models of ecosystem chemistry and, instead, a biotic explanation is offered, linked to observed peaks of litterfall and data from the UK Forest Condition Survey. The reasons for observed trends in crown condition at the level II plots are also evaluated. The simple mass balance equation for setting critical loads for acidity and nutrient nitrogen includes a growth uptake term for base cations and nitrogen, respectively. Progress is reported on a methodology for incorporating regional variation in productivity on the growth uptake term, together with an approach to represent the impacts of climate change. Progress is also reported on a review of peat pH values which is being conducted with the objective of providing a robust basis for confirming or amending the critical pH for organic soils in the UK.
16. *Task 15a (CEH Lancaster):* Work has continued to understand the increasing abundance of nutrient-demanding species in upland infertile habitats through analysis of Countryside Survey data. Results indicate that plant species less typical of upland vegetation types and more typical of lowland semi-improved grasslands, have increased in occupancy in Countryside Survey (CS) plots in upland Britain between 1978 and 1998. Relationships between the spatial and temporal changes in abundance of this species group with potential drivers (including change in growing season length, change in sheep density between 1969 and 2000, sheep numbers in 2000, cover of intensive Broad Habitat in 1998, total N deposition in 1996 additionally broken down into NHx and NOy) were examined as separate explanatory variables. Results indicated that whilst the probability of at least one nutrient-demanding, lowland mesophyte being present in an upland grassland, heath or bog plot in Countryside Survey data in 1998 was best explained by long-term average annual temperature and total N deposition in 1996, the richness of nutrient-demanding, lowland mesophytes in CS plots in 1998 was positively correlated with sheep grazing intensity and cover of intensive Broad Habitat in each 1km square. Conclusions were that a warmer, more lowland climate plus high N deposition seem to track the incursion of these nutrient-demanding species into upland Britain but more intense agricultural activity, particularly sheep grazing, is best correlated with a greater

richness of these species per square metre, and therefore with the most marked changes in local species composition and vegetation character.

17. *Task 15b (CEH Lancaster):* Work to identify potential botanical indicators of N deposition across upland, infertile vegetation types in Britain has identified significant correlations between modelled NHx deposition with 41% of the common higher plant species present in upland CS plots in the three survey years (total of 146 plant species tested). A smaller subset carried the strongest unique signal of NHx deposition. Overall more species-level differences in abundance between CS squares could be attributed to spatial differences in NHx deposition than to change in sheep density or growing season length. These results must be strongly qualified by the statement that they are correlative patterns that could be concealing other causal links, rather than causal relationships inferred more strongly from experimental manipulations.
18. *Task 16 (CEH Bangor):* Model based risk assessment of the vulnerability of rare coastal species to N deposition. No derliverables were due on this task.

Work Package 3

19. *Task 17 (CEH Lancaster):* Work has continued to develop the modelling framework for linking the soils-water chemistry model MAGIC and vegetation module GB-MOVE. This Task is linked to development and testing of biogeochemical models in the Defra Dynamic Modelling Umbrella coordinated by Chris Evans (CEH Bangor). A key focus this year has been the development of steps to estimate species pool of potential immigrants based on dispersal traits, species' broad habitat preferences and local broad habitat composition. Keynote talks on the approach have been presented at a series of UK and international conferences and a summary included in a major review of dynamic modelling approaches coordinated by RIVM.

Synthesis reports

20. Drafts of Synthesis reports for all key habitats are now available and are included as an Appendix. These will be available on the UKREATE website and at a range of national and international conferences where results are presented and circulated to key stakeholders including conservation agencies. In addition, a database for results from long term experimental field sites is under development and will be completed by all partners in Year 3 of the contract to fulfill quality assurance requirements.

Work Package 1:
Refinement of Critical Loads

Task 1:
Update and Refinement of Critical Loads

B Reynolds

CEH Bangor

Task 1 - Update and refinement of critical loads
B Reynolds

1 Summary

There have been no significant changes to the national critical loads data sets during the past year. The task has been mainly concerned with providing advice on the existing data sets in relation to the development of dynamic models to link soil biogeochemistry to above-ground biodiversity.

2 Policy relevance

This task provides technical input to the UK National Focal Centre (UKNFC) in support of their mapping activities and provision of data to the Co-ordination Centre for Effects. The work takes the form of contributing to the assessment and development of critical load models, provision and review of data and assessment of uncertainties.

3 Project update

The national steady-state critical loads data for acidity and nutrient nitrogen were significantly revised and updated in 2003. Currently there are no plans to further revise and update these data sets. Keeping the data 'fixed' will help in the current reviews of the NECD, the Gothenburg protocol and the Defra Air Quality Strategy, where the focus is on predicted changes in emissions and deposition.

The Co-ordination Centre for Effects issued a voluntary call for data in January 2006. However, as the UK submitted revised data to the CCE in February 2005 and as no further changes have been made since that time, no further data were submitted this year.

A watching brief on further developments in critical loads is being maintained by attendance at relevant national and international meetings such as TU project meetings and Acid Rain 2005.

4 Collaboration with Dynamic Modelling and Freshwater Umbrella

The current focus of critical loads activities in the UK and Europe is on nutrient nitrogen and the development of critical loads for nitrogen in relation to biodiversity. Dynamic models (eg MAGIC-GBMOVE) are being developed to link soil biogeochemistry to plant species diversity to inform this process. The national critical loads data and the data sets used to derive them are being used as inputs to these models to ensure consistency between the steady-state and dynamic approaches. This has required some consultation and review of assumptions and methods used in the Steady-state Mass Balance model, or example in relation to estimating base cation release from rock phosphate fertiliser applied to plantation conifer forests.

5 Outputs

The paper submitted in 2005 on uncertainties in critical loads has been published following peer review:

Heywood, E., Hall, J. and Reynolds, B. 2006. A review of uncertainties in the inputs to critical loads of acidity and nutrient nitrogen for woodland habitats. Environmental Science and Policy, 9, 78-88.

Work Package 1:
Refinement of Critical Loads

Task 2:
Refinement of Empirical Critical Loads of Nitrogen and their Application to the UK

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Task 2. – Refinement of Empirical Critical Loads of Nitrogen and their Application to the UK

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1. Summary

The aim of this Work Package is to develop a more sophisticated approach to mapping empirical critical loads of nitrogen, using differentiated values within the range recommended for application within UN/ECE, based on ecosystem characteristics. This will improve the quality of information that can be provided for national and local mapping, as well as for site-specific applications. The approach that has been developed, after active consultation with potential users, provides a decision support matrix based on a combination of data from the National Vegetation Classification (NVC), site-specific information where appropriate, and endorsement theory to assess the strength of evidence supporting a particular recommended critical load value. The decision support matrix has been developed for heathlands and will shortly be sent to user groups for further comment and consultation.

2. Policy Relevance

Definition of empirical critical loads as a range of values, with associated levels of certainty, is an important conceptual development in critical-load based risk assessment. However, application of this information in a policy context is poorly developed. For instance, in national critical loads risk assessment for the UK, a single critical load value has been selected within the recommended range and used to assess critical load exceedance. Because deposition rates often lie within the empirical critical load range, decisions on whether a site or grid square has a critical load exceedance may be highly sensitive to the choice of mapping value. This work package, although limited in scope, provides a framework decision support matrix that aims to provide a more objective and transparent basis for decisions on the most appropriate site-specific critical load within the range of values recommended for a particular habitat. This approach could be developed and applied in a range of contexts, including providing improved methods for national mapping of empirical critical loads of nutrient nitrogen, assessment of the impacts of deposition on individual sites under the Habitats Directive, and evaluation of the condition and future threats to the integrity of SSSIs.

3. Project Update

Based on the consultation workshop with potential users in 2005, a framework for the Decision Support Matrix (DSM) has been developed. Additional time has been spent to establish this matrix as an electronic tool with a user interface, because this will increase the range of applications and will provide a basis for further development and refinement in the future. However, adoption of this approach has led to some delays with the project. A document summarising the approach with an example of application to heathlands will be circulated to the user group by the end of June 2006, requesting a response within one month. Based on these responses, we expect that the final electronic DSM will be available for use by the end of September 2006. The text below summarises the approach which has been adopted, but it is emphasized that some of the details may be changed in developing the final version, after feedback from users.

At the consultation meeting, significant problems were identified with agency users interpreting the modifying factors identified at the Berne workshop, because of their limited knowledge and because of the lack of site-specific data on edaphic and climatic conditions. We have therefore adopted an approach, which emerged in discussion at the workshop, that links the DSM strongly to NVC categories, since this information would readily be available to site users. The use of the NVC has additional benefits:- it is a standardized database for the whole country, that can be automatically linked to the EUNIS categories used to define critical loads through the NBN national habitats directory; NVC categories are normally identified in site notifications; NVC categories reflect major climatic and edaphic gradients; and NVC databases can be readily be linked to information on component species, e.g. through the PLANTATT database. However, this assessment based on NVC categories will need to be complemented by site-specific information provided by the user.

Based in valuable input from Richard Wadsworth and Jane Hall (CEH Monks Wood), endorsement theory will then be used to combine the information provided by the user to identify the most appropriate critical load range from three options (lower end, mid-range, and upper end of 'Berne' values), with an indication of the strength of the underlying evidence (Wadsworth & Hall, in press). The workshop considered that using the critical load range in this way was more meaningful than using a single value given the uncertainties in data. This is a very important conclusion from the workshop, which has led to considerable modification of our planned approach and has wider implications for risk assessment using critical loads. An option is also under consideration to link the most appropriate part of the site specific critical load to the estimated deposition load for the site, derived from APIS using national grid references and hence to produce a likelihood of critical load exceeedance, although this is beyond the scope of this project.

4. Key Findings

Table 1 summarises the information which is entered into the DSM as a basis for the endorsement procedure to determine which part of the critical load range should be used. The PLANTATT database is used to calculate averages for the Ellenberg nutrient and acidity indices for each NVC category, and it is assumed that NVC categories with a low nutrient, a high acidity index and a higher proportion of lichens and bryophytes would have a lower critical load; sites with management leading to lower N removal would also have a lower critical load. The habitat status and presence of rare species are optional inputs which allow the user to take a more precautionary approach. Further development of the electronic DSM, which is beyond the scope of this project, would allow both addition of other variables, and refinement of the approach (e.g. by assigning nutrient and acidity values to lichens and bryophytes (not available in PLANTATT), and to rare species).

Table 1. Summary of Variables used to define Critical Load range

Variable	How defined	Link to user
Nitrogen status	Linked to Ellenberg nutrient index by NVC community	User defines NVC category
Acidity status	Linked to Ellenberg acidity index by NVC community	User defines NVC category
Proportion of lichens/bryophytes	Determined from species data for NVC community	User defines NVC category
Presence of rare species	Based on site-specific information	User defines number of rare plant species
Management of site	Based on site-specific information and history	User identifies from habitat-specific options
Habitat status	Based on Common Standards Monitoring for site	User defines site condition as 'favourable' or 'unfavourable' etc

5. References

Wadsworth RA & Hall JR (in press). Setting site specific critical loads: an approach using Endorsement Theory and Dempster-Shafer. Atmospheric Environment.

Work Package 2:
Impacts, Recovery and Processes

Task 3:
Grassland Soil and Vegetation Responses Following Nitrogen Saturation at Wardlaw Hay-Cop

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Task 3. – Grassland Soil and Vegetation Responses Following Nitrogen Saturation at Wardlaw Hay-Cop

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1. Summary

We have investigated the effects of 11 years of monthly applications of simulated atmospheric nitrogen (N) deposition on two of the most important types of species-rich limestone and acidic grasslands in the UK. Our experimental plots are located in the Peak District National Park, in an upland region that has experienced the highest cumulative 50-year N deposition rates in the UK. After a decade of simulated N deposition one set of experimental plots have been divided, one half of each now being in 'recovery' whilst the other half continues to receive treatment. Another set of plots established in 1990 have been in recovery since 2002. The N treatments are also combined with phosphorus fertilizer additions enabling the role of phosphorus availability in affecting responses to N deposition and 'recovery' to be determined.

- The mineral N pools in both soils have rapidly declined in the 'recovery' plots with no detectable enrichment in extractable mineral N in the limestone grassland after 3 years. There are residual effects of the previous N loads on extractable mineral N in the acid grassland (particularly of ammonium), but these are very modest by comparison to the mineral N pools when the plots were being treated with experimental N additions. Overall, the prospects for recovery of soil mineral-N pools following reductions in N deposition are very encouraging.
- Other soil chemistry changes are likely to present a more severe constraint on habitat recovery. The loss of base cations and associated acidification is very serious and unlikely to be readilly reversed by natural processes. Using a novel extraction procedure that mimics the activities of root exudates we have detected a 30-65% depletion of 'plant available' pools of calcium and potassium in the most heavilly N enriched soils.
- Consistent with the loss of bases is the evidence of decline in abundance of some of the most base-demanding species such as the calcicole grass *Briza media* and of many of the forbs characteristic of limestone grassland such as *Thymus polytrichus.* In the acid grassland there has been a major decline in *Conopodium majus* and *Anemone nemorosa.* In both grasslands the flowering of forbs has been even more severely reduced than their cover and we are monitoring flowering in order to detect early evidence of 'recovery' but no significant effect has been seen in the divided plots for the first 1-2 years after ceasing treatments.
- Changes in floristic composition, seed production and soil chemistry caused by long-term N saturation are likely to leave a long-lasting lagacy of habitat degradation in some of our most important types of semi-natural grasslands.

2. Policy Relevance

2.1 Grassland biodiversity loss- the scale of the problem

Grasslands are the most extensive semi-natural plant communities in the UK countryside, with over 20% of our native higher plants being specifically associated with these habitats (Preston *et al.*, 2002). Acidic and calcareous grasslands together cover an area in excess of 1.23 million hectares of the UK and are of exceptional conservation and amenity value as a result of their floristic diversity. Acidic and calcareous grasslands contain over 300 of the 540 native plants specifically associated with grassland (Preston *et al.,* 2002). Unprecedented decreases occurred in the area of semi-natural grassland communities in the UK from 1930-1988, with over 40% of the 179 native species whose distribution declined in this period being characteristic of calcareous, unimproved or acidic grassland/heathland (Rich & Woodruff, 1996). Recent evidence suggests that nitrogen pollution is a major factor explaining loss of species from acid grassland that has not been ploughed or fertilized, with over 55% of the variation in species richness across 68 sites throughout the UK being explained by their local rates of N deposition (Stevens *et al.,* 2004).

2.2 Nitrogen pollution- an ongoing threat to semi-natural grassland

Although increasingly effective emissions controls have decreased NOx emissions by 30% from 1990-2000 and are set to lower this further to less than half its 1990 value by 2010 (Air Quality Expert Group 2004), emissions of reduced N (NHy) remain at historically high values. Many grasslands are still experiencing N deposition rates in excess of their critical loads, particularly for ammonium as it is this form of N pollution that in national surveys has been shown to have greatest impact on grasslands (Smart *et al.,* 2004). Thus, despite major progress in emissions controls, N deposition continues to pose a significant threat to the long-term sustainability of species-rich semi-natural grasslands. Even where N deposition has now fallen below the critical load for a grassland the long-term legacy of having greatly exceeded this load is likely to continue to compromise the quality of the habitat.

2.3 Policy relevance of our experiments

Our long-term (11-16 year) experimental plots in semi-natural acidic and limestone grasslands in the Derbyshire Dales National Nature Reserve are ideally placed to help address the policy issues concerning N pollutant impacts on semi-natural grasslands. The plots are situated in one of the most heavily N polluted regions in UK, estimated to have received a total of over 200 g N m^2 from 1900-2000 (Fowler *et al.,* 2004), a value matched or exceeded on less than 10% of the total area of the UK. The grasslands are examples of the two most widespread and important types of species-rich limestone and acid grasslands in the UK, and are the kinds that have experienced the highest sustained N loads.

2.4 Key policy issues -pollutant impacts on semi-natural grassland

Much of our semi-natural grasslands have exceeded their currently defined critical loads for N over several decades- and there is increasingly strong evidence that these grasslands have suffered very significant loss of diversity and habitat quality as a result (Carroll *et al.,* 2003). Two key policy issues arise from our increasing awareness of the nature and extent of N deposition impacts on species-rich grasslands.

(a) The need for robust evidence of either continuing environmental damage or of recovery being constrained by current N deposition loads, to justify establishment of further cuts in emission targets- especially for ammonia.
(b) The need to establish whether recovery from the habitat damage due to N enrichment is constrained by long-lasting impacts that require active intervention (e.g. neutralisation of soil acidity and replenishment of base cations leached with nitrate; and re-seeding of species that have declined).

Our experimental studies of the long-term impacts of N deposition on plants and soil in the limestone and acidic grasslands at Wardlow Hay-Cop, and the responses of these habitats to reductions in simulated pollutant N inputs are directly addressing these key issues. Our findings are required to inform future emissions control policies and to guide the development of management of semi-natural grassland ecosystems that were saturated with N in the latter half of the 20[th] Century and have now entered a new era with progressively decreasing rates of N deposition.

3. Project Update

Our studies aim to address the following four questions:

1. *Can the floristic and soil changes be reversed in grasslands that have exceeded critical N loads?*
2. *How long will recovery take without active interventions?*
3. *What are the effects of the accumulation of a soil N 'reservoir' on ecosystem functioning and cation and nitrate leaching?*
4. *Will losses of cations from nitrogen saturated grassland soils have long-term irreversible effects?*

3.1 Eutrophication legacy in grassland recovering from N saturation?

The original $1m^2$ plots established on both limestone and acid grassland in 1990 were treated for 11 years with N additions ranging from 0-14 g m^2 y^{-1}. Since the end of August 2002 these plots have been in 'recovery'. We have continued to monitor the concentration of ammonium and oxidised N (nitrate) extracted by KCl from the soil through the period February-June each year (Fig. 1) as the peak annual concentrations of both ammonium and nitrate occur during this period.

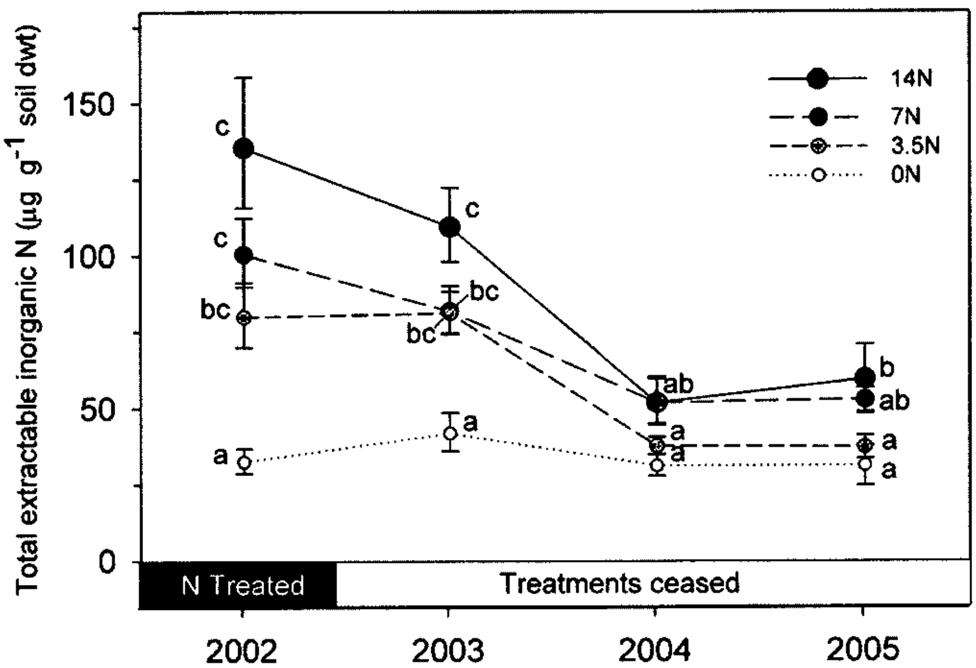

Figure 1. The mean concentration of total inorganic N extracted from acid grassland soil with 2M KCl (February-June) in the final year of an 11-year experiment on effects of simulated N deposition, and in the 4 years of 'recovery' that have followed. The N treatments were 0, 3.5, 7 and 14g N m^2 $year^{-1}$ above ambient deposition Vertical bars give SEM, n=4. Points sharing the same letter are not significantly different (Tukey test, P>0.05).

Both grasslands have shown rapid recovery in their 'available' mineral N pools, particularly in the second year after ceasing the treatments (data shown for the acid grassland - Fig. 1). Four years after ceasing N additions (April 2006), in the calcareous grassland there is now no evidence that the treatments continue to have lasting effects on the concentration of ammonium in the soil (Fig. 2). This is encouraging evidence that the plant-available nitrogen pools in the particular kind of limestone grassland we have studied can 'recover' if N inputs decrease. In the acid grassland, however, residual eutrophication effects are more persistent- significant N dose-related increases in soil ammonium are still found in the organic-rich Ah horizon in this grassland (Fig. 2) although the concentration is now less than half the April peak values (> 125 µg ammonium g^{-1} soil dry weight) that occurred in the last year of N treatment (2002) and the first year of 'recovery'.

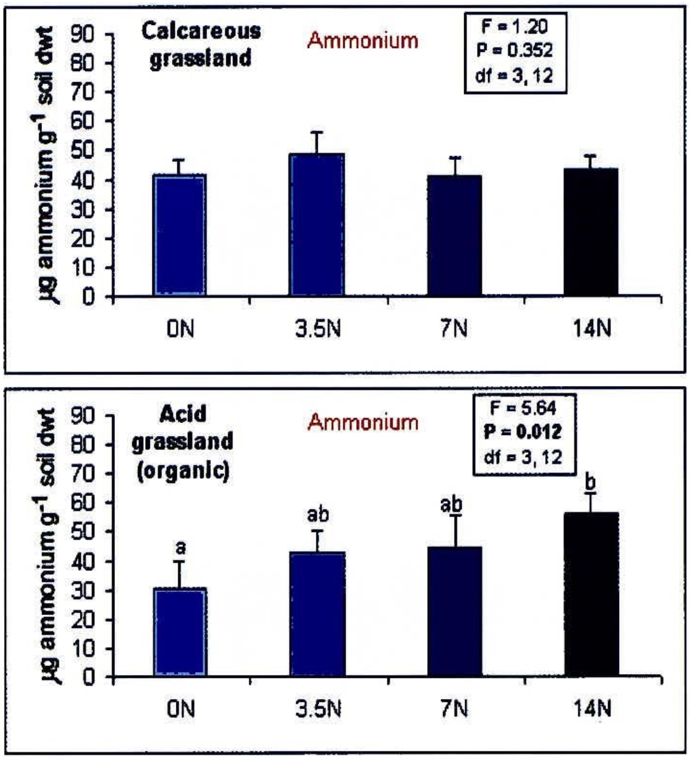

Figure 2. The concentration of ammonium extracted with 2M KCl from calcareous and acid grassland (organic horizon) soil at Wardlow Hay-Cop, four years (April 2006) after ceasing simulated pollutant N deposition inputs at rates of 0, 3.5, 7 and 14 g N m^2 y^{-1} for more than a decade. Vertical bars are SEM, n = 4. Bars sharing the same letter are not significantly different (Tukey test, *P*>0.05).

There is also a trend in both grasslands for the total oxidised nitrogen concentration (primarily nitrate) to be higher in the formerly treated plots, although this is not significant due to the large variation between replicates relative to the magnitude of the values measured (Fig. 3). In the calcareous grassland nitrate is a much more important component of the total extractable N pool with concentrations often an order of magnitude higher than in the acid grassland (Fig. 3). The leaching of nitrate is a much more important process in the limestone grassland (Phoenix *et al.,* 2003a) and may contribute significantly to the export of N from the soil, assisting the 'recovery' to back-ground concentrations, but also posing a threat of nitrate pollution of drainage waters.

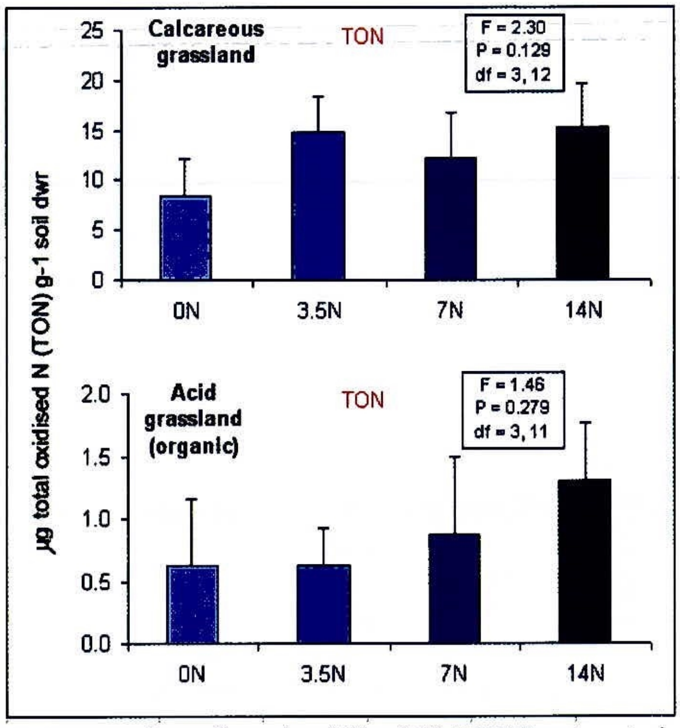

Figure 3. The concentration of total oxidized N (TON) extracted with 2M KCl from calcareous and acid grassland (organic horizon) soil at Wardlow Hay-Cop, four years (April 2006) after ceasing simulated pollutant N deposition inputs at rates of 0, 3.5, 7 and 14 g N m^2 y^{-1} for more than a decade. Vertical bars are SEM, n = 4.

Our analysis of exchangeable soil N pools provides encouraging evidence that semi-natural grasslands that have experienced extreme N saturation can quickly recover towards the concentrations seen in soil receiving much lower N inputs. The rapidity and extent of recovery we have seen indicates that soil N availability is unlikely to present a major constraint on floristic and overall habitat recovery. We must add one note of caution. Our experimental plots are a small area in a large grazed field, which is likely to result in progressive export of N from our plots, whereas in grasslands that have experienced N enrichment at the landscape scale a greater proportional return of N will occur in dung and urine.

3.2 Base cation depletion of N saturated grassland?

We have used low molecular weight organic acids, of the kinds released by plant roots, as extractants to determine the extent of depletion of 'plant available' base cation pools in the grassland soils by the acidifying effects of simulated pollutant N. Plant roots secrete low molecular weight carboxylic acids as a response to phosphorus and iron limitation, particularly in Ca rich soils, and as a mechanism to protect against aluminium and iron toxicity in acid mineral soils (Tyler & Ström, 1995). We have shown previously in these grasslands that P is the critical nutrient limiting plant growth, and that this limitation has been exacerbated by the N treatments (Phoenix *et al.*, 2003b). Citric acid is particularly important for limestone grassland plants, whereas oxalic acid is more important for acid grassland species (Tyler & Ström, 1995). The use of organic acid extracts has not only demonstrated significant base depletion from soils with the highest N loadings, but has confirmed that this base depletion affects the biologically important fraction of the cations that are mobilized by carboxylic acid secretions by roots (Fig. 4).

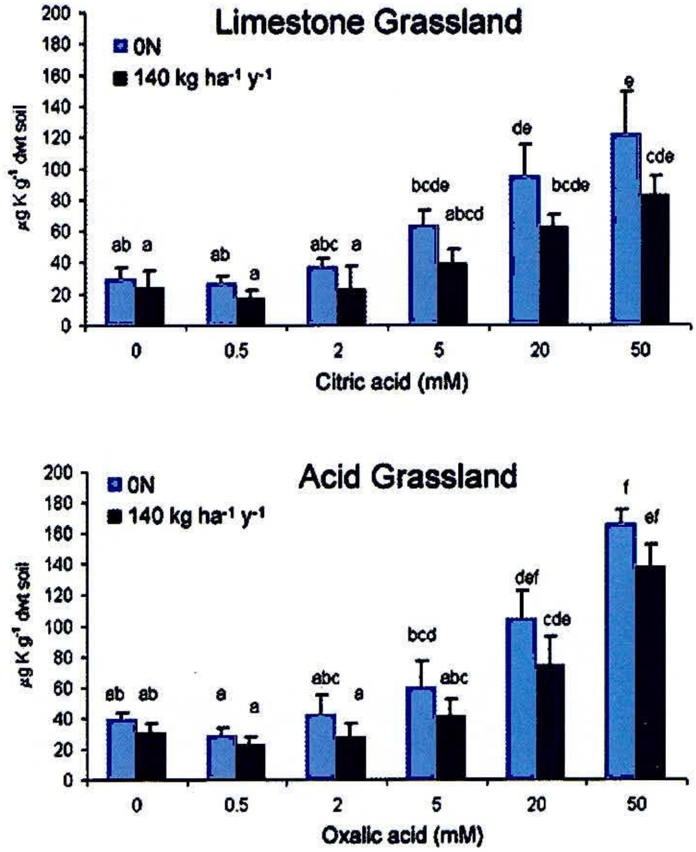

Figure 4.
The concentration of potassium (K) extracted by a range of concentrations of citric acid and oxalic acid added to limestone grassland and acid grassland soil sampled from control plots or plots that received 10 years of simulated pollutant N deposition (1995-2005) at a rate of 14 g N m^2 y^{-1} (140 Kg N ha^{-1}).

The concentration of the organic acids strongly affects the proportion of the cation pool that is extracted. After initial trials (Fig. 4) we have routinely used organic acids at 2mM organic acids as this is within the concentration range that can occur in the rhizosphere and this concentration showed the greatest relative change in cation concentrations caused by the N deposition treatments.

Extraction of the limestone grassland soil with 2mM citric acid has revealed a 28-30% decrease in 'plant available' calcium and a 30-36% decrease in magnesium extracted from the plots receiving 3.5 and 14 g N m^2 y^{-1}. Extraction of the acid grassland soil with 2mM oxalic acid has revealed no effects on calcium but 62-65% loss of 'plant available' potassium confirming the loss of potassium seen in both grasslands in the extractions with citric acid (Fig. 4).

The loss of base cations from soil caused by pollutant N not only reduces the buffering capacity of the soil against further acidification, but as many of the cations are essential plant nutrients the soil fertility may decline, with adverse effects on species that are best adapted to more plentiful supplies of such elements. These effects are likely to be long lasting and may be a serious constraint to recovery. The extent to which base depletion has occurred in upland grassland in response to acid deposition and exceedence of N critical loads requires further investigation. We are currently investigating the extent of loss of buffering capacity.

3.3 Floristic changes- prospects for recovery on reducing N deposition?

One of the most important impacts of N deposition is reduced flowering of forb species in both acid and calcareous grassland. Many forbs show proportionally greater reductions in flowering than in cover in response to high N deposition- so that flowering is a particularly sensitive indicator of N impacts. The flowering of forbs is a key component of the amenity value of semi-natural grasslands. Reduced flowering seriously diminishes the biological and aesthetic value of species-rich grassland. Due to the particular sensitivity of flower production to N deposition we have carried out detailed studies with the aim of being able to detect any early evidence of recovery on ceasing experimental N additions to the grasslands. In the first year of the split 'treatment' and 'recovery' plots there was no evidence of recovery (Fig. 5).

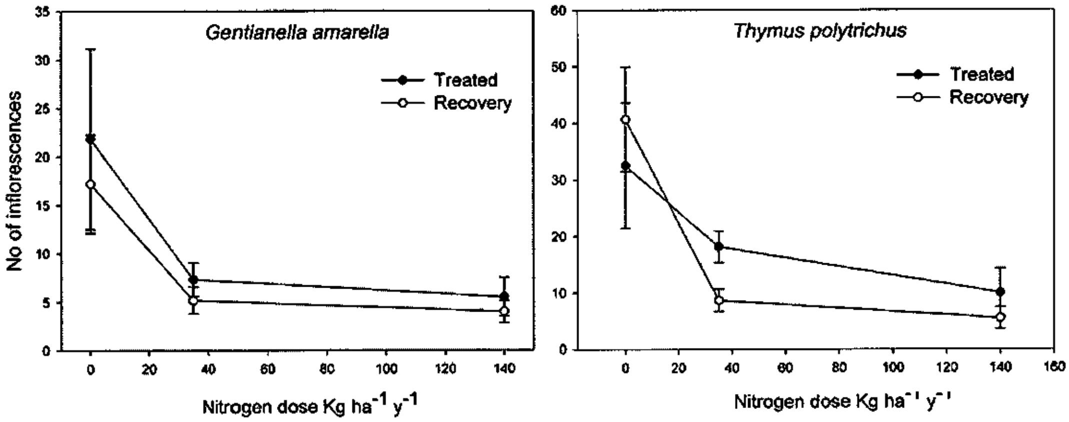

Figure 5. The effects of simulated N deposition on the number of inflorescences of *Gentianella amarella* and *Thymus polytrichus* in August 2005 in the limestone grassland plots one half of each have not received experimental N additions since the end of 2004. There is no significant effect of 'recovery' in either species, but there are significant effects of N treatment (ANOVA, $P<0.05$ in both cases). Vertical bars are SEM, n = 3.

Analysis of flowering of *Anemone nemorosa* in May/June 2005 and 2006 has also shown highly significant effects of the N treatments reducing flower production, but even using pair-wise analysis comparing differences between the treated and untreated sides of the plots in the two years we have found no evidence of recovery. *Conopodium majus* has also suffered a severe decline in the acid grassland plots. It shows no sign of recovery in the second year after ceasing treatments but its growth is greatly stimulated by P additions, and these alleviate the effects of N deposition. This species appears to be highly phosphorus limited and the previously reported effects of n enrichment decreasing P availability to plants (Phoenix *et al.*, 2003b) appears to be particularly important for this species.

Further studies are planned this summer to monitor flower production and abundance of key species in the plots.

4. Collaboration with Dynamic Modelling Umbrella and Freshwater Umbrella

Our first joint meeting with Dynamic Modelling and Freshwater Umbrella groups took place in October 2005 and identified opportunities for further collaboration. The findings of our soil chemistry studies are of direct relevance to these groups and we anticipate our interactions will be strengthened. Results from the current project were presented at the annual Umbrella project meeting in October 2005, and more recently at the CAPER conference.

5. Key Findings

- The plant-available N concentrations in the limestone grassland soil have recovered quickly from over a decade of N saturation. However, this soil has suffered extensive depletion of bases- over 30% of the 'plant available' pool of Ca, Mg and K has been depleted. These effects are likely to be highly persistent and are likely to impair recovery of species that have declined due to the N treatments.
- The acid grassland has also experienced significant loss of case cations and continues to have elevated concentrations of mineral N in the N treated plots.
- There is no evidence to date from the plots divided in 2004 that species whose flowering is severely reduced by N saturation are recovering after reducing N deposition to ambient levels for 1-2 years.
- Recovery of grasslands from N saturation is likely to be slow and may require careful management interventions such as small additions of base cations or seeds to re-establish plants that have declined.

6. References

Carroll JA, Caporn SJM, Johnson D, Morecroft MD, Lee JA 2003. The interactions between plant growth, vegetation structure and soil processes in semi-natural acidic and calcareous grasslands receiving long-term inputs of simulated pollutant nitrogen deposition. *Environmental Pollution* **121:** 363-376.

Fowler D, O'Donoghue M, Muller, JBA, Smith RI, Dragosits U, Skiba U, Sutton MA, Brimblecombe P, 2004. A Chronology of Nitrogen Deposition in the UK Between 1900 and 2000. *Water, Air, & Soil Pollution: Focus*: **4:** (6) 9 – 23.

Phoenix GK, Booth RE, Leake JR, Read DJ, Grime JP, Lee JA, 2003a. Effects of enhanced nitrogen deposition and phosphorus limitation on nitrogen budgets of semi-natural grasslands. *Global Change Biology* **9**: 1309-1321.

Phoenix GK, Booth RE, Leake JR, Read DJ, Grime JP, Lee JA, 2003b. Simulated pollutant nitrogen deposition increases P demand and enhances root-surface phosphatase activities of three plant functional types in a limestone grassland. *New Phytologist* **161**: 279-289.

Preston CD, Telfer MG, Arnold HR, Carey PD, Cooper JM, Dines TD, Hill MO, Pearman DA, Roy DB, Smart SM. 2002. The Changing Flora of the UK. DEFRA http://www.defra.gov.uk/wildlifecountryside/ewd/flora/index.htm

Rich TCG, Woodruff ER, 1996. Changes in the vascular plant floras of England and Scotland between 1930-1960 and 1987-1988: the BSBI monitoring scheme. Biological Conservation 75: 217-229.

Smart SM, Ashmore MR, Hornung M, Scott WA, Fowler DA, Dragosits U, Howard DC, Sutton MA, Famulari D, 2004. Detecting the signal of atmospheric N deposition in recent national-scale vegetation change across Britain. *Water Air and Soil Pollution: Focus* **4**: 269-278.

Stevens CJ, Dise NB, Mountford JO, Gowing DJ, 2004. Impact of nitrogen deposition on the species richness of grasslands. *Science* **303:** 1876-1879.

Tyler G, & Ström L, 1995. Differing organic acid exudation pattern explains calcifuge and acidifuge behaviour of plants. *Annals of Botany* **75:** 75-78.

Work Package 2: Impacts, Recovery and Processes

Task 4: Long-term impacts of enhanced and reduced nitrogen deposition on semi-natural vegetation

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Manchester Metropolitan University

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1. Summary

Key objectives of the research in 2004 - 2007 are to evaluate potential chemical and biological impacts of nitrogen deposition on moorland and heathland plant and soil systems, with detailed investigation of sensitive species (mosses and lichens); recovery from N loading; interactions of nitrogen (N) supply with management and phosphorus (P); and the processes controlling N leaching. These are being addressed by studies of plants and soils on the long-term nitrogen manipulation plots at the Ruabon moorland, and by a national survey of nitrogen indicators in similar *Calluna* (heather) - dominated communities.

The original Ruabon plots established in 1989 were burnt in April 2000 and regeneration of the *Calluna* canopy showed an initial negative effect of nitrogen treatment on height and cover together with increased moss cover, due in part to the proliferation of bare ground colonizers. Data from the past two years however, has shown comparable re-growth of *Calluna* at all N levels.

The new Ruabon plots, established in 1998 (N and P treatment combinations) continue to show a positive growth response to nitrogen, with some indication of N/P co-limitation at the high N + P treatments, and high levels of moss and lichen cover in response to phosphorus treatment. In 2005, a significant drop in exchangeable N in the surface peat and litter layers was found two years after a reduction in N treatment to half of the new plots at Ruabon, and lower moss and *Calluna* tissue N levels were also recorded.

On the lowland heath experiment at Budworth, established in 1996 and subjected to a management cut in October 2002, a strong negative effect of nitrogen on re-growth of the *Calluna* canopy was found at the highest N treatment. Previous research found low rates of N leaching except for during a period of heavy late summer rain in 2004. This indicated that large reserves of mobile N exist in the high N plots and these have recently been confirmed by measurement of exchangeable N pools in surface soil horizons.

A regional survey of nitrogen indicators in *Calluna* moorland from sites in Scotland, Wales and the Peak District has investigated a range of plant and soil N and P measurements and biodiversity. The exchangeable N in surface litter layers was closely related to N deposition and the consequence of this elevated pool of available nitrogen may be to reduce the diversity of the bryophyte community within the *Calluna* canopy.

2. Policy Relevance

The work on the long-term nitrogen manipulation plots at Budworth and Ruabon, and the wider scale survey work addresses the general questions below:

- What is the scale and extent of change caused by nitrogen deposition?
- How can we apply this knowledge to support and refine critical loads?
- What is the potential scale and speed of improvements after reduction in nitrogen deposition?
- What is the influence of management on nitrogen impacts and critical loads?

Long – term data sets spanning up to 17 years, show a pattern of increased total and exchangeable nitrogen content in soil, litter and vegetation, with deleterious effect on sensitive species such as bryophytes and lichens. The data now emerging from the regional survey, also suggests a strong link between N deposition, bryophyte diversity and exchangeable N in litter, with sudden increases in the exchangeable NH_4^+ content of litter in particular at N deposition levels above 25 kg $h^{-1}y^{-1}$. The combination of regional survey work, with possible nitrogen bio-indicators validated against the long-term manipulation experiments, should allow us to improve our understanding of the factors most relevant to the setting of critical loads.

High levels of nitrogen application on the plots have led to long-term accumulation in litter and peat/soil layers, with low overall leaching losses. There are however also clear signs of early recovery with reduced N levels in vegetation and litter layers, following only a few years of reduced N loading, most marked at highest rates of input. Recovery effects on vegetation could therefore be fairly rapid, but the nitrogen build-up in the deeper soil layers suggest that other aspects of recovery are likely to be much slower.

There was a strong interaction between nitrogen levels and management with reduced regrowth in the high N plots from both sites. A cycle of more frequent management has the clear potential to speed recovery, allowing nitrogen offtake to sensitive areas, and reducing deleterious N effects on regrowth and the possibility of invasion by competing species.

3. Project Update

3.1 Ruabon old plots:

The original plots, first established in 1989, have been treated monthly with 0 – 120 kg N ha^{-1} y^{-1} as ammonium nitrate. A controlled management burn in March 2000, removed the *Calluna* canopy, but left litter and peat layers intact. Since the burn, a steady increase has been seen in *Calluna* canopy height in the control plots over the last five years, together with a negative effect of nitrogen treatment from 2000 to 2003. Survey data for 2004 and 2005 however, has shown strong re-growth in all the plots, with a reduced effect of nitrogen (Figure 1).

The annual measurements of shoot extension showed high growth rates, and a strong positive nitrogen response for a number of years early in the experiment (1990 – 1993), and again in 2003. These responses are likely to be linked to the growth cycle of the

Calluna, with strongly N limited growth during the pioneer and building phases. Unlike height, there was no indication of a negative effect of nitrogen on shoot extension either following the burn or in the years immediately preceding it. In 2005, the N treatments stimulated shoot extension (Figure 2).

The total moss cover was also much lower in the high N treated plots than the low N controls in the first year following the burn. In 2002 – 2004 however, there was a trend towards higher bryophyte cover in the high N plots due to colonization by N tolerant species typical of bare and burnt moorland (see DEFRA report 2005). The 2005 survey data shows no clear effect of N on total moss touches suggesting a gradual transition to species more typical of building phase *Calluna* canopy.

3.2 Ruabon new plots:

The new Ruabon plots have enabled us to study different aspects – specifically interactions between N and P treatments, and the consequences of lowered nitrogen inputs. Lichens and bryophytes continue to be the most sensitive vegetation to nutrient treatments and a detailed bryophyte survey of the plots was performed in summer 2005 (data still in analysis).

As shown earlier (see Defra report 2004) the lower plants have continued to respond positively to P supply which has compensated for the negative effects of nitrogen. The structure and competitive interactions within semi-natural terrestrial ecosystems are strongly controlled by the availability of limiting nutrients. *Calluna* dominated moorlands are often N limited, and a shift to P limitation can indicate N saturation. The addition of P to experimental areas, in combination with N, allows the level of input or accumulation at which this shift occurs to be identified. Data from the new plots at Ruabon show a shift to P limitation in the growth of the lichens and bryophytes at very low levels of nitrogen input at which *Calluna* growth is still nitrogen limited, thus altering the balance between lower and higher plants.

To investigate potential recovery from future reductions in N deposition, an experiment started in March 2003 on the new plots at Ruabon. The plots started in 1998 were split into two 1 x 2 m^2 plots with one side receiving no further treatment. By 2005 there was fall in exchangeable ammonium in the litter and surface peat horizons on the untreated sub-plots at the highest N level (Figure 3). This decline in labile N could signal a reduction in eutrophication in the surface soils and this may benefit other organisms such as bryophytes and lichens. However, no other consistent responses to the change in N supply have yet been observed.

3.3 Lowland heath plots at Budworth:

The experimental plots at Budworth (1 x 2 m^2) established in March 1996, have been treated monthly with ammonium nitrate (0 – 120kg N $ha^{-1}y^{-1}$). In the autumn of 2002 the plots were subjected to a management cut, and the vegetation removed down to a height of 10 cm. There was a strong negative effect of nitrogen treatment on *Calluna* re-growth following the management cut in the autumn of 2002, with very little regeneration on the high N treated plots. At the same time grass (*Deschampsia flexuosa*) was increased by N treatment. The high *Hypnum jutlandicum* cover on the plots was also clearly reduced in the high N treatments, with large areas of dead moss. These responses were maintained in 2005.

Soil solution collected at 10 cm depth during 2002 – 2003 showed only small positive increases due to N treatment, but much higher losses were seen during a period of heavy rain in late summer 2004 with a clear treatment effect (see DEFRA report, 2005). This episodic event indicated that large reserves of mobile N exist in the high N plots after ten years treatment and these have recently been confirmed by measurement of exchangeable N pools in surface soil horizons (Figure 4) showing a strong treatment response with 5 to 10 fold increases in litter layer NH^{4+} from the high N plots, in comparison with the controls.

3.4 Regional surveys:

Collections of *Calluna* litter were initially made from four Peak district and five Welsh sites in a pilot study carried out in February 2004, and the samples assayed for total N and P content, and litter phosphatase activity. Phosphatase activity was strongly correlated with total P content and N/P ratio, across all the sites (see DEFRA report 2005), suggesting wide differences in the balance of N/P demand. Two further surveys in June 2004 and April 2005 included a wider range of sites from Wales, Peak District and Scotland. Measurements included phenol oxidase activity, phosphatase activity, total N and P, KCl extractable ammonium and nitrate, and total metal content. Further surveys in late summer 2005 measured total bryophyte biodiversity, and *Calluna* and *Hypnum jutlandicum* total N on sites from Wales and Peak district.

Much of this data is still undergoing statistical investigation, but a number of clear indicators of nitrogen deposition have emerged on the basis of preliminary analysis. The pools of exchangeable ammonium in surface litter were strongly related to N inputs and suggested a threshold deposition of 20-25 kg N ha^{-1} y^{-1} above which exchangeable ammonium rose dramatically (in the Peak District sites, Figure 5). This large change in available N in surface soils could affect many processes and organisms in this component of the soil-plant system. Surveys of bryophytes at the same sites found a negative relationship with N deposition, and these plants may be directly affected by the nitrogen in the litter layer within which they grow.

4. Collaboration with Dynamic Modelling Umbrella and Freshwater Umbrella

Soil C/N and leachate data from the Ruabon and Budworth plots have formed an integral part of the development and evaluation of the MAGIC soil dynamic model. (Evans et al DEFRA 2004), (Evans *et al* 2006 a), extending the use of the C/N ratio as an indicator of soil susceptibility to enhanced nitrogen leaching.

The testing of MAGIC simulations against the nitrogen manipulation sites has also led to an adapted version of the model, incorporating enhanced carbon sequestration in response to nitrogen addition (Evans *et al* 2006 b), which successfully simulates conditions at both sites and raises important questions concerning the carbon balance of nitrogen polluted moorlands.

N addition treatments are likely to change the plant species composition of the experimental plots. Predicting these changes is the concern of the soil/vegetation modelling group, which use a soil process model (MAGIC) linked to empirical niche models for plant species (GBMOVE). To add realism and local accuracy, the MAGIC model can be parameterised using local soil chemistry data as has already been done at Budworth and Ruabon. However, adding realism to predicted plant species changes includes, among other things, taking account of the composition of the local species pool. A method has been developed to define local pools around modelled sites. The list includes species most likely to be abundant around, and most able to disperse into a site. The selection process uses existing GB-wide datasets (BRC 10km square species lists and Land Cover Map 2000) and therefore requires validation at the site level. To this end a joint visit was made to Budworth where a) constraints on species change in experimental plots was discussed in light of the surrounding vegetation composition, b) species pool membership defined using the above datasets was validated based on observations of the abundance and adjacency of habitats on the ground compared to LCM2000."

Testing of the SMART/SUMO successional model at Budworth Common has also raised the question as to whether the constraints on possible invading species at both nitrogen manipulation sites, are related to dispersal, or to climatic or soil related factors, which we hope to investigate further.

5 Key findings

5.1 Early effects of reduced nitrogen loading

Previous work on the Ruabon moorland plots has found large accumulations of N in peat and soil layers, with relatively low levels of leaching and only a moderate positive growth response, suggesting that the accumulated soil N would be difficult to remove and recovery of the system could be slow. However, the recovery experiment on the new plots at Ruabon, suggests that in more rapidly cycling soil/plant compartments, available N (nitrate and ammonium) may decline much more quickly in response to reduced deposition. It follows from this that beneficial effects could be seen on the vegetation structure of polluted moorlands in the first few years of reduced nitrogen loading.

5.2 Need for more frequent management

The post-management re-growth of heather from the bases of the stems was slowed in the higher N treatments, following management, on both the moorland and lowland heath plots. Since nitrogen treated heather was further developed, the retarded re-growth after management is probably an indirect result of nitrogen-enhanced ageing of heather. Slowed regeneration of heather may allow invasion and establishment of competitor plant species – such as *Deschampsia flexuosa* - as found on the lowland heath experiment. More frequent management would therefore probably benefit heather-dominated communities on nitrogen affected upland and lowland heaths.

5.3 N/P limitation affects species differently

Critical loads for N with respect to bryophytes and lichens are potentially altered by availability of phosphorus. *Calluna* dominated moorlands are often N limited, and a shift to P limitation may indicate N saturation. Data from the new plots at Ruabon show a shift to P limitation in the growth of the lichens and bryophytes at very low levels of nitrogen input at which *Calluna* growth is still strongly nitrogen limited, thus altering the balance between lower and higher plants.

5.4 Mosses and liverworts as regional bio-indicators of N impacts

In the Ruabon experiment, we have identified potential biomonitors of nitrogen eutrophication at a UK scale. Recent regional surveys indicate that exchangeable nitrogen in *Calluna* litter is closely related to N deposition and could offer a simple measure of eutrophication status. A consequence of eutrophication in upland heath appears to be a decline in moss, liverworts and lichens.

5.5 Future research in heathlands is required to understand:

- Long term changes in plant communities in response to changing nitrogen deposition.
- The influence of nitrogen on heather re-growth after management burning or cutting.
- Consequence of reduced nitrogen deposition for lower plants and nutrient cycling in surface soil layers.
- Interactions between nitrogen and phosphorus supply and impacts for bryophytes, lichens and vascular plants.
- Scaling up – how we can use knowledge from field experiments to detect responses to changing nitrogen deposition in heathlands at the UK scale.

6 References

Evans CD, Reynolds B, Jenkins A, Helliwell RC, Curtis CJ, Ferrier FC, Goodale CL, Emmett BA, Pilkington M, Caporn SJM, Carroll JA, Norris D, Davies J, Coull MC. 2006a. Evidence that soil carbon pool determines susceptibility of semi-natural ecosystems to nitrogen saturation. *Ecosystems* 9 453-462.

Evans CD, Pilkington MG, Caporn SJM, Wilson DB, Carroll JA. 2006b. Modelling nitrogen saturation and carbon accumulation in moorland soils in response to nitrogen deposition. *Environmental Pollution* in press

Pilkington MG, Caporn SJM, Carroll JA, Cresswell N, Lee JA, Ashenden TW, Brittain SA, Reynolds B, Emmett BA. 2005. Effects of increased deposition of atmospheric nitrogen on an upland moor:Leaching of N species and soil solution chemistry. *Environmental Pollution* 135 29 – 40.

Pilkington MG, Caporn SJM, Carroll JA, Cresswell N, Lee JA, Reynolds B, Emmett BA. 2005. Effects of increased deposition of atmospheric nitrogen on an upland moor: Nitrogen budgets and nutrient accumulation. *Environmental Pollution* 138 474 -485.

Pilkington MG, Caporn SJM, Carroll JA, Cresswell N, Lee JA Emmett BA. Johnson D. 2005. Effects of increased deposition of atmospheric nitrogen on an upland *Calluna* moor:N and P transformations. *Environmental Pollution* 138 474 -485.

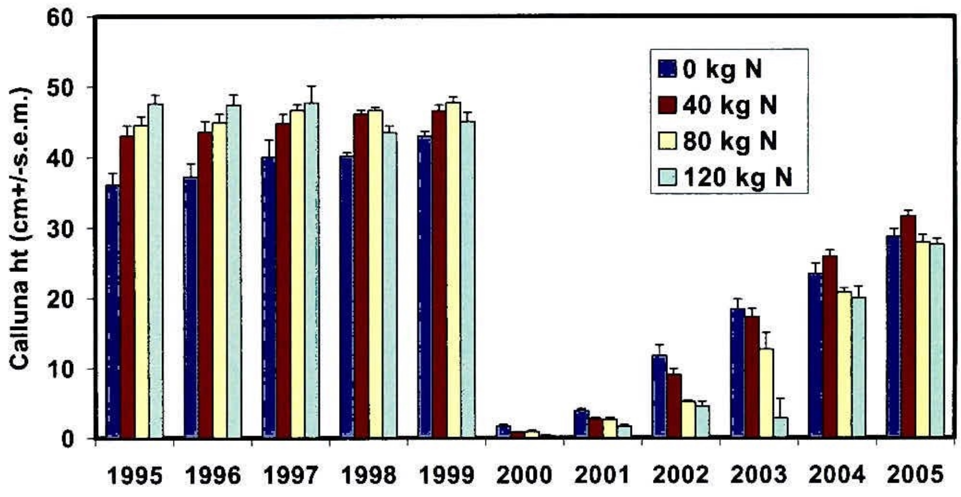

Figure 1. Annual *Calluna* canopy height on the Ruabon old plots, treated since 1989 and burnt in 2000.

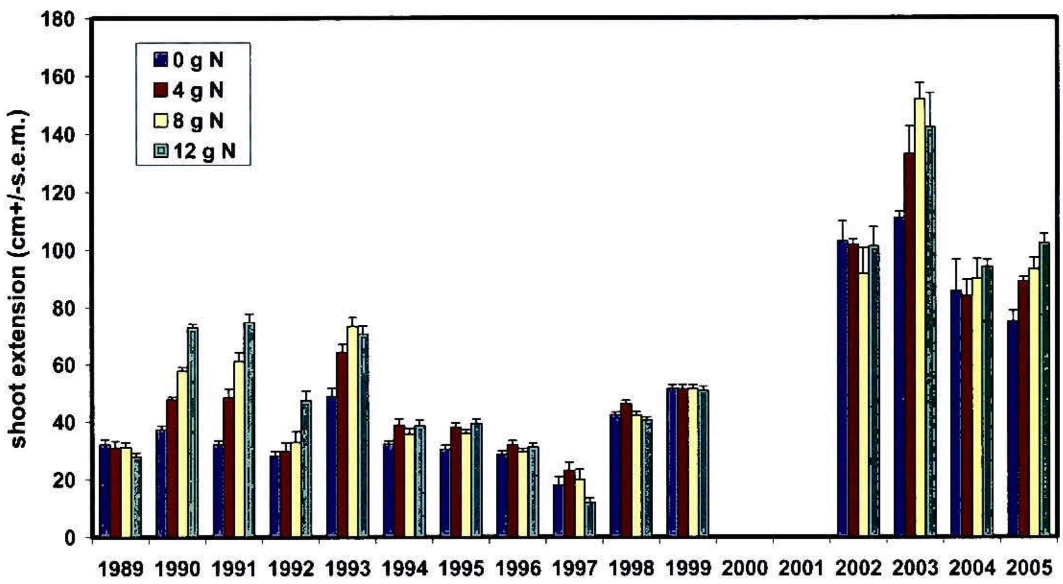

Figure 2. Annual *Calluna* shoot extension on the Ruabon old plots, treated since 1989. Little regrowth to record for 2000 and 2001.

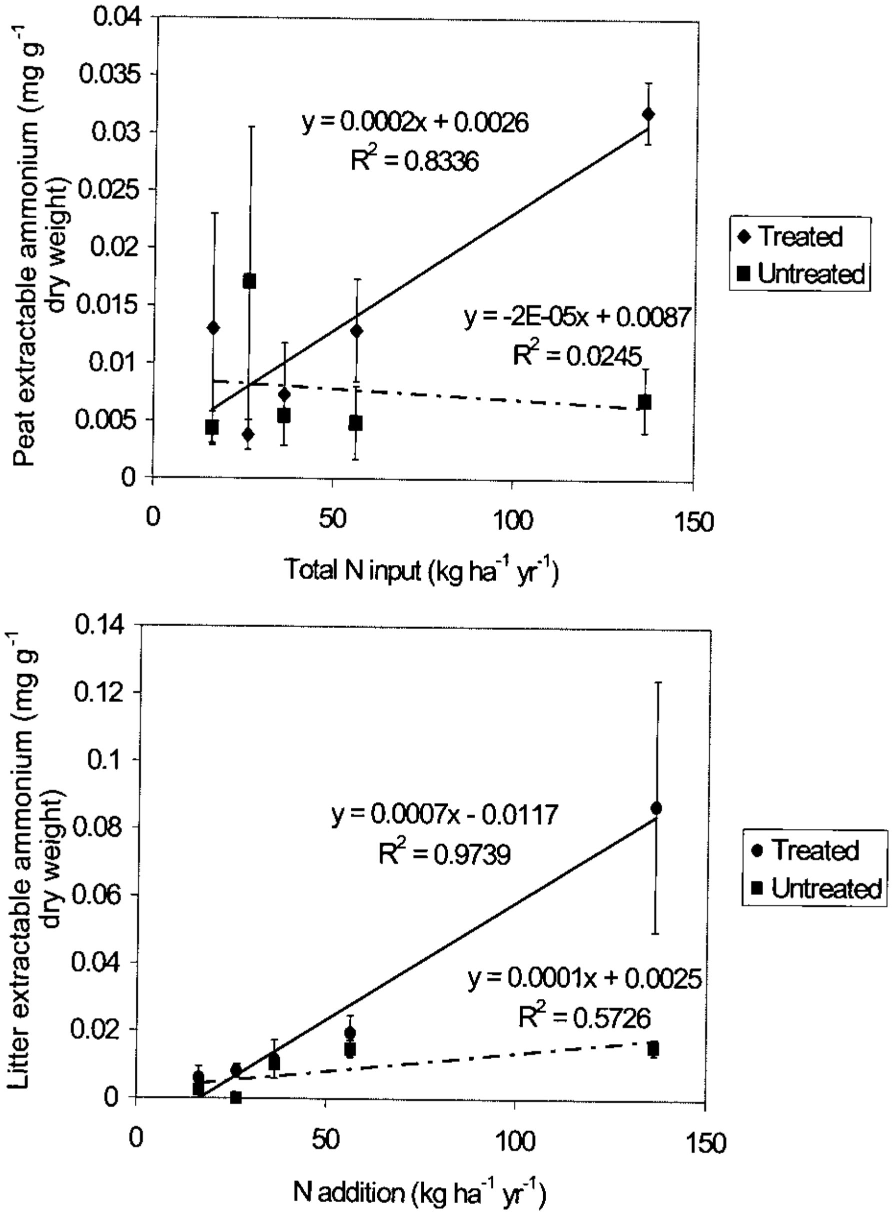

Figure 3. Extractable NH_4^+ in surface litter and peat horizons in the Ruabon New Plots in Spring 2005: Effect of the recovery experiment. Treated plots have received N since 1998. Addition to untreated plots in spring 2003.

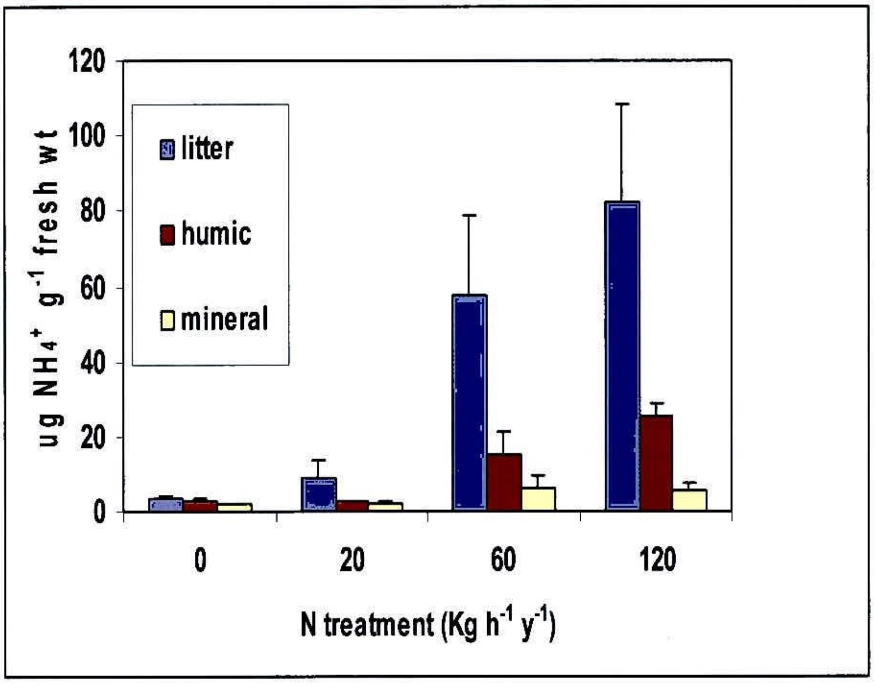

Figure 4. KCl-extractable NH_4^+ in cores from the Budworth lowland heath experiment in January 2006 after 10 years of treatment. Litter taken as top 1.5 cm, humic as 1.5 – 4.0 cm, mineral as 4.0-6.5 cm.

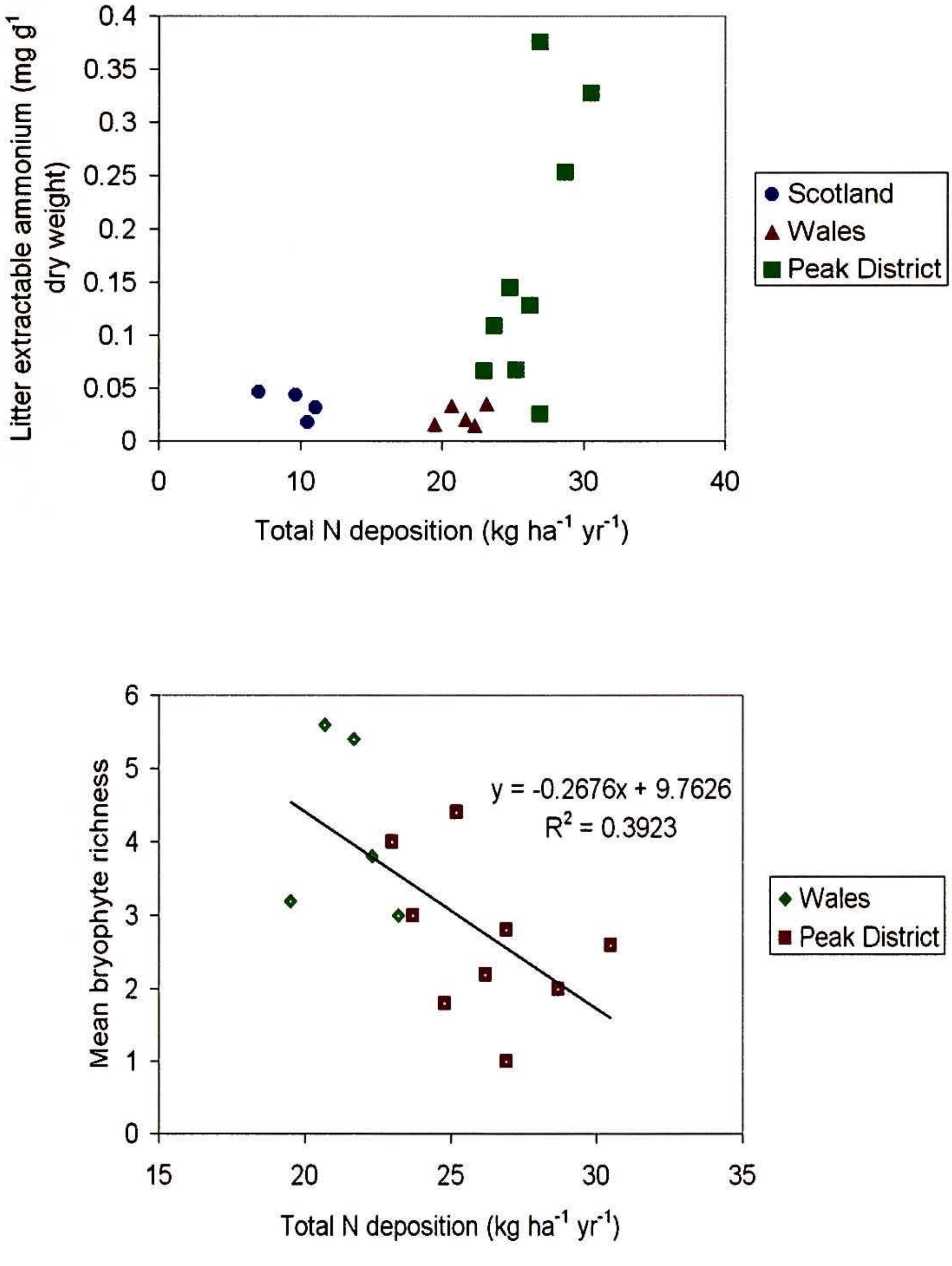

Figure 5. Exchangeable ammonium in leaf litter (above) and bryophyte species richness (below) in *Calluna* canopies of upland heath surveyed in northern Britain in 2005 in relation to modelled (APIS) nitrogen deposition.

Work Package 2:
Impacts, Recovery and Processes

Task 5:
Impacts of nitrogen deposition on lowland heathland

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1. Summary

Nitrogen addition to experimental plots at Thursley Common NNR continues to have large and significant effects on the growth, canopy development and phenology of *Calluna*. The management treatments carried out in 1998 no longer have differing effects on shoot growth or phenology. However, there are persistent effects of management on *Calluna* canopy cover and the associated bryophyte flora.

Nine years after the cessation of nutrient additions, there is still clear evidence of former N treatments on the *Calluna* canopy, as well as effects on flowering and bud burst. This confirms that ecosystem recovery from eutrophication is a slow process, even when initial nutrient loadings are relatively low, and highlights the need for long term experiments to monitor these responses.

Experimental addition of N and P to P-limited heathland mesocosms has demonstrated a strong growth and phenological response to P (confirming P limitation), and a negative effect of N addition on shoot length and drought damage. This is the first evidence to emerge which suggests that elevated N deposition can have both direct and indirect detrimental effects on P-limited systems, with implications for the role of P-limitation as a potential modifier of critical loads.

A survey of 33 lowland heaths in southern and eastern England has provided information on the extent of variation in soil and plant nutrient status, and the relationships between these. Foliar N:P ratios were found to range from 8.6-17.9; nearly half of the sample sites had ratios above 14, indicating either NP co-limitation, or P limitation. Relationships between soil and foliar chemistry, and N:P ratios, were variable. However, there was an indication that soil PME activity was higher, and P sorption capacity lower at sites where N:P ratios indicate P limitation. Data also show that soil N and P contents increase with stand age, highlighting the importance of controlling for such factors in wide scale surveys of this nature.

2. Policy Relevance

Critical loads and their exceedance underpin UK pollution control policy, and require continuous updating and refinement to evaluate the impact of emissions reductions on biodiversity and ecosystem services. The long term manipulation experiments at Thursley show that exceedance of N critical loads results in large and significant effects on the performance of higher and lower plants, and the microbial community. Visible changes in vegetation are now apparent, providing a visual demonstration of the short to medium term consequences of critical load exceedance in lowland heathland. The capacity for heathland systems to act as sinks for atmospheric N has been clearly demonstrated; information on the magnitude of this sink, and timescales over which this may change, can be used to improve models of national and regional N cycling, and thus contribute to policy decisions on emissions targets.

The (short term) value of high intensity managements to reduce nutrient accumulation and offset the effects of N deposition has been shown. However, the high levels of physical disturbance associated with intense forms of management result in persistent effects on heathland vegetation which must be taken in to account, alongside simple evaluation of nutrient budgets, when evaluating policy options for the protection of semi-natural ecosystems.

The long term persistence of significant effects of former N additions on plant and microbial communities indicates that heathland recovery from eutrophication will be a slow process. This is particularly so given that earlier inputs were very low (7.7 / 15.4 kg N ha^{-1} yr^{-1}), plots had been managed when treatments ceased, and effects are still significant more than nine years on. The potential for, and speed of ecosystem recovery are important issues for evaluating the consequences of recent (and potential new) policies aimed at reducing emissions of nitrogenous pollutants. Furthermore, the recovery study at Thursley continues to provide strong support for the low end of the range for heathland critical loads.

Newly emerging evidence of detrimental effects of N deposition on P-limited heathlands highlights a need for careful consideration of the role of P limitation as a modifier of N critical loads. The informal use of modifiers, such as perceived P limitation, to assign site-relevant values from within the broad critical load ranges is widespread. However, our work is now showing that effects of N on these systems may, in fact, be as great as, or greater than, in N-limited heathlands. If this is a widespread phenomenon, it will have important implications for heathland critical loads and, potentially, also for other ecosystems.

3. Project Update

The second year of this project has involved a combination of manipulation experiments and sample collection from a wide range of lowland heaths, encompassing all of the major heathland areas in southern and eastern England. Data obtained following detailed field and laboratory analyses have allowed us to make good progress towards answering the following questions:

- What are the long term consequences of prolonged inputs of N on heathland ecosystems and how important is management as a modifier of responses?
- Is recovery from the effects of earlier nutrient loading possible and, if so, over what timescale?
- How do P-limited heathlands respond to increases in N deposition?
- What is the extent of N and P limitation in UK lowland heathlands?

Over the past 12 months, we have assessed higher and lower plant response in both the ongoing nitrogen-management experiment and the recovery study, at Thursley Common. The former experiment continues to demonstrate large growth responses to N addition and evidence of prolonged management-related differences in vegetation cover for both higher and lower plants. The recovery experiment is now entering its 10^{th} year following the cessation of experimental N inputs and continues to demonstrate a significant legacy of earlier treatments. In addition, we have continued the nutrient manipulation experiment on P-limited heathland mesocosms, which was introduced last year. A major thrust of the past year's work has, however, been to carry out a field survey of more than 30 lowland heaths in southern and eastern England. The broad aims of this survey were to: quantify the variation in plant and soil chemistry across a representative range of lowland heathlands; determine the relationships between plant P limitation (as indicated by foliar N:P ratios), soil PME activity and different indices of soil P availability; and quantify the effect of stand age on nutrient characteristics.

4. Collaboration with Dynamic Modelling Umbrella and Freshwater Umbrella

Monthly leachate samples have been collected from control and +N plots, using suction lysimeters. These have been analysed for cations, anions and DON, to provide data on soil solution chemistry for model parameterisation. Soil samples have also been provided, for C/N analysis. These, and other plant chemistry data, will allow the MAGIC model to be calibrated for the experiments at Thursley for the first time. It is anticipated that this will allow past and future changes in soil acidity and N status to be simulated and the likelihood of species changes to be predicted, using the GB-MOVE statistical model.

5. Key Findings

5.1. Long term effects of N addition on a lowland heathland, and interactions with management

Nitrogen additions (30 kg ha^{-1} yr^{-1}) have continued to have significant effects on *Calluna* shoot growth ($P<0.01$), canopy development (height $P<0.001$, density, $p<0.001$), flowering ($P<0.05$) and phenology (bud burst $P<0.001$). Plant growth was generally lower in 2005 than in previous years (including during the 2003/4 drought period, reported last year); mean shoot lengths were 10.6 mm and 14.4 mm in control and +N plots respectively, compared to values of 21.0 mm and 34.4 mm averaged between 2001-2004. Evidence of treatment-related effects on drought injury in 2004 were no longer apparent; indeed the proportion of dead leading shoots was actually lower in +N plots in 2005 (6.8%, compared to 12.4% in controls), suggesting that compensatory growth by side shoots (taking over a leading role) may also be greater in plants receiving additional N.

The height of the *Calluna* canopy is now 60% greater in +N plots, compared to controls, representing the substantial, cumulative effects of eight years of N additions. The greater density ($P<0.001$) and percent cover ($P<0.001$) of the *Calluna* canopy in the N treated plots continues to be associated with a substantial reduction in the percent cover of lichens ($P<0.001$). Interestingly, lichen cover showed a significant interaction between N and management treatments ($P<0.01$), with a lower proportional effect of N in the high temperature burn plots.

The only significant effects of management treatments apparent in 2005 were on the percent cover of *Calluna* ($P<0.01$) and bryophytes ($P<0.01$). The lower *Calluna* cover in high temperature burn plots evident immediately after management has persisted, with values of 79.5%, compared to 89.8% in the low intensity burn treatment, in 1995. The bryophyte flora is dominated by *Campylopus introflexus*, which covers 34.5% of the ground area in high temperature burn plots, compared to 11.0% in the low intensity mow. This species is know to be a rapid post-fire colonizer, and it would appear that the *Calluna* canopy is still sufficiently open in the plots which were most intensively managed (and disturbed) in 1998 to allow it to persist at high levels.

Plant growth responses to N additions still indicate strong N limitation at Thursley.

However, in order to assess whether experimental treatments have increased the demand for phosphorus, an assay of soil PME (phosphomonoesterase) activity was carried out in May 2006 (Tabatabai & Bremner, 1969). Figure 1 shows that both N addition ($P<0.001$) and management ($P<0.001$) had a significant effect on enzyme activity; levels were 30.4% higher in N-treated soils compared to controls.

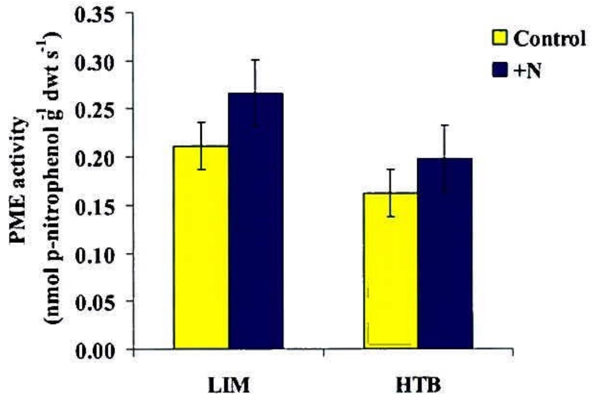

Figure 1. Effects of N addition and management on phosphomonoesterase (PME) activity. LIM=low intensity mow; HTB=high temperature burn.

5.2. Heathland recovery from eutrophication

Effects of earlier N treatments are still apparent for *Calluna* canopy height ($P<0.05$) and percent cover ($P<0.01$). Plants in former low and high N treatments have canopies 22.0 % and 27.4% higher, respectively, than their counterparts in control plots, nine years after treatment additions were suspended. Bud burst in May 2006 was also significantly more advanced in plots which had received earlier N additions, compared to controls ($P<0.01$).

The effects of former N and management treatments on lichen percent cover were not quite statistically significant in October 2005. However, there is some indication that cover was lowest in plots which had the highest former N additions and the lowest management removal of nutrients (Figure 2).

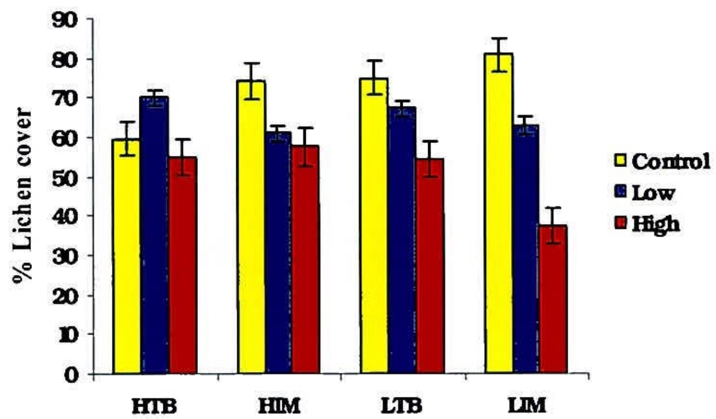

Figure 2. Percent lichen cover in recovery plots, nine years after N additions ceased. HTB – high temperature burn; HIM- high intensity mow; LTB – low temperature burn; LIM – low intensity mow.

5.3. Phosphorus-limited heathland mesocosms

Results from 2005 show a significant effect of P addition on shoot length ($P<0.001$), canopy height ($P<0.001$) and canopy density ($P<0.01$). The effect of N addition overall was not significant, although there was a significant ($P<0.001$) interaction between N and P treatments. Shoot lengths in the cores receiving low N (20 kg ha^{-1} yr^{-1}) and high N (60 kg ha^{-1} yr^{-1}) treatments alone were lower than those in control and P treated cores; plants in the high N+P treatment (NNP) grew less well than those receiving either low N+P (NP) or P only (Figure 3). Similar patterns were found for canopy height and density. In May 2006, bud burst was significantly more advanced in P-treated cores ($P<0.01$), with no effect of N treatment.

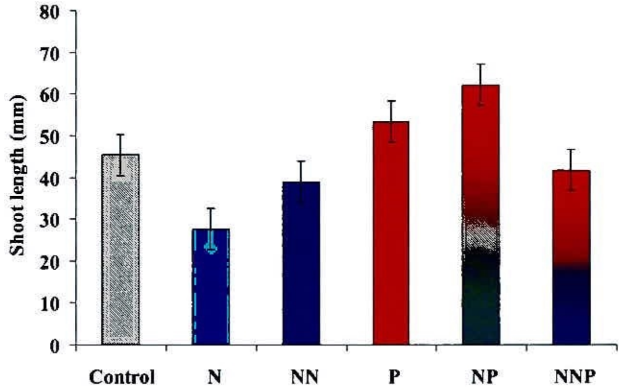

Figure 3. Effect of N and P addition on Calluna shoot length in heathland mesocosms (2005). N and P treatments as follows: N=20kg ha^{-1} yr^{-1}, NN=60 kg ha^{-1} yr^{-1}, P=20 kg ha^{-1} yr^{-1}

These results confirm the primary P-limitation of the mesocosms, and demonstrate a detrimental effect of N addition on *Calluna* growth in P-limited heathland systems. This is supported by data from 2004 showing that drought-related damage was significantly higher in N-treated cores than in either control or P-treated ones (Figure 4).

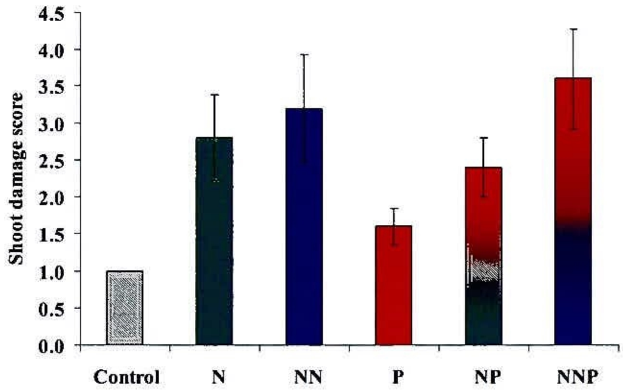

Figure 4. Treatment effects on shoot damage in P-limited heathland mesocosms (2004)

5.4. *Field survey of the nutrient status of lowland heathlands*

In order to provide a context for ongoing N manipulation experiments, a field survey of 33 lowland heathlands was carried out in the summer of 2005. Sites were selected to cover a range of different geologies and, therefore, potential nutrient limitations. Figure 5 shows the location of survey sites in relation to the areas of lowland heathland in England (English Nature, 2006). At each site, soil and plant samples were collected from areas of pioneer phase, building phase and mature phase *Calluna.* Analyses were carried out for the following: plant N and P concentrations, soil PME activity, KCl extractable NH_4, Olsen's extractable PO_4, organic matter content, pH and total soil N and P concentrations. Phosphorus sorption capacity (PSC) was also estimated using the levels of ammonium-oxalate extractable Al and Fe to indicate the number of sorption sites for P (Lookman *et al.*, 1995). Unless otherwise stated, data are presented for building phase (5-15 years) *Calluna* stands.

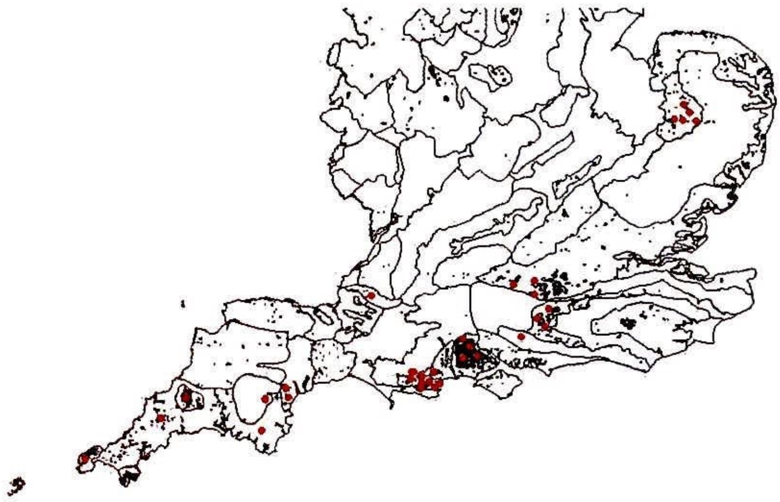

Figure 5. Location of survey sites (red), in relation to lowland heathland areas (black) in southern and eastern England

Initial investigation focussed on a contrast between sites expected to represent low (n=3) and high (n=3) soil P availabilities. This was based on published data for isotopically exchangeable phosphorus (IEP) and phosphorus adsorption capacity (P_{max}) (Chapman *et al.*, 1989). Table 1 summarises the building phase data for contrasting groups of sites. Those sites identified in Chapman's earlier survey as having low soil P availabilities had slightly lower foliar P, lower extractable and total soil P, and higher PSC values in the 2005 survey. However, contrary to expectations, foliar N:P ratios were lower at these sites. No strong patterns in P-related parameters were seen when data from all growth phases were combined.

	Chapman's low P sites	Chapman's high P sites
Foliar N (mg kg $^{-1}$)	10409	17719
Foliar P (mg kg $^{-1}$)	797.6	1073
Foliar N:P ratio	13.0	16.8
PME activity (nmol g dwt^{-1} s^{-1})	4.5	5.6
Total soil N (mg kg $^{-1}$)	776.0	6077
Total soil P (mg kg $^{-1}$)	132.4	172.6
Extractable N (mg kg $^{-1}$)	2.4	4.8
Extractable P (mg kg $^{-1}$)	1.9	4.2
Ammonium-Oxalate PSC (mg kg $^{-1}$)	139.5	89.3

Table 1. Summary of plant and soil characteristics for building phase stands. Values represent means of three sites for each category.

Of the building phase stands included in the full survey, 15 had N:P ratios of <14, nine were between 14-16 and a further four were >16. These cut offs, proposed by Koerselman and Mueleman (1996) to indicate relative N and P limitations, suggest that a little over half of the lowland heaths surveyed are currently N-limited. N:P ratios ranged from 8.6-17.9, which is comparable with data from Thursley Common (6.3-15.7, Green, 2006), Surrey heathlands (8.3-20.3, Spankie, 2002) and UK upland heathlands (11-24, Kirkham, 2001). However, values in the UK are generally lower than those in the Netherlands (19-37, Roem & Berendse, 2000), where P-limitation appears widespread.

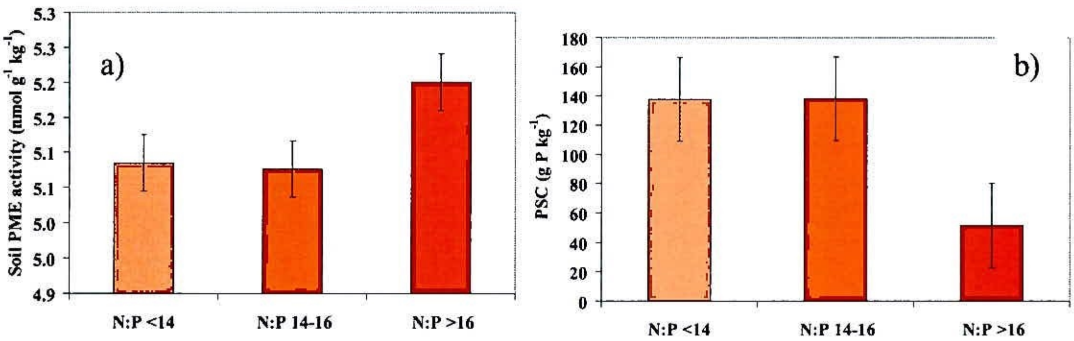

Figure 6. Relationship between foliar N:P ratios and a) soil PME activity and b) phosphorus sorption capacity (PSC)

Analysis of the full data set showed a positive relationship between extractable and total soil P concentrations ($P<0.01$, $r^2= 0.195$). There was little evidence of a relationship between foliar N:P ratios and either total or extractable soil P concentrations. However, Figure 6 shows that a) PME activity is higher and b) PSC is lower at sites where foliar N:P ratios indicate P limitation. Analysis of data from the three different *Calluna* growth phases revealed an increase in soil P and N status with increasing time since last management (Figure 7). Foliar N and P concentrations were not significantly different between growth phases, although mean values in pioneer phase *Calluna* (1.52%, 0.118%) were slightly higher than in either building (1.48%, 0.114%) or mature (1.47%, 0.113%) stage stands.

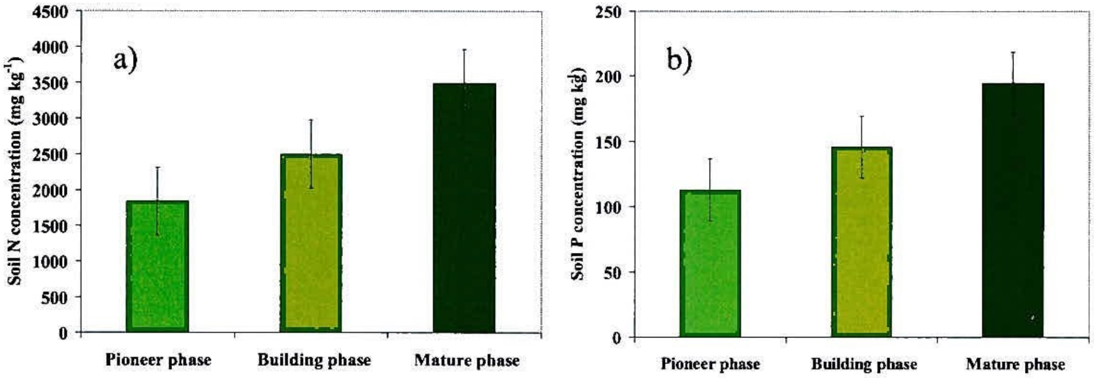

Figure 7. Effect of Calluna growth phase on a) soil N and b) soil P concentrations

Overall, the field survey indicates that there is a considerable variation in plant and soil nutrient characteristics across lowland heathlands in England, that values vary according to stand age, and that a large number of sites may be either P or NP co-limited.

Relationships between N:P ratios and soil P variables were relatively weak. However, there is some evidence that sites with the highest vegetation N:P ratios have a greater demand for P (as evidenced by higher PME activity). More detailed, multivariate analysis of the full data set will be carried out to establish which characteristics are associated with N and P limitation. In addition, the relationship between modelled N deposition rates and both soil and plant variables will be investigated over the next few months.

6. References

Chapman, S.B., Rose, R.J., Basanta, M. (1989) Phosphorous adsorption by soils from heathlands in southern England in relation to successional change. *Journal of Applied Ecology,* **26,** 673-680

English Nature (2006) GIS data obtained on 30.05.06 from: http://www.english-nature.org.uk/pubs/gis/GIS_register.asp

Green, E.R. (2006) *The effect of nitrogen deposition on lowland heath ecosystems.* PhD thesis, Imperial College London.

Kirkham, F.W. (2001) Nitrogen uptake in six hill moorland species in relation to atmospheric nitrogen deposition in England and Wales. *Journal of Ecology,* **89**, 1041-1053

Koerselman, W., Meuleman, A.F.M. (1996) The vegetation N:P ratio: a new tool to detect the nature of nutrient limitation. *Journal of Applied Ecology,* **33**, 1441-1450

Lookman, R., Freese, D., Merckx, R., Vlassak, K., Van Riemsdijk, W.H. (1995) Long-term kinetics of phosphate release from soil. *Environmental Science and Technology,* **29**,1569-1575

Roem, W.J., Berendse, F. (2000) Soil acidity and nutrient supply ratio as possible factors determining changes in plant species diversity in grassland and heathland communities. *Biological Conservation,* **92**, 151-161

Spankie, A. (2002) *What are the likely responses to nitrogen in heathland in southern England ?* MSc Thesis, Imperial College London

Tabatabai, M.A., Bremner, J.M. (1969) Use of *p*-nitrophyenyl phosphate for assay of soil phosphatase activity *Soil Biology & Biochemistry,* **1**, 301-307

Work Package 2:
Impacts, Recovery and Processes

Task 6:
Interactions between nitrogen deposition and grazing at Pwllperian

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[1] Centre for Ecology and Hydrology
[2] ADAS Pwllperian

Task 6. - Interactions between nitrogen deposition and grazing at Pwllperian

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1. Summary

This task aims to evaluate the interaction with of nitrogen deposition with the intensity of sheep grazing together with the relative importance of reduced versus oxidised nitrogen on species change. Nitrogen as either ammonium sulphate or sodium nitrate is applied fortnightly to 24 replicated plots in experimental paddocks of either light or heavy sheep grazing (0.5 or 1 sheep/ha). The grazing treatments started in 1990 and N treatments in 1996. Phosphorus additions were applied to one set of plots to test the hypothesis that N impacts would be greater in non-P limited systems. No measurements were made this year with final vegetation composition measurements due to be made in 2007. A synthesis of the findings from this study were presented at the Biogeomon 5th International Symposium on Ecosystem Behaviour in Santa Cruz, California, June 2006.

2. Policy Relevance

Findings from two long term monitoring programmes and one spatial survey suggest that there have been wide-ranging changes in species occurrence in the UK during the latter half of 20th century associated with increased N availability. These are:

(i) The New Plant Atlas of the UK (Preston et al. 2002) which indicated a decline in the frequency of occurrence of plant species characteristic of low nutrient availability between 1930-69 and 1987-99 and an increase in the geographic range of species associated with high nutrient availability.

(ii) The Countryside Survey (www.cs2000.org.uk) which has reported on results from repeated surveys of higher plant species data from permanent quadrats from 1978-1990 and 1990-1998. Results again suggest a shift towards plant species associated with high nutrient availability particularly in low nutrient habitats such heathland and infertile grasslands (Haines-Young et al. 2003).

(iii) A spatial survey undertaken in acid grassland by Stevens et al. (2004) in the UK (Figure 2) which identified a decline in species richness across a N deposition gradient.

All three studies suggested N deposition as a major factor contributing to the reported shift in species composition and used various statistical approaches to support this conclusion. However, a large increase in grazing animals over the second half of the 20th century could have contributed to the shift in species particularly in infertile grasslands and heathlands. The Pwllperian nitrogen/grazing study aims to identify the differential signals of these two drivers and their interaction which will inform application of critical loads at the site specific level, identify suitable indicators, help in the interpretation of output from monitoring programmes, and assist with model development.

3. Project Update

No update is due for this report.

Work Package 2:
Impacts, Recovery and Processes

Task 7:
Whim Moss N Manipulation Experiment and Open-Top Chamber Flux Work

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1. Summary

- Effects of ammonia gas, from point sources, on plant species are concentration mediated.
- NH_3 sensitive ombrotrophic bog species will not be protected from irreversible damage from high NH_3 concentrations by either the monthly or annual Critical Levels proposed by Van der Eerden *et al.* (1991).
- The frequency and intensity of NH_3 exposure experienced by vegetation in the field, in the vicinity of NH_3 point sources, is spasmodic invoking bi-directional NH_3 transfer between plants and the atmosphere and involves much higher NH_3 concentrations than the values estimated from integrating monthly ALPHA samplers. Critical Levels have been derived from short-term, continuous exposures to constant NH_3 concentrations.
- Acidification of the peat from N deposition is significant, but the direction depends on the N form: Reduced wet N acidifies, oxidized wet N increases soil pH and the dry deposition of NH_3 increases soil pH.
- Acidification through the addition of ammonium –N affected the availability of nitrate – N via effects on nitrification which was minimal below pH 4.2.
- Eutrophication by all forms of N was related to the N dose with respect to the pore water and potentially available mineral pools.
- The use of N tissue status of mosses as a bioindicator for N impacts has some caveats. Extended exposures to high N deposition or [NH_3] can lead to low N concentrations as cells become damaged and leaky.
- Amino acid levels may be more indicative of negative N impacts, but on a routine basis these are costly to analyse and require controls and a range of N doses for interpretation.
- Reduced N is more readily taken up than oxidized N by mosses and thus amino acids increase most in response to reduced N inputs but not significantly.
- Loss of individual species appears to require the interaction between high N doses and a stress, the accumulation of N on its own does not appear to be toxic.
- N deposition at ~ 2 x the Critical Load for moorland inhibits the regeneration of *Calluna* from old wood but not the establishment of *Calluna* seedlings. Previous treatment with either ammonia or wet ammonium has not inhibited colonization by mosses.

2. Policy Relevance

All aspects of the N manipulation experiment at Whim Moss remain highly relevant to both Defra and the Conservation Agencies. The Whim field study on an ombrotrophic bog is unique in providing near natural treatment conditions with respect to the timing and frequency of application and the low ionic concentration of treatment solutions. In addition

Whim has a history of relatively low background N deposition and so the responses are not those of an acclimated system. The Whim N manipulation experiment therefore provides real evidence of the effect of different N forms under near natural conditions on bog species. Previous N manipulation studies funded through the Defra N / acidification Umbrella have demonstrated the importance of management in mediating ecosystem N responses. The Whim experiment has highlighted the importance of environmental factors in determining the scale of N impacts. The relative abundance of different major groups of vegetation, e.g. mosses, lichens, liverworts and ericoids at the site has enabled us to demonstrate how species specific many responses are, and that generalizing across major groups, such as the mosses, is not informative and can mask important differences.

Separation of the different N forms into reduced (NH_3 and NH_4^{+}) and oxidized (NO_3^-) forms enables source attribution and appropriate control measures. The experiment will deliver on:

1. Identification of N sensitive species and traits that can be monitored to comply with the European Habitats Directives.
2. Validation of critical loads (CL) for bogs and evaluation of the role of concentration (Critical Levels). Demonstration of N impacts on a blanket bog, under near natural exposure conditions, and at a site that has a low N history of ambient inputs to identify species at risk and validate the UNECE agreed CL taking into account modifiers i.e. P&K availability and environmental conditions.
3. Evaluation of the potential of N impacts on wetlands (eutrophication or acidification) to affect climate change via effects on C and N sequestration and greenhouse gas emissions (CO_2, N_2O, CH_4).
4. Examination of the potential for recovery, recolonisation and invasion of bog species in peat monoliths, previously exposed to 5 years of N inputs, in open-top-chambers.
5. Refinement of N deposition estimates from NH_3 concentration data to individual species and mixtures of bog vegetation and identification of the role of environmental influences on NH_3 – N deposition processes based on canopy resistance theory.

In order to validate the impact of policy on the environment with respect to N effects and to target legislation at the source of the offending N source we need to identify robust bioindicators specific to oxidized or reduced N. We have been evaluating a range of above and below-ground changes in response to our N inputs to see how well they describe N impacts and equally important how temporally and spatially robust they are.

3. Project Update

3.1 Whim Moss:

All year round treatment of the wet and dry plots/transect has been maintained, although the annual dose for 2005/06 has been under achieved by ~ 25% due to the extended night frosts in April and May. Data capture has been high, >98 %. The meteorological data files have all been verified, ready for lodging with the CEH data centre, and are available as annual or, cumulative trends for meteorological and treatment dose/concentrations. ALPHA samplers used to monitor ammonia concentrations along the transect have been relocated to improve estimates of NH_3 concentration data over the whole area to correspond with biological recording. These data are currently being used to calculate N deposition along the transect for each month at up to 12 distances since May 2002. These revised N deposition estimates are based on the flux chamber data (Jones 2006) and take into account NH_3 concentration dependent R_c values, diurnal release and wind speed. Since last year the key measurements on nutrient status and species cover along the NH_3 transect and N status of the vegetation in the 44 wet plots, have been repeated. Assessments of visible damage to a range of species are bing made routinely to help verify critical loads and Levels. A lichen transplant study was undertaken over an 8 month period to evaluate damage in relation to changes in nutrient status to test the robustness of foliar N as a damage indicator. In addition pilot measurements of the δ ^{15}N signal in lichen, moss, *Sphagnum, Calluna* and *Eriophorum* were undertaken to evaluate their potential for determining where different species source their N and which N forms the different vegetation utilize. Twenty eight wet deposition plots and 12 positions along the NH_3 transect have now been equipped with static chambers for trace gas flux studies and dipwells to monitor water table, locating collars have also been inserted for soil respiration measurements. Additional funding from NITROEUROPE will facilitate a monthly measurement programme for N_2O and CH_4. Measurements began in May following the NEU static chamber protocols. In addition CEH and MMU are funding a joint student, Chris Field, to look at the dynamics of C an N cycling in response to the different N forms at Whim and the MMU sites at Ruabon and Budworth. This project will provide data for the dynamic modelling group. A PhD studentship, sponsored by SNH, to look at N impacts on the vitality of *Sphagnum* in Scotland will commence in September 2006 in collaboration with Edinburgh University. The research will concentrate on the effects of N interactions with stress with a view to management, understanding how N effects are mediated and testing N bioindicators over an N deposition range in Scotland. Lisbon University will provide a postgraduate biochemist to investigate responses at the cellular level in a collaborative project on understanding N impacts at the cellular level.

Collaboration with Jans Roelofs team at the University of Nijmegen (Netherlands) via York University (Mike Ashmore / Leon van den Berg) through the supervision of 2 Masters student placements from October to December 2005, enabled a significant amount of chemical analysis to be undertaken on both above and below-ground components. A third student is currently repeating some measurements for verification, prior to writing up the work for publication. Supervision of an MSc project with SAC Edinburgh provided information on soil chemistry and encytraeid numbers and species composition along the ammonia transect in relation to soil chemistry. PhD students from Macaulay, Bangor University, Nottingham University and Imperial College have also used samples from the site for analyses of phosphatase activity in lichens to determine how N eutrophication increases the demand for other nutrients: Phosphatase activity in peat, as an indicator of N effects on decomposition processes, and in mosses are also being undertaken; together with microbial biodiversity and enzyme activities with respect to DOC as part of cross cutting studies on N impacts.

3.2 The OTC recovery experiment:

This has been reassessed for the cover of original species and the appearance of invasive species.

3.3 Flux chamber studies:

Evaluation of species effects on ammonia deposition and the significance of the NH_3 concentration and environmental variables for deposition are now complete. Four papers have been written from the thesis and 2 papers are being written to describe how the observations have been used at Whim to refine the deposition estimates. An evaluation of the critical levels estimated by Van der Eerden *et al.* (1991) has been undertaken using the 4 years of available data.

Progress on the agreed deliverables with respect to data collection is on schedule and through collaboration many more measurements have been undertaken. The key aim over the summer will be to produce 1 or 2 papers. In the autumn the species cover data assessments will be repeated for both the wet and dry plots. A background document will be prepared in relation to the effects of infrequent ammonia exposure on a bog ecosystem for the UNECE ammonia workshop.

4. Collaboration with Dynamic Modelling Umbrella and Freshwater Umbrella

In last years report we highlighted the heterogeneity at Whim Moss, reflecting the undulating surface and different layers of vegetation that have given rise to hummocks and hollows. Soil sampling and the collection of soil water have now been undertaken at different depths. These results indicate that while depth and the above-ground vegetation do influence the absolute amounts of different chemicals the overall treatment trends remained relatively constant. Thus soil rhizon samplers have now been inserted into 28 wet plots (excluding PK treatments) and 12 positions along the NH_3 transect at 5-10 cm depth. Data is being collected in conjunction with MMU. Over the next 12 months, pH, C:N ratios, soil N mineralization, available P and exchangeable cations, the major cations and anions will be measured monthly. Ecosystem CO_2 exchange will be measured on a campaign basis and assessments of standing biomass and litter production and breakdown will be made together with annual productivity on key species: *Calluna* and *Sphagnum* for modelling purposes.

5. Key Findings

5.1. Reduced versus oxidized N: acidification and eutrophication

5.1.1 Acidification potential

The major effects below-ground relate to acidification potential, additions of oxidized N increased peat pH, while reduced N (NH_4^+) reduced the pH and acidified the peat. By contrast reduce N inputs as NH_3 increased peat pH. Over 4 years the change in pH approximates to 0.5 pH units over the pH range 3.8 to 4.6. Along the ammonia release transect 90% of the variation in soil pH was explained by the mean NH_3 concentration, ie the change in pH followed the concentration curve not the linearity of N deposition with distance from the NH_3 source (see Fig 1). By contrast pH changes in response to the wet inputs of NH_4^+ and NO_3^- were linear. pH changes have implications for N transformations in the peat as seen in measurements of availability of the different N pools. Along the NH_3 transect, where the peat pH exceeded 4.2, NO_3^- could be measured in the soil solution in proportion to the concentration of NH_3, indicating a proportion of the NH_3-N deposited had been nitrified. By contrast in the wet plots where reduced N was added as NH_4^+ the proportion that was nitrified was negligible, due to the low pH restricting the activities of nitrifying bacteria.

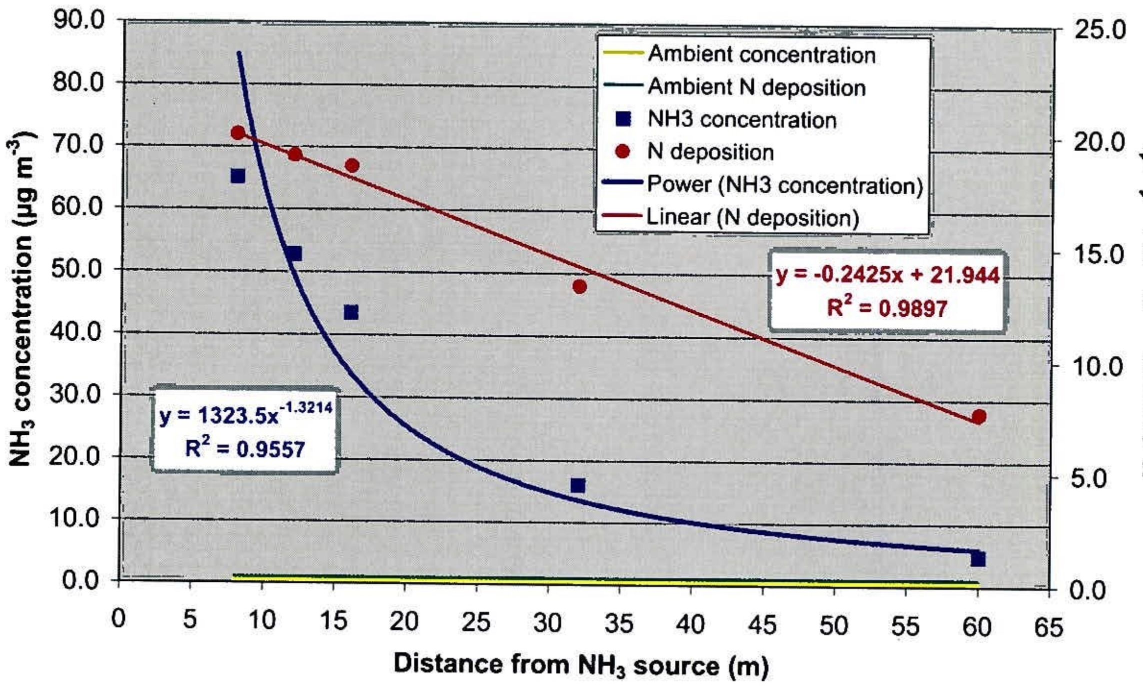

Figure 1. Average ammonia concentrations along the NH_3 release transect at Whim Moss measured using ALPHA samplers 0.1 m above the vegetation and estimated N deposition incorporating refinements from Jones (2006).

5.1.2 Eutrophying, fertilization effect of ammonia:

Depending on the soil pH along the NH_3 transect, upwards of 4 times more NH_4-N was present in the rhizon pore water samplers compared to NO_3-N. As soil pH increased the proportion of NH_4-N: NO_3-N declined. There was a linear increase in both soluble N forms in response to [NH_3]. Likewise large increases in the mineral N pool (KCl extractable) were measured for NH_4- N. This surface mineral fraction (0-6 cm), showed a 100-fold enhancement in response to mean peak NH_3 concentrations of 80 ug m^{-3} and a 10-fold increase in NO_3-N. By contrast the total N pool of the system was not significantly affected along the NH_3 transect. This observation is consistent with the estimated annual deposition along the NH_3 transect, maximum 20 kg N ha^{-1} y^{-1} which equates to < 5% of the soil N content.

The *eutrophying, fertilization effect:* was demonstrated in significant increased in available (soil pore water) N and significant increases in the storage pool of mineral N (KCl extractable) in response to N deposition. Total N which represents the mineral and organic N pools was not affected by the N deposition along the NH_3 transect. Measurements are only available for pore water N for the wet plots. Results indicate much smaller enhancement of the soil water when the N is supplied as wet deposition. However, straight comparisons between dry and wet deposition are confounded by temporal treatment differences with respect to rain and the concentration of then reaching the peat

5.2. *Bioindicators*:

For the first 3 years of N deposition enhancement foliar N concentrations at the high N inputs were significantly increased in the moss *Hypnum jutlandicum*. Uptake of reduced N was significantly greater than uptake of oxidized N. Data for 2006 indicates this memory effect is not sustained and that N concentrations in *Hypnum jutlandicum* reach a ceiling and then decline. Such declines in foliar N have also been recorded in *Racomitrium lanuginosum* (Pearce I. and Jones L. *pers comm.*) in response to continuing high N inputs. The mechanism behind this decline is not known but it may reflect internal membrane damage arising from the accumulation of NH_4^+ ions or protons. This observation is consistent with Curtis *et al.* (2005), who measured reduced N retention in moorland systems where the moss flora has declined under enhanced N deposition.

A comparison of these peak % N values with those for woodland mosses growing near poultry farms (Pitcairn *et al.* 2006) suggests different moss species have different capacities for N accumulation. Miller (2006) suggested this ability to store/detoxify N may be related to vacuole size. If this is the case it means we may be able to use vacuole size to predict N tolerance in mosses. Our observations indicate that N status provides some indication of N impact but the presence and absence of different moss species from a habitat may also be a useful indicator of N compacts. So, when N deposition is increased we would expect the mosses with small vacuoles to disappear first. The observation that not all mosses accumulate N to the same extent means that comparisons of elevated N concentrations for bioindication purposes, to indicate N enhancement, should ideally be restricted to the same species.

Amino acid concentrations were measured in *Sphagnum capillifolium, Calluna vulgaris* and *Erica tetralix* and these appeared to provide considerable information concerning the vitality of these species. All three N forms enhanced the amino acid concentrations, with the relationship with NH_3 relating best to [NH_3] not NH_3-N deposition. Arginine was the main amino acid in all 3 species. Arginine consists of 4 N to every C so is an efficient way of detoxifying N. In *Sphagnum* concentrations ranged from peak values of 50 down to < 20 μmol g^{-1} dwt along the NH_3 transect. In the wet plots both the 64 and 32 kg N ha^{-1} y^{-1} treatments significantly enhanced arginine and also asparagine concentrations, with reduced N having a bigger effect as would be predicted from the higher foliage N concentration with reduced N. Such elevated concentrations are symptomatic of impending damage to the *Sphagnum* in these treatments however, there are no visible damage symptoms, as yet, in the wet plots. By contrast along the NH_3 transect the green form of *S. capillifolium* is in a very poor state. In *Erica* amino acid concentrations were significantly enhanced but only along the ammonia transect. Its arginine concentrations exceeded those in *Calluna* which is counter intuitive given the *Erica* appears much healthier than the *Calluna*.

5.3 *Recovery in OTC's*

As noted in 2005 (Emmett *et* al. 2005) *Callluna* does not resprout from the old wood in the high wet N treatments, > 32 kg N ha^{-1} y^{-1} or > 6 μg NH_3 m^{-3}, confirming that N depositions at~ 2 x the critical load for moorland inhibits the regeneration of *Calluna* from old wood. However, establishment and growth of *Calluna* seedlings was not affected up to 64 kg N ha^{-1} y^{-1} as NH_4^-N. Cover of the dominant planted species has changed; *Molinia* remains the dominant plant in the treatments that received high NH_3 along with *Potentilla erecta. Deschampsia flexuosa* has sustained its presence in all chambers. Only one of the invasive grasses, *Holcus lanatus* is still present, though not expanding, *Agrosis tenuis* has gone. The moss flora has sustained and expanded its cover in most chambers. *Dicranum scoparium* and *Hypnum jutlandicum* are the dominant mosses present in all chambers. *H. jutlandicum* is thriving in chambers which received high NH_3 whereas *Pleurozuim schreberi*, one of the two dominants pleurocarpous mosses at Whim, was absent from all but one chamber. *Sphagnum* species are colonizing chambers with ≤ 64 kg NH_4 or 6 μg m^{-3} NH_3.

5.4 *Critical N loads and Critical Levels for NH_3*

The flux chamber work has shown that ammonia deposition is highly species dependent with *Sphagnum* moss and *Cladonia* lichens providing a large potential sink for NH_3, linked in part to their respective wetness and surface area (Jones 2006). Non-vascular plants such as mosses and lichens lack stomata and thus show no diurnal pattern in NH_3 deposition. By contrast NH_3 deposition to vascular plants such as *Calluna* is much greater if the NH_3 is present during daylight hours. Wet surfaces, a characteristic of bogs, increase the potential for NH_3 deposition. However, because the resistances to deposition are NH_3 concentration dependent and bog species, having evolved under conditions of low N availability tend to have low internal N concentrations and high resistances, N deposition at high [NH_3] is relatively small to bog species as internal saturation occurs very quickly. Thus the high external ammonia concentrations that exist in the immediate vicinity downwind of NH_3 point sources are unlikely to result in high N deposition to the semi-natural plant communities growing there. Estimates of N deposition along the release transect at Whim Moss have now been revised to take account of environmental and concentrations effects so that the maximum deposition estimate has come down from a 300 kg N ha^{-1} y^{-1} maximum to ~ 20 kg N $ha^{-1}y^{-1}$ (Fig. 1). These revised N deposition estimates confirm the view that the detrimental impacts observed and being measured along the NH_3 transect are most likely to be induced by the periodic high NH_3 concentrations (levels) (Emmett *et al.* 2005).

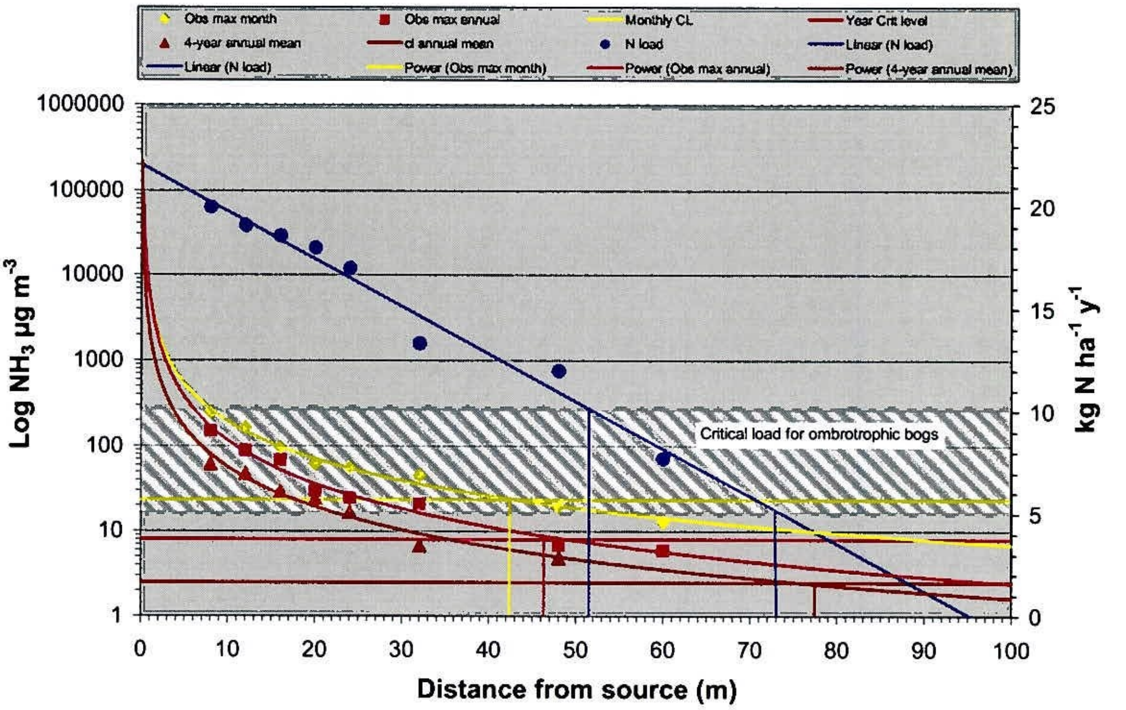

Figure 2. Observed maximum monthly and annual ammonia concentrations measured using ALPHA samplers at 0.1 m above the vegetation, along the ammonia release transect at Whim Moss, in relation to Critical Levels as proposed by Van der Eerden *et al.* (1991) and Burkhardt *et al.* (1998). Calculated ammonia deposition (blue symbols) in relation to the

Critical N Load (CNL) range for ombrotrophic bogs (depicted by the pale grey background). The distance along the transect, where the blue vertical lines intercept represent the protection zone offered by these CNL.

Ammonia concentrations collected over the 4 year period of NH_3 exposure have been used to test the validity of the Critical Levels for NH_3 proposed by Van der Eerden *et al.* (1991) to protect heathland species: **1 month at 23 µg m^{-3} NH_3 and 1 year at 8 µg m^{-3} NH_3.** Figure 2 compares the Critical levels with those observed i.e. measured with ALPHA samplers (Burkhardt *et al.* 1998). The figure shows the maximum of the highest monthly [NH_3] for each of the 4 years at each distance along the Whim ammonia transect, the maximum annual value at each distance, taken from the maximum of the annually averaged monthly values, the estimated Critical Level taken from Van der Eerden *et al.* (1991) and extrapolated to the 30 year protection period used to estimate N Critical Loads and the N deposition load. Van der Eerdens' Critical concentrations for one month and one year of continuous exposure at a constant NH_3 concentration have been included (horizontal lines) for reference together with the Critical N deposition Load for ombrotrophic bogs (5-10 kg N ha^{-1} y^{-1}). From these data it can be seen that the monthly Critical Level offers the least protection to the ecosystem while the 30 year annual Critical level offers the most protection (observed data dissect the horizontal lines further from the source). Plotting the timing of death of the sensitive species growing along the NH_3 transect (Fig 3), **(no deaths observed for the wet deposition plots)** in relation to the average [NH_3] prior to death being recorded it can be seen that the annual Critical Level would not have protected the most sensitive species at Whim Moss, *Cladonia portentosa* beyond 3 years and even this is generous given our recording of the time of death provides for absolutely no recovery.

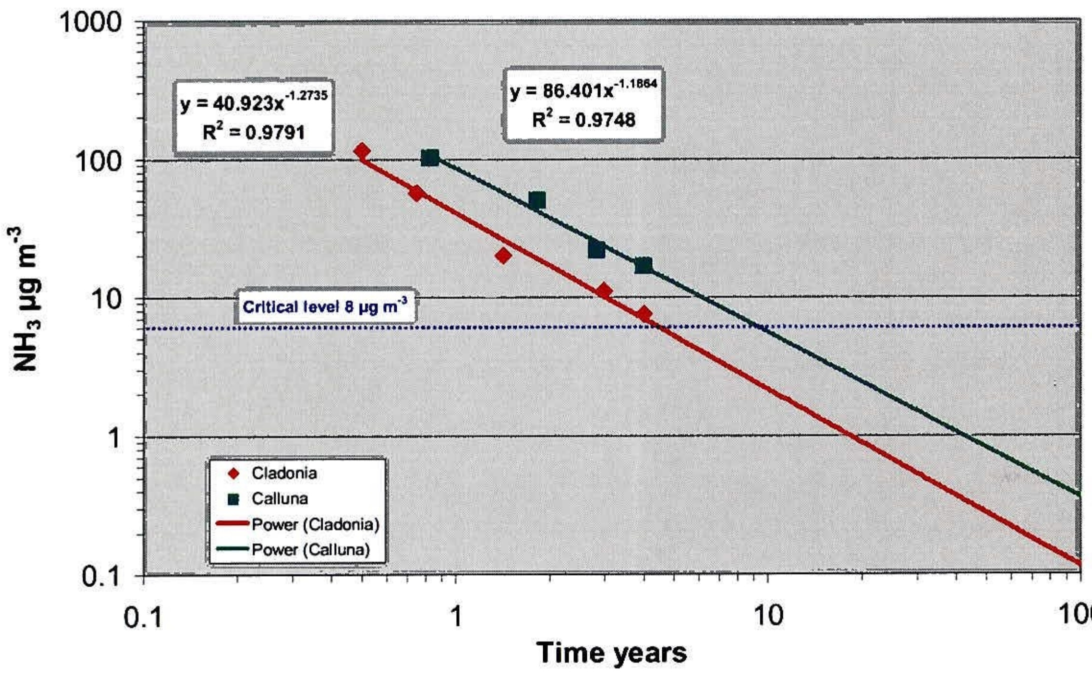

Figure 3. Observed times of death of *Cladonia portentosa* (eradication) (red symbols) and *Calluna vulgaris* (~90% dead) (turquoise symbols) growing along the ammonia release transect at Whim Moss in relation to the previous mean ammonia exposure concentration. Based on Van der Eerden's annual CL (8 µg m^{-3}) (Van der Eerden *et al.* 1991) it can be seen

that for *Cladonia* it will be exceeded after 4 years whereas for *Calluna* it will take at least 7 years.

The extrapolated Critical Level is much more protective at 2.5 μg m^{-3} NH_3 but even this assumes a threshold concentration for toxicity (see dotted line). Overall the Whim data suggest lichens are insufficiently protected by Critical Levels developed for all heathland vegetation (Van der Eerden *et al.* 1991), especially considering we are basing our observations on death and irreversible damage not just a negative impact on a plant characteristic *e.g.* growth.

6. References

Burkhardt, J., Sutton, M.A., Milford, C., Storeton-West, R.L. & Fowler, D. (1998).Ammonia concentrations at a site in southern Scotland from 2 yr of continuous measurements. Atmospheric Environment, **32**, 325-331.

Curtis, C.J., Emmett, B.A., Grant, H., Kernan, M., Reynolds, B. & Shilland, E. (2005). Nitrogen saturation in UK moorlands: the critical role of bryophytes and lichens in determining retention of atmospheric N deposition. Journal of Applied Ecology, **42**, 507-517.

Emmett BA, Williams B, Ashmore M, Benham S, Britton A, Broadmeadow M, Caporn SJM, Carroll J, Cresser M, Cresswell N, Crossley A, Edmondson J, Grant H, Green E, Green S, Helliwell R, Hinton C, Hughes S, Jones A, Jones M, Leake J, Lee J, Leith ID, Oakley S, Ostle N, Power S, Price EAC, Ray N, Reynolds B, Sheppard LJ, Smart R, Smart S, Sowerby A, Vanguelova E, Williams D. Wildig J. 2005 Terrestrial Umbrella Annual Report Contract No CPEA 18. July 2005. Defra, London.

Jones, M.R. (2006). Ammonia deposition to semi-natural vegetation. Ph.D Thesis. University of Dundee.

Miller, T., Pearson, J., Smith, S. & Stanley, C. (2006). Moss tissue structure, pH regulation and sensitivity to atmospheric nitrogen pollution. CAPER Abstracts, 10-12 April 2006, Edinburgh.

Pitcairn, C.E.R., Skiba, U.M., Sutton, M.A., Fowler, D., Leith, I.D., Munro, R.C. & Kennedy, V. (2002). Defining the spatial impacts of poultry farm ammonia emissions on species composition of adjacent woodland groundflora using Ellenberg Nitrogen Index, nitrous oxide and nitric oxide emissions and foliar nitrogen as marker variables. Environmental Pollution, **119**, 9-21.

Van der Eerden, L.J., Dueck, TH. A., Berdowski, J.J.M., Grevan, H. &Van Dobben, H.F. (1991). Influence of NH_3 and $(NH_4)_2SO_4$ on heathland vegetation. *Acta Bot. Neerl.*, **40(4),** 281-296.

Work Package 2:
Impacts, Recovery and Processes

Task 8:

The potential for recovery and interactive effects of climate change in acid grasslands

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CEH Bangor

Task 8. The potential for recovery and interactive effects of climate change in acid grasslands

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CEH Bangor

1. Summary

This project aims to evaluate the rate and direction of recovery of a heavily polluted upland ecosystem and identify potential implications of future climate change. An experimental approach is used which uses retractable curtains (Beier et al. 2004) to reduce N deposition, create repeated summer drought and produce whole ecosystem warming. Changes in water quality, species composition and soil organic matter turnover are being assessed . The field site has now been successfully moved from Mid Wales to a site in the Peak District although this was delayed due to bad weather and staff illness. Baseline measurements have been made and treatments started in early July 2006. A second field site with 8 year long climate change manipulations on a heathland in less polluted conditions has been continued and results have highlighted the importance of repeated summer drought in mobilising both carbon and nitrogen from the soil store. Results from this site continue to be presented at a range of national and international conferences and meetings and future work will integrate the results to inform both model development and critical load assessments.

2. Policy Relevance

Substantial areas of the UK are currently identified as being in exceedance of empirical critical loads of nutrient nitrogen. Emission control policies are intended to decrease deposition loadings and thus initiate recovery from N enrichment. Climate change may interfere with this recovery and/or mask signals due to both direct and indirect effects on N cycling and the competitive balance between species. Whilst monitoring work is undertaken to follow ecosystem responses due to changes in air pollution and climate change, concurrent changes in drivers such as land management, air pollution and climate can make attribution of change in ecosystem structure and function to individual drivers problematic. Experimental approaches such as this undertaken here enable these causal links to be isolated and quantified. This type of information is required by both model developers and the UK National Focal Centre on Critical Load Mapping to enable the potential implications of deposition reductions and climate change on critical loads exceedance to be evaluated.

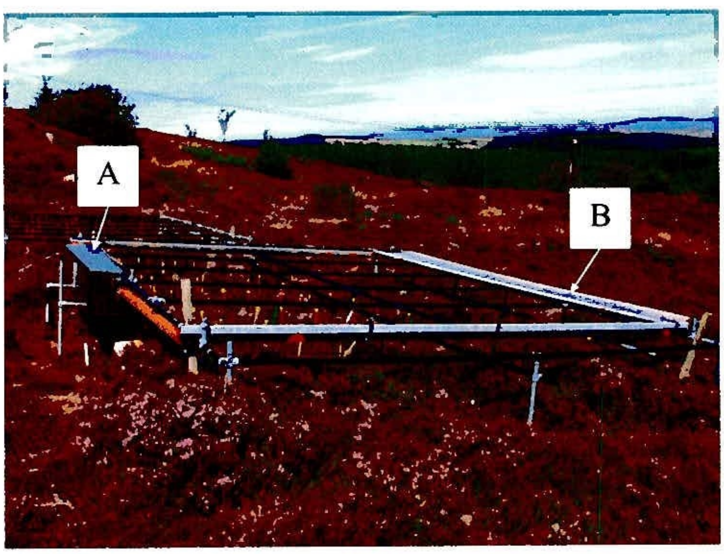
3. Project Update

The experimental site has been established on Peaknaze Moor, Derbyshire (Figure 1). The experiment utilises a retractable roof system (Figure 2) that allows roofs to be draw across the vegetation during rain events thus excluding N inputs in bulk precipitation, whist 'clean' rain is substituted using an irrigation system. The experimental design also has an addition set of roofs for climate manipulation treatments. Repeated summer drought uses transparent roofs similar to that used in the pollution recovery treatment, however without the irrigation system to reapply the rain. Another treatment at the site is warming, which uses reflective covers rather than transparent polyethylene covers, and are drawn across the experimental plots at night (thus preventing heat loss due to infrared radiation).

The roofs and motors are housed on structures constructed from scaffolding poles. These were built over the experimental plots as well as control plots, to ensure the monitoring for possible experimental error from the structures shading the plots.

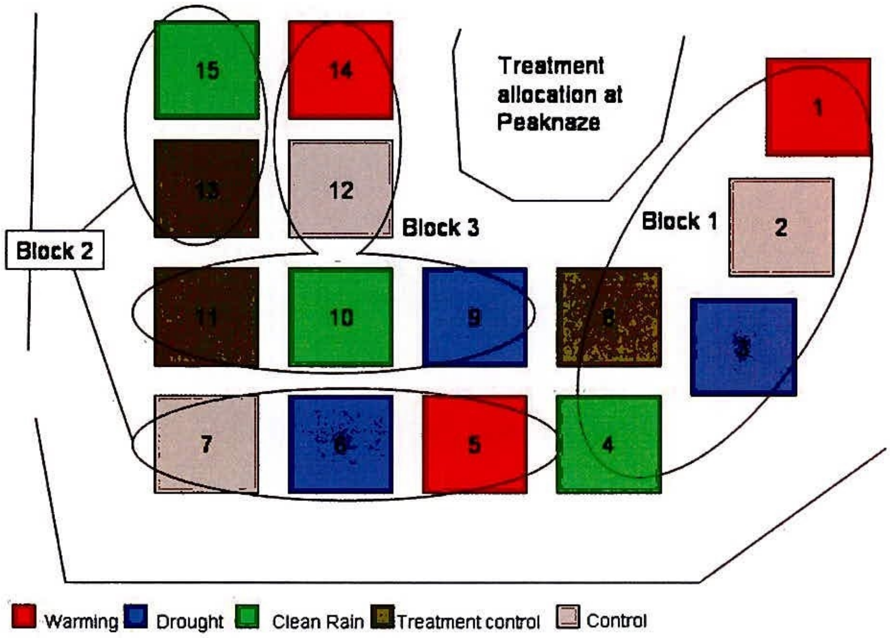
Figure 1. The Peaknaze project, Derbyshire January 2006. Boardwalk now surrounds the plots to protect the heathland from trampling.

Figure 2. The retractable roof system utilised in the Peaknaze roof project. When triggered, a motor (A), pulls the transparent polyethylene curtain (B) (for the pollution recovery and repeated summer drought treatment, or reflective curtain for the warming treatment) across the plot.



During this first year, we have also constructed a boardwalk connecting the plots and experimental areas in order to protect the SSSI designated heathland. Solar panels and wind generators have also been installed and are now generating power for both sensors and loggers as well as the climate and pollution manipulations. A shed has also been constructed to house scientific equipment, treatment controllers and loggers. During year 1 of the project, we have constructed the site, installed equipment and sensors, and collected key baseline data. Treatments were assigned to plots in 2005 according to collected baseline information to ensure as few inherent differences between treatment plots as possible (Figure 3).

Figure 3. Treatment and block allocation for the Peaknaze roof project.



Sensors have been installed in all 15 plots and are logging data. Soil temperature (5cm down the profile) and air temperature (20cm above soil level) are being recorded hourly (the mean of 10 minute intervals). TDR sensors and moisture blocks are measuring soil moisture parameters, and are installed in all plots and again recording hourly.

Baseline plant biomass, using the pin-hit methodology, has also been measured in the plots (Figure 4). The habitat is classified within the M20b unit of the NVC: *Eriophorum vaginatum* blanket mire, Calluna-Cladonia sub-community. At the same time as baseline plant biomass was measured, areas of each of the plots were also destructively harvested following pin-hit estimation to gain conversation factors between the number of pin-hits and the actual plant biomass in the plots for future years pin-hit recording.

4. Key Findings

A similar experiment in North Wales at Clocaenog has shown that the different heathland species respond differently to climate manipulations (Penuelas et al. 2004). Small magnitude changes in temperature (~1oC) has resulted in a 10% increase in overall plant biomass, mostly as a result of the response of *Calluna vulgaris*. However, the same increase in temperature has resulted in a decline in *Empetrum nigrams* at the site (Figure 5). A key objective of the Peaknaze experiment is to test the relative importance of climate change and nitrogen deposition in the decline of this species reported in the Peak District as the plant community is dominated by this species.

Figure 4. Species composition at Peaknaze in baseline measurements using the pin-hit method. Error bars show the standard error of the mean of three replicate plots.

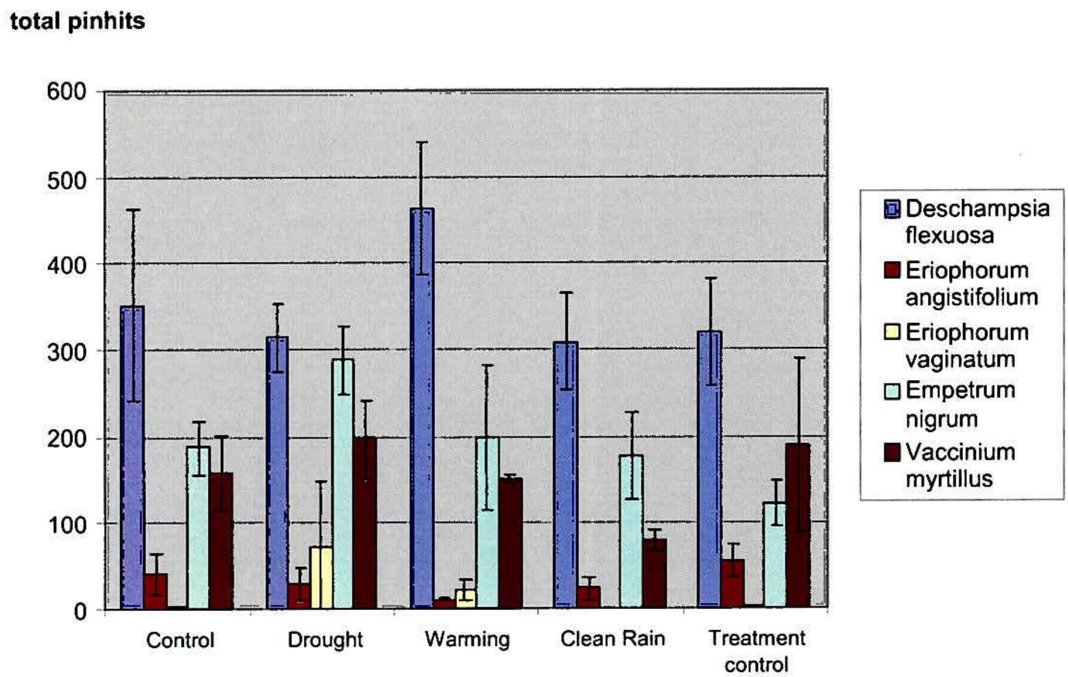
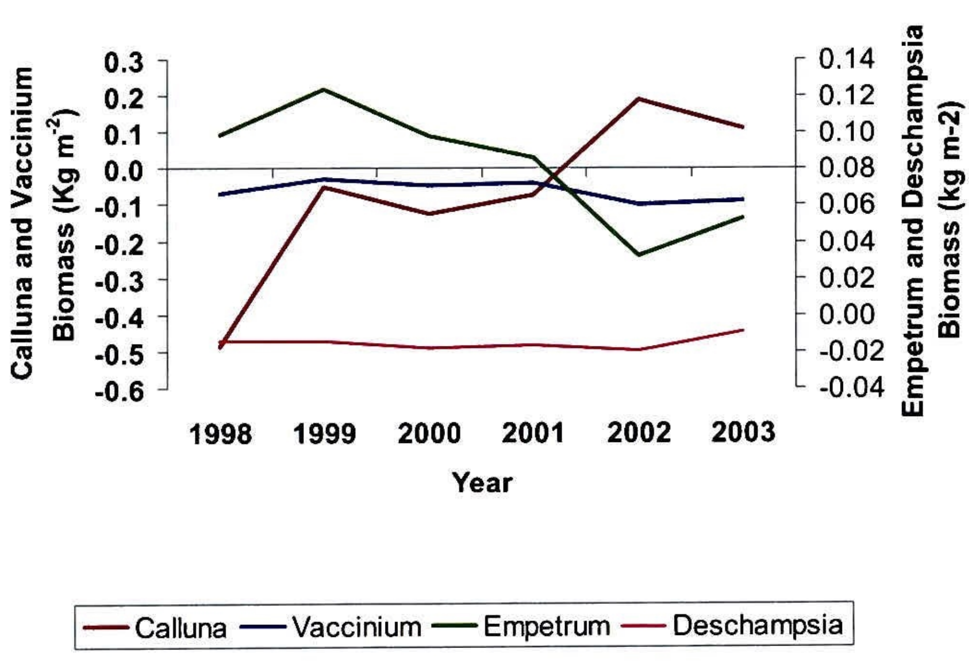


Figure 5. The impact of a 1 °C increase in temperature at a heathland in Cloacenog, North Wales. The figure shows change from control in plant biomass in the warming treatment compared to the control treatment.



Soil respiration and ecosystem level CO_2 exchange have also been measured regularly to establish baseline data (Figure 6). Measurements will attempted every two weeks although weather frequently makes this problematic. This baseline period will be essential for establishing inherent environmental variability between plots, so any treatment differences as a result of the climate and pollution level manipulations can be identified. As an example, Figure 7 shows data from soil respiration measurements, as no treatments have been applied, a 1:1 relationship should be observed. Although there is a strong correlation between soil respiration between the two plots, the relationship does vary from 1:1 revealing small differences between the plots assigned to the treatments.

Figure 6. Soil respiration at the Peaknaze roof project.

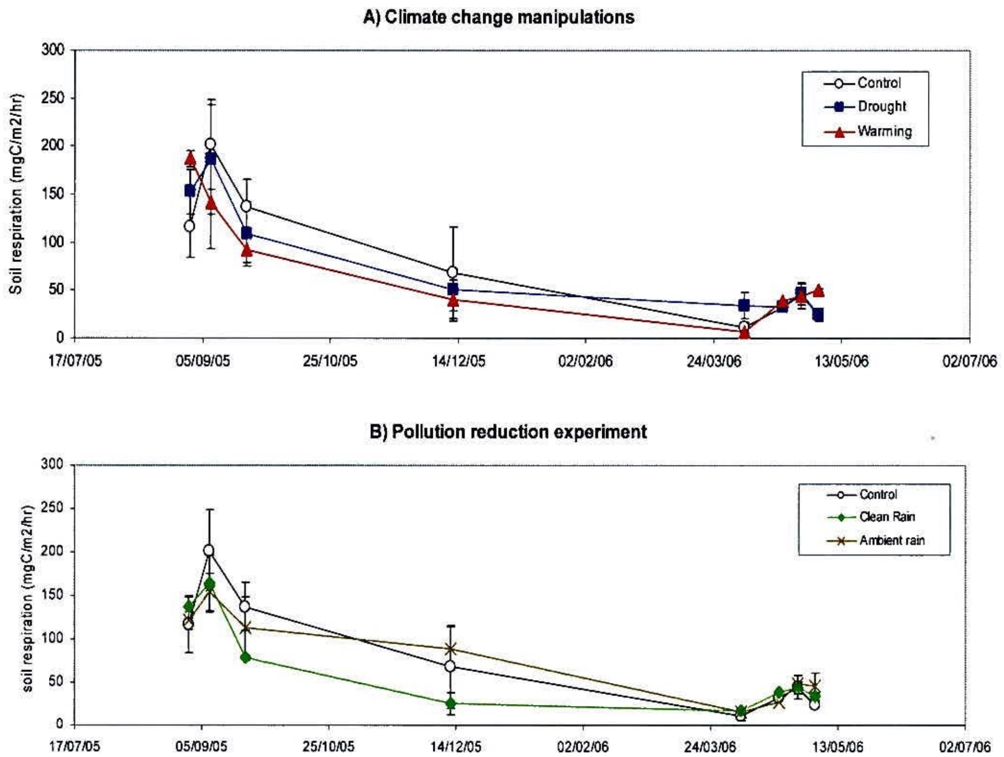
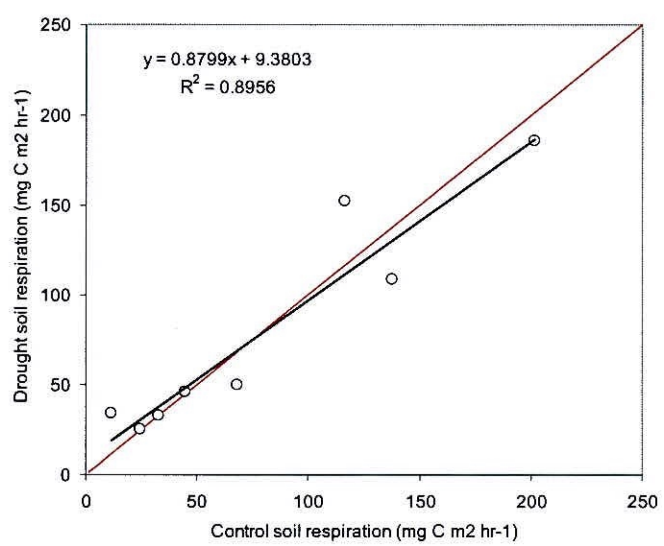


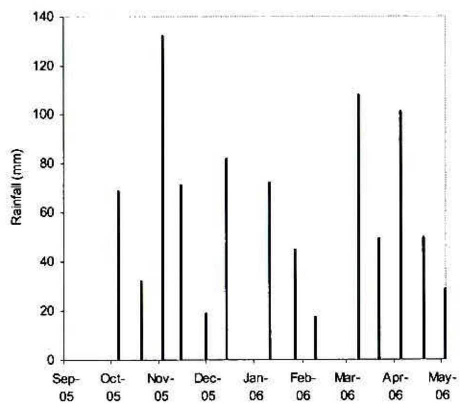
Figure 7. The relationship between soil respiration in the control and drought plots, n=3.



Long term monitoring of soil respiration in Clocaenog, North Wales has shown that both repeated drought and warming increased soil respiration. The increase took a number of years of climate manipulation to be realised, however the increase was seen year-round for both warming and drought although the drought treatment was only operated for 3 months a year (Emmett et al. 2004). The Peaknaze roof project will allow us to compare how soil respiration and CO_2 flux (with the addition of the net ecosystem CO_2 exchange measurements) is affected in a more polluted site and to test the relative importance of these climate drivers relative to the response to reduced inputs of nitrogen deposition.

Soil water has been collected fortnightly since Oct 05 using zero tension lysimeters, and following a settling in period is now being analysed each month for Na, K, Ca, Mg, Total Al, NH_4, NO_3, SO_4, Cl, Total P, DON, DOC. Rainfall has been collected and the amount measured and then also chemically analysed the same determinants as soil water samples. Cloud samples are also being collected and analysed. We are still waiting for data from the chemical analysis of the water samples (for both soil and rain waters), however Figure 8 shows rainfall at the site, as measured by a ground level rain gauge.

Figure 8. Rainfall (mm) at the Peaknaze roof project.



Measurements are continuing at the experimental site and we are continually updating the databases that the experiment is generating. We are now ready to begin the climate change manipulations and envisage a 'switch on' of experimental treatments in early July 2006.

5. References

Beier, C., Emmett, B., Gundersen, P., Tietema, A., Penuelas, J., Estiarte, M., Gordon, C., Gorissen, A., Llorens, L., Roda, F. & Williams, D. 2004 *Novel Approaches to Study Climate Change Effects on Terrestrial Ecosystems in the Field – Drought and Passive Night Time Warming*. Ecosystems 7:583-597

Emmett, B.A., Beier, C., Estiarte, M., Tietema, A., Kristensen, H.L., Williams, D., Penuelas, J., Schmidt, I. & Sowerby, A. 2004 *The response of soil processes to climate change: Results from manipulation studies of shrublands across an environmental gradient.* Ecosystems 7:625-637.

Penuelas, J., Gordon, C., Llorens, L., Nielsen, T., Tietema, A., Beier, C., Bruna, P., Emmett, B., Estiarte, M. & Gorissen, T. 2004 *Non-intrusive field experiments show different plant responses to warming and drought among sites, seasons and species in a North-South European gradient.* Ecosystems 7:598-612.

Work Package 2:
Impacts, Recovery and Processes

Task 9:

Evaluating the Impacts of Environmental Change Using Archive Vegetation Data

R Helliwell and A Britton

MLURI

Task 9. – Evaluating the Impacts of Environmental Change Using Archive Vegetation Data

R Helliwell and A Britton

MLURI

1. Summary

During 2005/6 the main objective for this task was to carry out further fieldwork, re-recording points with archive vegetation data. Following the successful pilot phase in 2004/5 which showed that the original survey points could be accurately re-located in the field, a sampling strategy was drawn for the following two field-seasons. This identified four areas with a high concentration of archive data points, which were contrasting in terms of land-use intensity and pollutant inputs and contained a range of plant communities. These four areas were the Cairngorms, Southern Uplands, the Isle of Mull and the Orkney and Shetland islands. Re-sampling will be focused on these four areas. In 2005/6 a total of 95 sample points were re-located and re-recorded on Mull (34) and in the Southern Uplands (61). An initial, basic analysis of the data collected so far shows that a shift in the vegetation composition can be detected across all the habitats sampled, but that there is some evidence that the shift may be greatest in snowbed communities and *Racomitrium* heath.

2. Policy Relevance

This work aims to provide evidence for the impact of nitrogen deposition on potentially sensitive montane communities which are of high conservation value but are relatively little researched. It also aims to explore the relative roles of nitrogen deposition, climate and land management in driving observed vegetation changes.

3. Project Update

3.1 Sampling design for re-survey of archive data

A total of 677 montane vegetation records are available within the archive dataset. These are dispersed across Scotland's mountain regions and it is not practicable to re-survey them all within the time available. In order to develop a feasible re-survey strategy, those areas with high concentrations of records were identified, which would allow re-recording of the maximum number of sample points within three summer seasons. Four areas were selected to be targeted for re-sampling. These were: The Cairngorms National Park, the Southern Uplands, the Isle of Mull and the Orkney and Shetland Islands. These four areas cover the geographic range of Scottish montane areas and provide a contrast of land-use intensity, climate and pollutant inputs. All four areas share common community types, including montane dwarf shrub heaths, grasslands, moss heaths and snow beds, thus allowing comparisons between them.

3.2 Survey work 2005

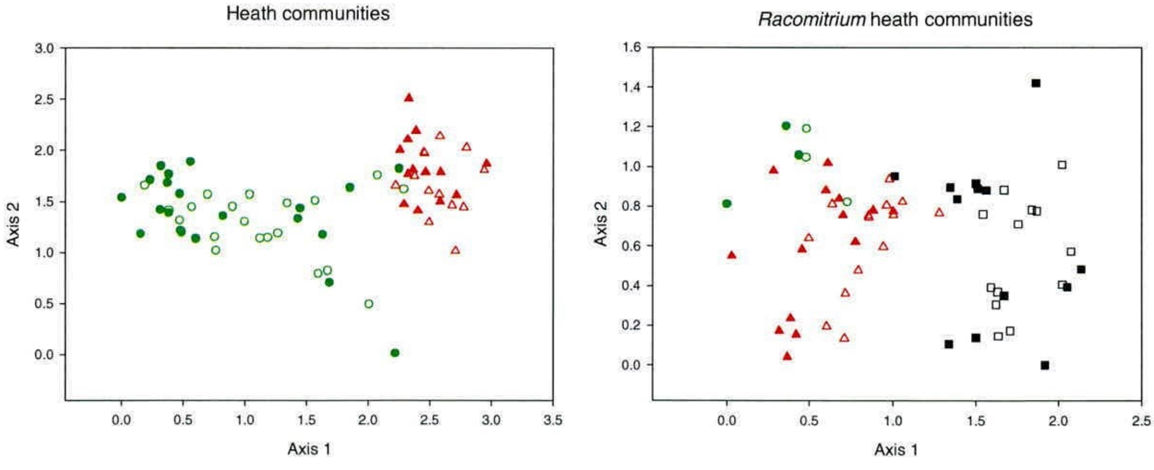
The Southern Uplands and Mull were the focus of fieldwork in summer 2005 with four surveyors working in each area for 10 days. As with the survey points in the Cairngorms in 2004, there were few problems with achieving satisfactory re-location of stands within a range of +/- 50m. In total 95 sample points were re-located and re-recorded including 34 on Mull and 61 in the Southern Uplands giving a total to the end of 2005 of 135 samples. Work to confirm the identification of difficult specimens (bryophytes and lichens) in the laboratory is ongoing, with confirmation provided by staff at the Royal Botanic Gardens Edinburgh.

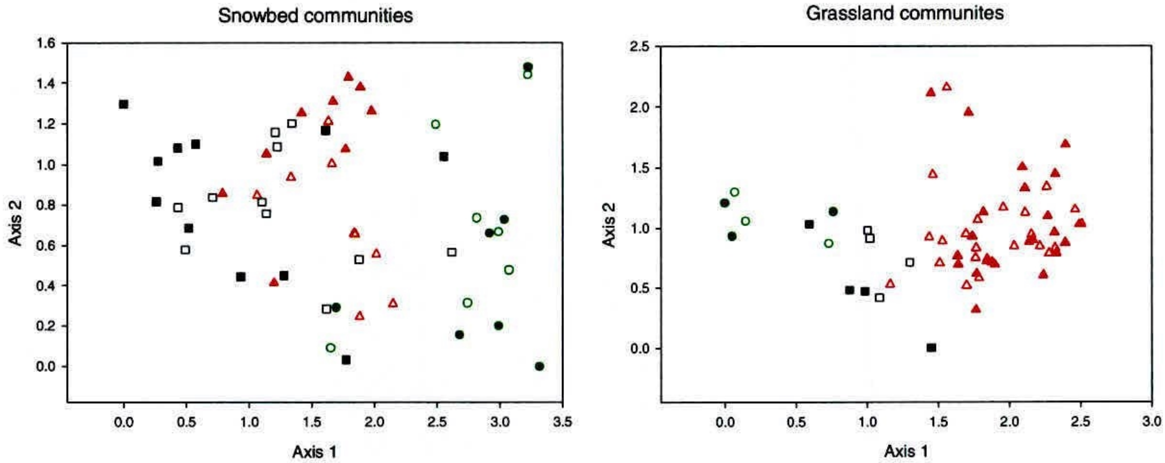
4. Key Findings

4.1 Initial analysis of 2004 and 2005 data

A preliminary analysis of the data collected so far was carried out in early 2006. Detrended Correspondence Analysis (DCA) was used to explore changes in community composition between the archive and re-surveyed data points based on higher plant species and a small number of key, easily identified lichen and bryophyte species. The ordination diagrams (Fig. 1) show that there appears to have been some change between archive and re-survey data points for all four of the community groups, but that the direction and magnitude of the shift is not always the same between areas. The greatest change appears to have occurred in snowbed (U7, 8, 12) and *Racomitrium* heath (U10) communities. Future analyses will concentrate on establishing which species and species groups have been most affected and on the links between species change and its potential drivers such as N deposition, grazing and climate change.

Figure 1. Multivariate (DCA) analysis of archive (1970's and 80's) and recent (2004/5) montane plant community composition data grouped by community type (heaths, snowbeds, *Racomitrium* heath and grasslands). Each point represents the species composition of one sample. Solid symbols show archive data, empty symbols show re-surveyed data. Green circles – Cairngorms, Red triangles – Southern Uplands, Black squares – Mull.




Snowbed communities
Axis 2
Axis 1
Grassland communites
Axis 2
Axis 1

Work Package 2:
Impacts, Recovery and Processes

Task 10:
Changes in the Flora of Calcareous Grasslands

Mike Ashmore

University of York

Task 10. – Changes in the Flora of Calcareous Grasslands

M Ashmore

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1. Summary

This study aims to address specific gaps in knowledge by assessing whether there is evidence of long term changes in the species composition of calcareous grasslands at different sites across the UK that are spatially associated with modeled nitrogen deposition. The work involves a botanical re-survey of a sample of the network of calcareous grassland sites that were established in the early 1990s by Prof. John Rodwell and colleagues of Lancaster University, with support from Defra, to monitor long-term impacts of air pollution and climate change. Bryophyte species composition was surveyed at 16 of the sites in the summer of 2005. While significant changes in frequency of individual species were found, there was no significant association with modeled nitrogen deposition. Further survey work focusing on higher plants, with associated soil analysis, at a wider range of sites is planned for the summer of 2006.

2. Policy Relevance

Substantial areas of the UK are indicated to be in exceedance of empirical critical loads of nutrient nitrogen, which are largely derived from field manipulation experiments. Evidence that there are real changes in species composition across the UK which are associated with changes in nitrogen deposition is very important in supporting the use of these critical loads, and as evidence that measures to decrease emissions will have significant benefits for habitats of national conservation significance. The research under this Work Package aims to provide new evidence to support critical loads set for on particular habitat of conservation value, calcareous grasslands. The evidence for bryophyte species reported this year is equivocal: while changes in species frequency have occurred since the early 1990s, there is no significant association with modeled nitrogen deposition. However, final evaluation in policy terms will only be possible after the field work planned for 2006.

3. Project Update

The work in 2005 focused, as planned, on the bryophyte flora of calcareous grasslands. The aim was to test whether there is evidence from field sites, with varying rates of N deposition, across the UK, of changes in bryophyte species composition that are consistent with those that we reported at the Wardlow Hay Cop long-term field manipulation experiment under the previous Umbrella contract – specifically changes towards a more nitrophytic and acidophytic bryophyte flora.

The work focused on a national network of permanently marked monitoring plots at 56 calcareous grassland sites, established in 1991-2 and designed to represent the full range of calcareous grassland communities in the UK (Rich *et al.*, 1993). In summary, between one and six 12x12 m plots were set up within each site. Each plot was marked out with tapes at 2m intervals and the vegetation was recorded using 36 0.5x0.5 m quadrats.

A total of 16 sites were re-surveyed, giving a range of modeled N deposition from 12.5 to 35.8 kg ha^{-1} yr^{-1}. Sites were also chosen in order to represent a wide geographical spread and with a dominant National Vegetation Classification of CG2, although CG1 grasslands were included where N deposition was particularly low. Within each of the sixteen sites, one plot was chosen for the re-survey. Other than meeting the NVC requirement, there were no special criteria for choosing the plot. Surveys took place in May and June 2005. A 0.5x0.5 m quadrat was placed as close as was reasonably practicable to the original 36 quadrat locations in turn. All bryophyte species within each quadrat were recorded as presence or absence. Mean soil pH measurements for the whole plot were taken in the field using a portable pH meter with a standard glass electrode at randomly selected points in each of the 36 plots.

Survey work and analysis was completed in the summer of 2005 and a preliminary report of findings was made verbally at the Umbrella meeting in September 2005.

4. Key Findings

Of the 16 sites surveyed, bryophyte species richness had increased at 11 (69%) sites and decreased at 5 (31%) sites (Table 1). A simple linear regression demonstrated no significant associations between species richness and modelled nitrogen deposition or soil surface pH.

When taking the mean frequency change across all sites, for each species, significant differences were found for (Z=-3.255; P<0.01), *Eurhynchium striatum* (Z=-2.659; P<0.05), *Neckera complanata* (Z=-2.385; P<0.05), (Z=-2.393; P<0.05), *Weissia microstoma* (Z=-2.559; P<0.05), and *Weissia spp* (Z=-2.385; P<0.05). The significance disappears for *Eurhynchium striatum* if combined with *Eurhynchium sp.*, for *Neckera complanata*, if combined with *Neckera crispa*, and for *Weissia microstoma* and *Weissia spp.* if combined; therefore, these changes should be treated with caution as they may reflect inconsistency in species identification between the two surveys. However, significance increases in frequency remain for *Calliergon cuspidatum* and *Rhytidiadelphus squarrosus*. Of the species that had sufficient points to be able to conduct a simple regression analysis, none exhibited a statistically significant pattern, in terms of frequency change over time, to either mean soil surface pH or modelled nitrogen deposition.

Ellenberg indicator values for British bryophytes, relating to fertility (iN), were applied to each species found at each site and a mean Ellenberg value was calculated for both survey years at each site. Of the 16 sites surveyed, the average Ellenberg Index values for fertility had increased at 9 (56%) sites and decreased at 7 (44%) sites (Table 1). One needs to bear in mind that not all species recorded have an assigned Ellenberg value. A simple linear regression demonstrated no significant (F=2.673; P>0.05) association between changes to the mean Ellenberg Index fertility values and modelled nitrogen deposition. The trend-line shown in Figure 2 suggests, however, that a nitrogen response may be evident.

Since these results are indicative of a change in species composition associated with higher levels of N deposition, a change to the details of the field survey planned for 2006 has been agreed, to gain further information about species composition on these sites. The same 16 sites will be re-surveyed to obtain data on higher plant species composition and soil samples will be taken adjacent to the plots for C:N determinations. A further 9 plots will be added at the high and low end of the available gradient of N deposition in the original sites. This should provide a greater power to detect and interpret impacts of N deposition, although it is necessary to include a wider range of CG communities to achieve this goal.

5. References

Rich TCG, Cooper EA, Rodwell JS & Malloch AJC (1993). Effects of climate change and air pollution on British calcicolous ecosystems. Final report to UK Department of the Environment.

					Species Richness		Ellenburg (iN) Values	
Site	Grid Ref	NVC	N Dep	pH	1990-93	2005	1990-93	2005
Aston Rowant	SU 726 973	CG2a	21.1	7.92	10	8	2.92	3.02
Barton Hills	TL 089 297	CG2a	26.7	7.9	8	10	3.17	3.29
Boxhill	TQ 177 518	CG2a	19.2	7.89	9	10	2.83	2.86
Cressbrookdale	SK 174 743	CG2d	31.1	7.83	16	13	2.78	2.9
Crooks Peak	ST 392 558	CG1d	19	7.88	13	11	2.72	2.76
Ellerburn Bank	SE 853 849	CG2d	21.1	7.92	10	11	2.72	3.06
Hambledon Hill	ST 847 122	CG2a	27.6	7.88	10	12	2.85	2.76
Hog Cliff	SY 615 975	CG2b	24.2	7.9	5	7	3.14	3.1
Kingley Vale	SU 822 117	CG2a	16.9	7.89	3	9	3.19	2.99
Knocking Hoe	TL 131 309	CG2b	21.8	7.91	5	6	3.27	3.19
Monksdale	SK 116 738	CG2d	35.8	7.78	12	13	2.53	2.82
Mount Caburn	TQ 444 088	CG2b	17.4	7.89	3	7	4.97	3.26
Old Winchester Hill	SU 641 204	CG2a	19.6	7.97	8	7	3.17	3.08
Pewsey Downs	SU 121 635	CG2b	23.8	7.9	1	5	n/a	3
The Great Orme	SH 762 831	CG1d	12.5	7.91	13	11	2.75	2.7
Wye	TR 085 445	CG2b	20.4	7.84	6	8	3.27	3.52

Table 1. Summary of the modelled N deposition, pH measurements, species richness response and change to the Ellenburg (iN) values for each site.

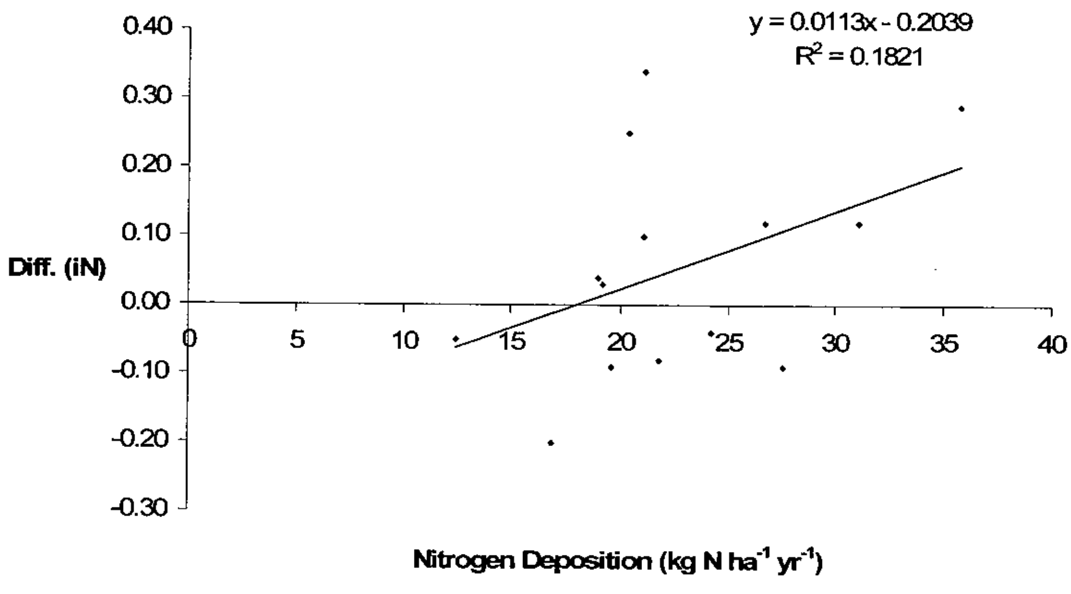

Figure 1. Mean bryophyte Ellenburg (iN) response to N deposition.

Work Package 2:
Impacts, Recovery and Processes

Task 11:
Factors Controlling Nitrate Breakthrough at the Landscape Scale

M.S. Cresser and S. Green

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Task 11. – Factors Controlling Nitrate Breakthrough at the Landscape Scale

M.S. Cresser and S. Green

University of York

1. Summary

Two models have been developed that allow prediction of seasonal variations in nitrate concentrations in river water for unmanaged or minimally managed catchments across the UK. These will allow prediction of long term changes in nitrate leaching in response to changes in N deposition. Winter maximum and summer minimum nitrate concentrations have been shown to be related primarily to deposition flux of oxidized N species, and are apparently not significantly influenced by precipitation amount. However, annual precipitation has been shown to be a key driving variable in the prediction of high flow and low flow calcium and alkalinity concentrations across the UK, in accordance with a hypothesis based upon intuitive expectation.

Work on the effects of road-salting upon the disruption of N cycling in upland soils has continued. Effects on the N cycle attributable to impacts of past increased solubilization of soil organic matter and associated changes in soil cation exchange capacity, elevation of pH, and salinity of soil solution have been hypothesized. The hypotheses are a consequence of time series data gathered for soils and soil solutions along altitudinal transects below roads and away from roads. Effects on N cycling are sustained and substantial.

Preliminary results on assessment of N cycling at the Hob Moor Local Nature Reserve in York indicate that N deposition has resulted in soil N accumulation, low soil C:N ratios, and mobile ammonium and nitrate in the soils even during periods of active plant growth.

2. Policy Relevance

To be robust, pollution abatement or management strategies for N deposition must be based upon sound understanding of how a diverse range of plant/soil/water systems respond to different N species deposited upon over a wide range of levels of deposition and under different climatic conditions. It is not possible to explain impacts of N deposition upon plant biodiversity or surface water quality with any confidence over diverse time scales until we can confidently predict their effects upon soils. This task is aimed at improving our quantitative understanding of the absolute and relative importance of key soil processes in the N cycle.

- We need to know just how much N pollution soils can store before they start to leak N species, and what controls this storage capacity.
- We need to know how land use and management influences the capacity of soils to deal with N pollution.
- We need to know what forms the leaked N will be in, under what conditions will it leak, and what concentrations of each species will occur as a result in associated surface waters.

- We need to know how N inputs will disrupt the cycling of other elements in soils, and especially carbon cycling.

3. Project Update

3.1 Objectives for Task 11

The objectives for task 11 stated in the proposal were:

- To evaluate the chemical impacts of N deposition on soils.
- To evaluate the interaction(s) with management
- To assess the processes controlling nitrate leaching from soils to freshwaters and the rate of leaching for use in dynamic models.

This component of the Umbrella Contract research aimed to answer the question: *What are the key factors controlling nitrate breakthrough and leaching from soils?* It also aimed to contribute to answering the questions: *Can indicators be identified for N status of soil/plant systems and N saturation?* and *What indicators are appropriate to gauge critical load exceedence for nutrient nitrogen?* The aim therefore was to improve our quantitative understanding of how N species input fluxes interact with other catchment characteristics at the landscape scale to influence the capacity of soils to store N inputs, and thus N species leaching (especially nitrate leaching).

3.2 Progress since June 2005

3.2.1 Progress with Taking Regional Variations in Climate into Account

In last year's report we noted that we had been modifying the G-BASH model so that it could be used to predict, for both mean and high flow conditions, the alkalinity and calcium concentration in any river in GB uplands. This was done to test the hypothesis that a simple dilution concept could be used quite simply to account for national-scale variations in annual precipitation and evapo-transpiration. The model was further developed this year to show that acid deposition also has to be taken into account (we used National Network Data) when predicting alkalinity to correct for partial neutralization. A description of the development and validation of the new model has very recently been published in *Environmental Pollution* (Cresser *et al.*, 2006). We have since tested the same concept in the further development of our nitrate model for predicting spatial and temporal variation in nitrate leaching from soils to rivers in upland drainage basins. Initial investigation suggests that this simple dilution effect is <u>not</u> a major factor regulating nitrate concentrations in river waters following leaching from upland catchments.

3.2.2 Progress with the Soil Nitrate Leaching Model

Last year we also flagged that we had developed a novel nitrate leaching model to predict the spatial variation <u>and</u> the temporal variation in nitrate leaching throughout the heavily N-impacted Nether Beck catchment in the Lake District (Calver *et al.*, 2004; Smart *et al.*, 2005). Prediction for this relatively modestly sized area was very good, but N species deposition fluxes and precipitation did not feature in the model, making successful application of such a model at regional and national scales unlikely. Moreover, as it stood the model could not be applied to predict long-term (decades) changes in nitrate leaching in response to changes in N deposition loads.

We have extended the model this year to take spatial variation in N deposition and precipitation at national scale into account, using data from AWMN catchments (data provided by Chris Evans from CEH). A thorough statistical investigation indicated that flux of oxidized N species deposited at a site was the major factor driving to minimum (summer) nitrate concentration found in rivers and the amplitude of the seasonal trend (the maximum winter concentration) in nitrate. The initial model was based upon a novel empirical truncated cosine function of day number to account for seasonality (Smart *et al.*, 2005). We also have been developing an alternative model with CEH using quadratic functions of month number from 1 to 12 for February to January. The constants in the quadratic derived for each catchment can again be best predicted from the catchment's oxidized N deposition flux.

3.2.3 Progress with the Soil Profile N Storage (SPNS) Work

We mentioned in our previous report that we had extended our work on the factors regulating soil profile nitrogen storage (SPNS) capacity. Our first publication on this, in *Water, Air and Soil Pollution*, was very significant in that it showed that SPNS in upland podzol soil profiles within specified altitude and slope constraints could be modeled from pollution N inputs and annual precipitation. A paper on quantification of SPCS using Scottish Soil Survey Data has been revised and recently resubmitted to the *European Journal of Soil Science*. Our interest in prediction of SPCS stems from the importance of organic matter content to the capacity of soils to store N.

We have recently been doing additional work on SPCS and SPNS at Hob Moor, a Local Nature Reserve that is an area of permanent "ancient", supposedly low nutrient status grassland in York. The management strategy at this site is failing because of the high atmospheric N inputs. Soil C:N ratios are low, often being below 10, while the P status is low, and as a consequence both ammonium and nitrate are mobile in this soil even during periods of active growth. The Hob Moor Nature Reserve is of particular interest because it contains areas of more acidic and freer draining soils alongside more predominant, circum-neutral heavier textured soils. There are significant differences in C:N ratio (higher in the more sandy soils) between these two groups of soils.

3.2.4 Progress with Research on the Disruption of Soil N Cycling by Road Salting

We mentioned in last year's report that a PhD student, Sophie Green (50% DEFRA-50% University of York-funded) had started work on a new sub-programme evaluating the importance of the disruption of the nitrogen cycle by road salting. Salting and gritting can have a major effect on mobility of N species within and from soils in UK uplands. Although both sampling/ analysis and data interpretation are still underway, some preliminary data is reported here that shows how important this practice is likely to be to N species leaching in affected catchments. When Professor Mike Hornung was steering the previous Umbrella contract we were requested to establish why catchments such as that of the R. Etherow apparently gave anomalously high rates of nitrate leaching. Our preliminary assessment suggested that road salting effects on mobilization of soluble organic N and ammonium, and on nitrification and nitrate utilization rates, could be a major factor. We will concentrate our efforts in this area in the final year of the contract. Figure 1 below illustrates how road salting runoff, especially near direct drain outlets to down-slope soils, pushes up the sodium and magnesium dominance of cation exchange sites and lowers exchange acidity. Soils were sampled in March, towards the end of a long period of salt application at the site. Figure 1 also provides clear evidence of reduction in ammonium retention and/or ammonification in salt-impacted areas. This agrees with our conclusions based upon the KCl-extractable ammonium data in our previous annual report.

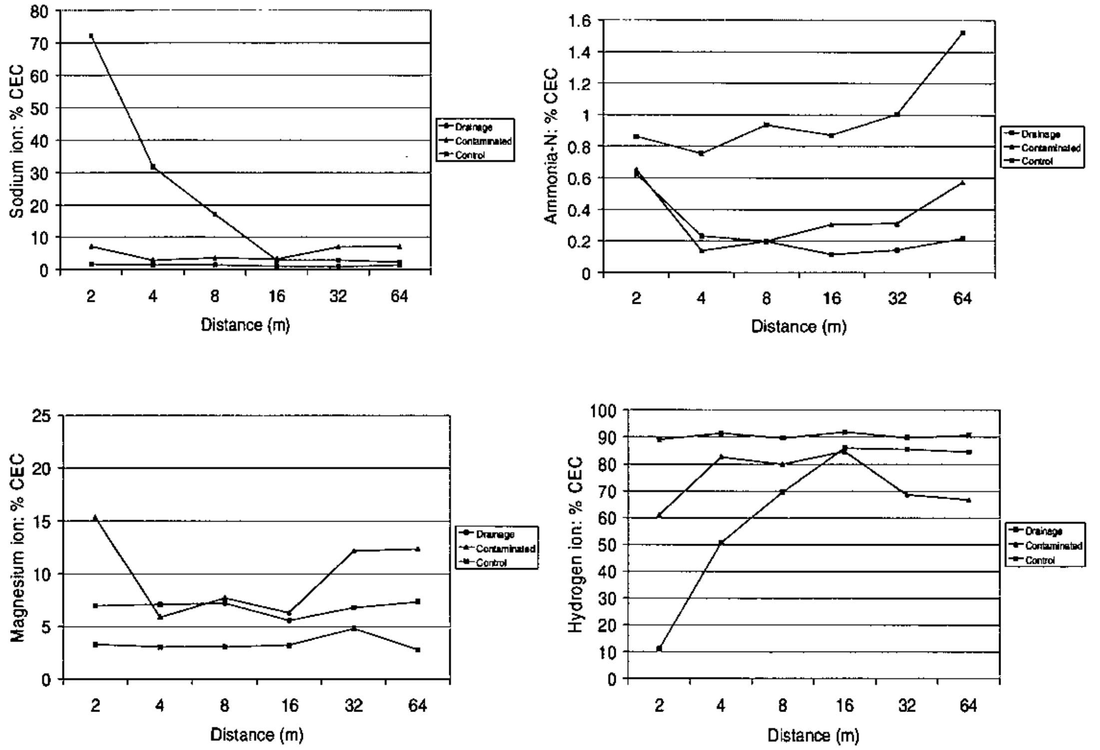


Figure 1. **Change in proportion (%) of CEC sites occupied by sodium, ammonium, magnesium and hydrogen, March 2005, as a function of distance (m) from a wall adjacent to the A6 near Shap.**

When soils were re-sampled and re-analysed in October 2005, after several months with no road salting, recovery of the soils was incomplete (Fig. 2), so impacts on the N cycle will be very substantial and sustained.

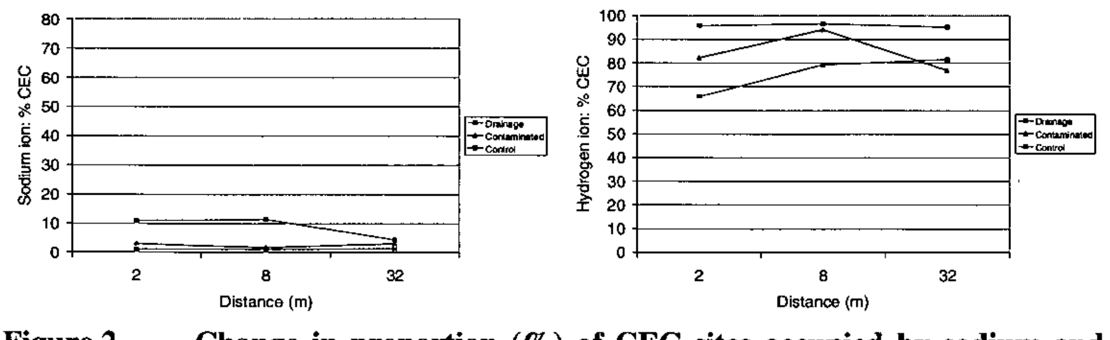
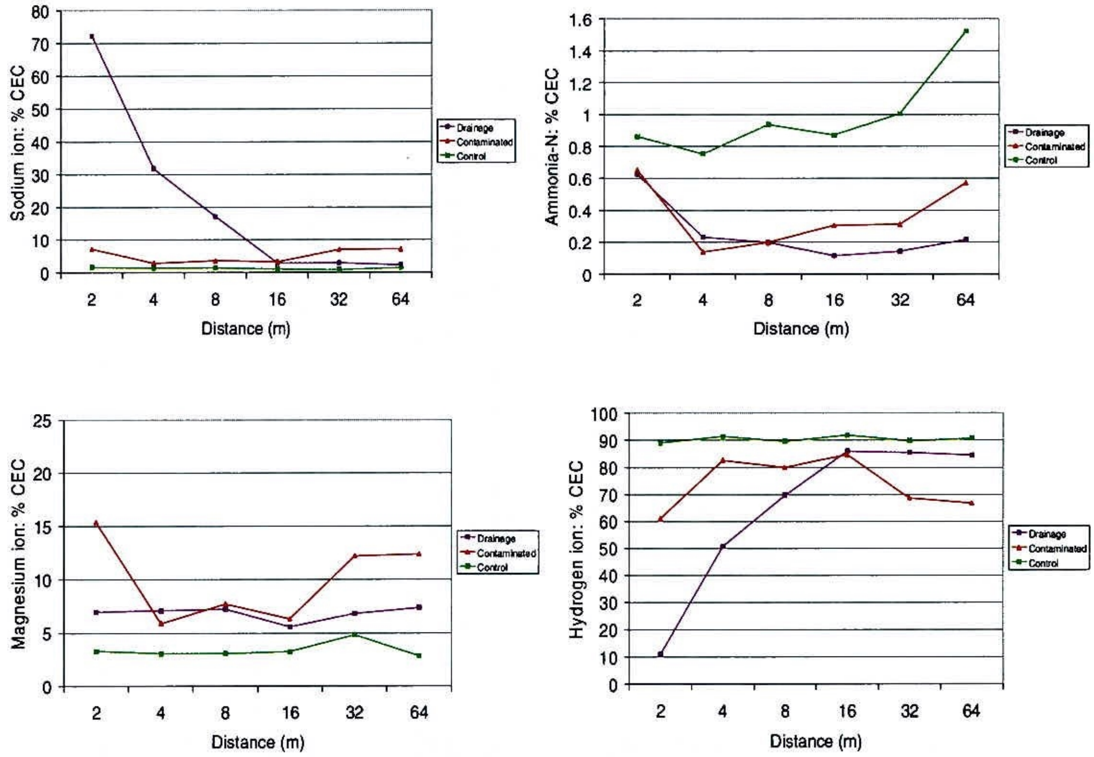


Figure 2. **Change in proportion (%) of CEC sites occupied by sodium and hydrogen, October 2005, as a function of distance (m) from a wall adjacent to the A6 near Shap.**

One especially interesting finding of the study to date was that loss on ignition of soils subject to road runoff was much reduced over the first 8 m distance from the and magnesium dominance of cation exchange sites and lowers exchange acidity. Soils were sampled in March, towards the end of a long period of salt application at the site. Figure 1 also provides clear evidence of reduction in ammonium retention and/or ammonification in salt-impacted areas. This agrees with our conclusions based upon the KCl-extractable ammonium data in our previous annual report.



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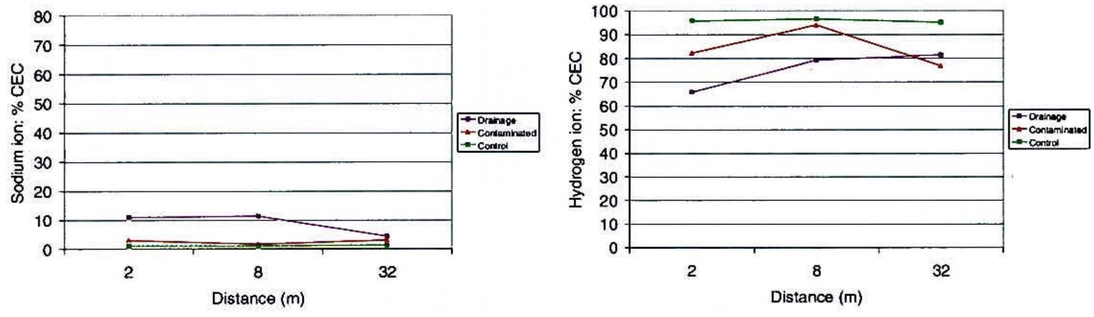


Figure 2. **Change in proportion (%) of CEC sites occupied by sodium and hydrogen, October 2005, as a function of distance (m) from a wall adjacent to the A6 near Shap.**

One especially interesting finding of the study to date was that loss on ignition of soils subject to road runoff was much reduced over the first 8 m distance from the wall. We had expected that mobilization of organic matter could be an important mechanism for transport of N as organic N from soils to rivers. However it appears from soil solution analysis along transects that solubility of organic matter close to the wall is in fact very low, in spite of the elevated soil pH and high sodium saturation. This suggests that formation of fresh plant litter is reduced by the salinity (up to 6000 μg Na ml^{-1} in soil solution compared with ca. 5 μg Na ml^{-1} for control transects), and any mobile humic substances have long since been lost from the soil, being transported down slope. Further examination of this very extensive time-series data set is now underway, but it also appears that maximum ammonium concentrations in soil solutions of polluted soils are at ca. 32 m from the road. This probably reflects optimal pH conditions for mineralization of organic N, optimal organic matter inputs, and maximum effect of down-slope mobilization of ammonium. However lower soil organic matter content close to the road results in lower cation exchange capacity, implying that equilibration of exchange sites with ammonium inputs would occur more rapidly, facilitating ammonium leaching down slope. Similar considerations could apply to calcium and magnesium contributions to base saturation, assuming weathering rate is approximately constant down slope

Calcium concentrations in soil solution were indeed very significantly enhanced close (2 – 4 m) to the road as a consequence of road runoff, or the mechanism proposed above, or both. This could be a consequence of grit transport and deposition or of soluble calcium in the road salt, and this will be investigated further. Magnesium showed similar trends over the autumn/winter study period.

Extractable nitrate concentrations in soils were very significantly increased overall as a consequence of road runoff, as were nitrate concentrations in soil solution (in spite of the dilution effect of the extra water inputting the soil locally). Soil solution concentrations were appreciably higher than concentrations of nitrate in precipitation.

4. Collaboration with Dynamic Modelling Umbrella and Freshwater Umbrella

Links remain similar to those flagged in last year's report. However:

1. Progress has been made with our model for spatial and seasonal and long-term temporal variation in nitrate leaching from upland soils, taking into account the spatial variation in N species deposition fluxes. Spatial variation in precipitation at regional and national scales has been shown to be less important for nitrate concentrations in upland rivers. This work is still directed at providing robust prediction of dynamics of nitrate losses from soils under diverse soil, climatic and N deposition conditions. As mentioned last year, it should prove to be of great value in the context of providing inputs to other dynamic models.

2. Quantification of how landscape and other soil formation factors influence SPNS capacity of different soil types to store N is invaluable in dynamic modeling because it sets limits on when substantial changes in quality of soil drainage water can be expected to occur over longer time scales.

3. Establishing the time scales over which road salting effects occur and the magnitude of their impacts upon N cycling in upland areas is crucial for dynamic models to encompass road salting effects as part of their attempt to quantify land use effects in dynamic models.

5. Key Findings

- We have developed two novel models for predicting spatial and temporal variations in nitrate concentrations in rivers at a national scale.
- We have established that oxidized N deposition is a key determinant of nitrate leaching from upland soils and nitrate winter maximum and summer minimum concentrations in rivers across the Acid Waters Monitoring Network sites.
- We have shown that road salting appears to reduce organic matter content of roadside soils, reducing DOC and increasing ammonium and nitrate concentrations in associated drainage waters. It also can substantially increase soil pH and decrease (via loss of organic matter) CEC.
- Road salting effects persist throughout the year, being reduced, but not eradicated, by the autumn.
- At the Hob Moor Local Nature Reserve, atmospheric N deposition has lowered soil C:N ratios to very low values, resulting in high extractable nitrate and ammonium concentrations even during periods of active plant growth.

6. References

Calver, L., Smart, R., McClean C. and Cresser, M.S., Modelling the spatial distribution of soil chemical characteristics and associated drainage waters: A case study in the Lake District, *Proceedings of the GIS Research UK 12^{th} Annual Conference*, UEA, Norwich, 18^{th} – 30^{th} April, 389-392 (2004).

Cresser, M.S., Ahmed, N., Smart, R.P., Arowolo, T., Calver, L.J. and Chapman, P.J., Predicting Gran alkalinity and calcium concentrations in river waters over a national scale using a novel modification of the G-BASH model, *Environmental Pollution*, **143:**361-366 (2006).

Crowe, A.M., Sakata, A., McClean, C. and Cresser, M.S., What factors control soil profile nitrogen storage? *Water, Air & Soil Pollution: Focus*, **4:**75-84 (2004).

Smart, R.P., Cresser, M.S., Calver, L.J., Clark, M. and Chapman, P.J., A novel modelling approach for spatial and temporal variations in nitrate concentrations in an N-impacted UK upland river basin, *Environmental Pollution*, **136:**63-70 (2005).

Work Package 2:
Impacts, Recovery and Processes

Task 12:
Investigation of Long-Term Soil and Water Dynamics in Sensitive Lake District Catchments

E Tipping, SA Thacker, D Wilson & J Hall

CEH Lancaster

Task 12. – Investigation of Long-Term Soil and Water Dynamics in Sensitive Lake District Catchments

E Tipping, SA Thacker, D Wilson & J Hall

CEH Lancaster

1. Summary

There is concern over the possibility of soil nitrogen saturation, coupled with "nitrogen breakthrough" and nitrate acidification in upland UK catchments, given that there may be a limit to the ability of the plant-soil system to take up and retain atmospherically-deposited N. In the present project, we carried fieldwork and modelling to enhance existing long-term (50-year) data showing increasing nitrate levels in three oligotrophic Cumbrian lakes. The Lake District is probably one of the most sensitive UK upland sites to the effects of nitrogen deposition, because the deposition is high (c. 35 kgN ha^{-1} a^{-1}) and the soils are organic-rich but thin.

During the period of the study, CEH colleagues reported nitrate data for 2005, extending the available information on trends to the period 1955 to 2005. After removing a few outliers from the 1950s data, the upward trends in all three lakes are highly significant ($p < 0.01$). For Wastwater, the average annual increase in [NO_3] was 0.29 μM ($r^2 = 0.67$), while the increases for Buttermere and Crummock Water were both 0.18 μM ($r^2 = 0.50$ and 0.76 respectively). However, the increases have not been steady, and there has been little change over the past 20 years.

Sampling of streamwaters in the catchments of the lakes showed (a) that nitrate concentrations are not significantly affected by the small areas of improved grassland in the valley bottoms, and (b) that brown podzolic soils release less nitrate than rankers.

We sampled 22 complete soil profiles in the catchments of the lakes, and determined C and N pools. The C pool of the 11 ranker soils was 11.2 (sd 4.6) g m^{-2}, while that of the 11 brown soils was 7.8 (sd 5.0). However, the N pools were more similar, leading to lower C:N ratios for the brown soils, 11.9 (sd 1.9) g g^{-1}, than for the rankers, 16.7 (sd 2.9) g g^{-1}). The combined results for streams and soils indicate that nitrate leaching is not directly related to C:N ratio, as has been found for forest soils, and proposed for UK moorland soils.

We are currently carrying out budget modelling based on inputs of N from long-term atmospheric deposition, and the nitrogen and carbon pools of the two main soil types (brown and ranker). Preliminary results indicate that the N pools in the soils can be accounted for in terms of deposition and leaching, primarily of nitrate and dissolved organic nitrogen.

2. Policy Relevance

The results will establish the necessary baseline for continued soil and water monitoring of N build up in these sensitive catchments. This provides a good means of monitoring the effects of emission control policy.

Preliminary modelling results suggest that the information obtained from this study can be exploited to improve the modelling of soil-water N dynamics at other locations in the UK. This could lead to a better explanation of observed nitrate leaching, and therefore forecasting of extents and timescales of change in N dynamics, in response to emission control measures.

3. Project Update

This work was funded for just one year (2005-6). We have successfully completed the field work, and are currently working on the modelling. We expect to be able to present the final conclusions from the study at the TU annual meeting in September 2006, and to submit a paper for publication before the end of 2006.

4. Collaboration with Dynamic Modelling Umbrella and Freshwater Umbrella

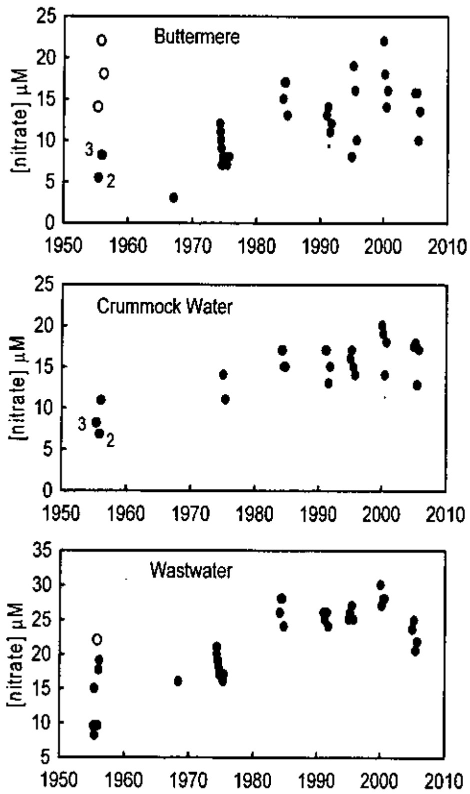
No direct collaboration has yet been undertaken. However, our results should contribute significantly to both Dynamic Modelling and Freshwater nitrogen studies, because they provide new insight into the soil and water behaviour of atmospherically-deposited N. Moreover, we believe that our modelling approach, based on soil N pools and soil properties, as well as C:N ratios, is superior to current modelling efforts. Our work could benefit from insights obtained in Freshwater studies about annual variations in N behaviour, for example those connected with the NAO (Monteith et al. 2000). Thus, there is considerable potential for future collaboration.

5. Key Findings

5.1 Lake water nitrate concentrations

These figures show the increasing concentrations of nitrate in the three lakes. The four points shown by open circles are rejected, on the grounds that such high values are inconsistent with the majority of the data, bearing in mind the relatively long residence times of the lakes. We then obtain the following results for annual average trends, expressed as $\Delta[NO_3]/\Delta t$ in $\mu M\ yr^{-1}$;

	n	$\Delta NO_3/\Delta t$	r^2
Buttermere	36	0.18	0.50
Crummock Water	29	0.18	0.76
Wastwater	41	0.29	0.67



These data sets are the longest runs of results for nitrate in surface waters, for the UK and possibly for the World. Although the long-term trends are highly significant ($p < 0.01$), there has been little change in $[NO_3]$ over the last 20 years.

5.2 Possible effects of improved (fertilized) grassland

We measured nitrate concentrations in streams draining sub-catchments with different areas of improved grassland. The results showed that for Buttermere and Wastwater, the effects of improved grassland can indeed be neglected. In the case of Crummock Water, there may have been a small influence from one part of the catchment, but this can be quantified. The overall conclusion is that nitrate losses from improved pasture have not caused the observed increases in the lake water nitrate concentrations.

5.3 Soil nitrogen and its release as nitrate

We sampled 22 complete soil profiles in the catchments of the lakes, and determined C and N pools. The C pool of the 11 ranker soils was 11.2 (sd 4.6) g m^{-2}, while that of the brown soils was 7.8 (sd 5.0). However, the N pools were more similar, leading to lower C:N ratios for the brown soils, 11.9 (sd 1.9) g g^{-1}, than for the rankers, 16.7 (sd 2.9) g g^{-1}). Concentrations of nitrate in streams draining sub-catchments dominated by the ranker soil were several times higher than in brown soil-dominated sub-catchments. Taken together, the results show that C:N ratio is not a good guide to nitrate leaching, as has been found for forest soils (Gunderson et al. 1998) and proposed for use in UK moorland soils (Jenkins et al. 2001).

5.4 Modelling N budgets

Preliminary modelling results indicate that N deposition over the past several hundred years can be balanced by accumulation in soil, coupled with leaching in stream water, principally as nitrate and dissolved organic nitrogen.

6. References

Gundersen P, Callesen I, de Vries W 1998 Nitrate leaching in forest ecosystems is related to forest floor C/N ratios. Env Poll 102 S1 403-407.

Jenkins A, Ferrier RC & Helliwell RC 2001 Modelling nitrogen dynamics at Lochnagar, N.E. Scotland. Hydrol. Earth Sys. Sci. 5, 519-527.

Monteith DT, Evans CD & Reynolds B 2000 Are temporal variations in the nitrate content of UK upland freshwaters linked to the North Atlantic Oscillation? Hydrol. Proc. 14, 1745-1749.

Work Package 2:
Impacts, Recovery and Processes

Task 13:
Wetlands as Nitrate Regulators

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Task 13. – Wetlands as Nitrate Regulators

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1. Summary

The first phase of this Task is near completion. The conventional ^{15}N pool dilution technique had to be redeveloped for meaningful application in the often anaerobic environments of peaty wetland soils. ^{15}N pool dilution studies have been made at a wetland catena (Plynlimon, Wales) on three occasions in the past 18 months. The data are currently being processed and early analyses suggest that there is significant difference in the gross soil nitrogen fluxes between ombrotrophic, rheotrophic and mieneratrophic wetland soils. It is notable, however, that seasonal differences are less pronounced than intersite variability. A full data analysis will be made once all four data sets are intact.

2. Policy Relevance

Wetlands represent ecologically and agriculturally significant components of the rural landscape often containing rare species of plants and invertebrates that support unique bird and wildlife populations. In the UK, wetlands occupy large areas of marginal and upland landscapes that are characterised by nutrient paucity and low productivity (either unmanaged, grazed or maintained for game birds). The diversity of marginal-upland wetlands, from *Sphagnum* dominated blanket peat (designated as Priority habitats by the UK Biodiversity Action Plan) to waterlogged *Juncus/Agrostis* acid grassland (many designated as EU Special Protection Areas for their provision of habitat for rare birds such as yellow wagtails), is a reflection of differences in their biogeochemistry and biodiversity e.g. ombrotrophic/blanket peatlands (Countryside Vegetation System 94-100), valley bottom peat (CVS 86-88) , marshes and rushy acid grasslands (CVS 51-55), linear riparian grasslands/marshes (CVS 41, 48). The attribution of climate and seasonality as a driver of changes wetland functions and services is clearly an important issue and will become increasingly so throughout the 21st Century (IPCC; Soil Strategy for E&W; Environment Agency State of Soils).

Despite the ecological and landscape significance of these ecosystems, their role in landscape nitrogen cycling remains poorly understood. A greater understanding of the nitrogen dynamics in these wetlands is required to determine their role as regulators of nitrogen accumulation and release as breakthrough NO_3 into catchment streams. Scientific evidence is required to ; 1. to develop better landscape scale models to predict the effects of nitrogen deposition on terrestrial nitrogen dynamics and biodiversity and 2. to underpin landuse policy decision making with respect to changes EU agricultural subsidy payments to upland agri-businesses to manage landscape nitrogen dynamics.

3. Project Update

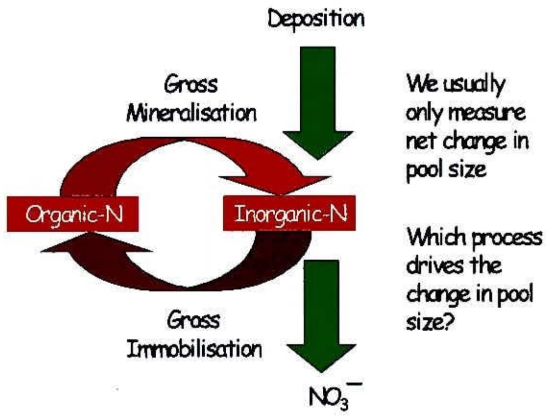
Background

Wetland N regulation

In the UK, wetlands occupy large areas of marginal and upland landscapes that are often characterized by nutrient paucity and low productivity. Consequently these 'bottlenecks' of nitrogen biogeochemistry are thought to play key role in nitrogen cycling at the ecosystem, catchment and landscape scales. Nitrogen release from organic matter rich upland ecosystems typical of much of the UK is ultimately determined by the net balance between nitrogen supply and biological demand. Gross nitrogen supply is composed of agricultural and atmospheric inputs whereas biological demand is dominated by microbial immobilization and plant uptake. Excess of supply over biological demand can result in net nitrogen losses as nitrate (NO_3^-) breakthrough or in some cases as N_2O from nitrification and/or denitrification.

^{15}N pool dilution approach

Figure 1. Soil N pathways and gross fluxes



Standard methodologies enable a measure the net change in the two microbial transformations processes which affect soil inorganic nitrogen supply (ammonification + nitrification = mineralization) and microbial demand (immobilisation) (Figure 1). Using this approach is, however, problematic as changes in inorganic N concentrations and net fluxes may obscure changes or differences in the contributory gross fluxes e.g. an increase in net inorganic nitrogen supply may be due to an increase in the gross mineralization rates and / or a decline in gross immobilisation rates by the soil microbes (Schimel, 1996).

The application of ^{15}N techniques has already clarified that different ecosystem N biogeochemistry varies considerably (Hughes, Emmett et al. 2005). However, there has, as yet, been no attempt to examine the relative importance of landscape and seasonality on these gross N fluxes. The assessment of the effect of these factors is all the more important considering that the primary regulators of ecosystem nitrogen transformations are biological and therefore potentially sensitive to changes in the associated abiotic conditions.

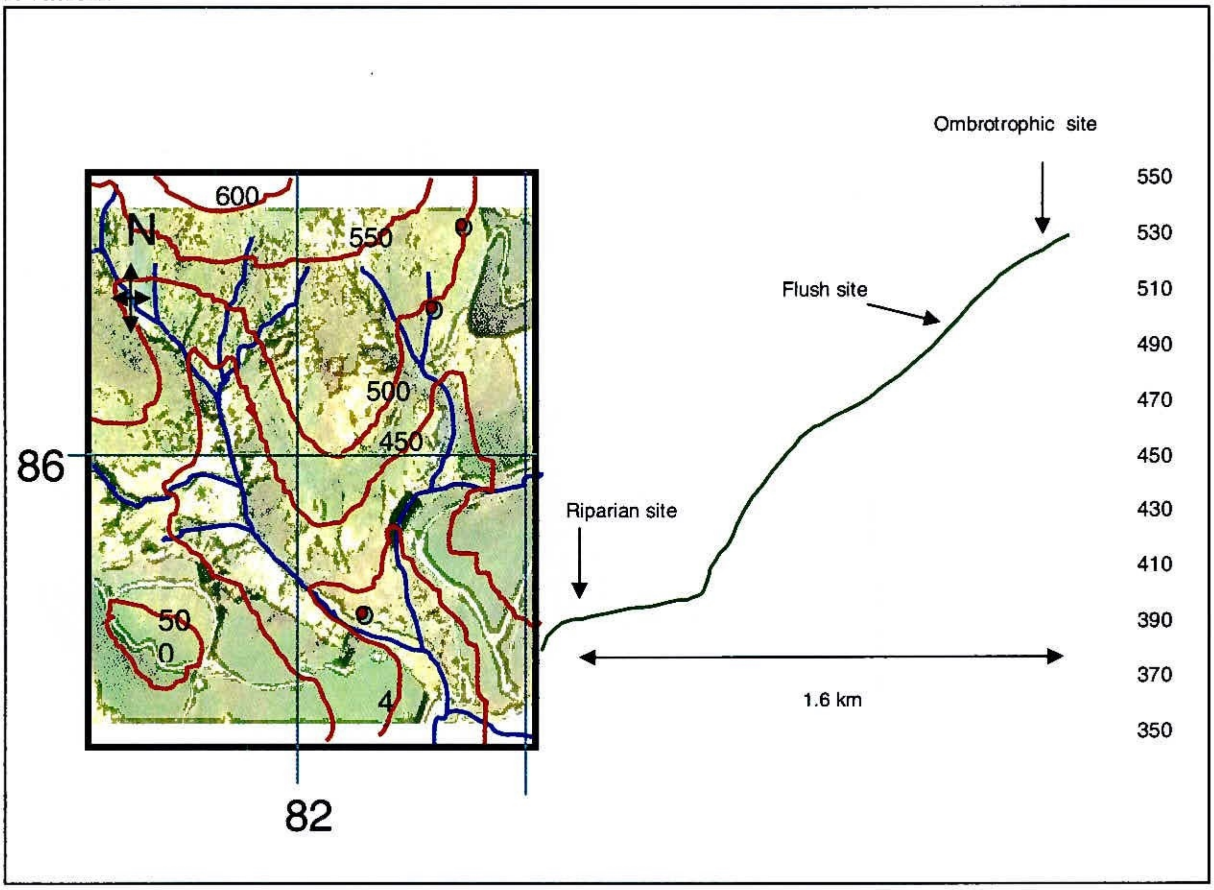
Project aims

The aim of this task is to separately quantify these two gross nitrogen transformation processes in a catena of 3 wetland soils located within the intensively studied N catchment at Plynlimon (SN827868) experimental sites supported by the NERC-DEFRA Terrestrial Umbrella. A gradient of three marginal-upland wetland ecosystems; 1. blanket peatland, 2. riparian flush and 3. a valley bottom bog are to be studied on a seasonal cycle. The ^{15}N pool dilution method has been optimized for application in wetland peats and waterlogged soils to enable the partitioning and quantification gross microbial mineralization and immobilisation processes. Data from this investigation will be made available for landscape scale predictive modeling in MUMBLES.

3.2 Methods

Sites : A catena of wetland units were selected at a field site in Plynlimon in Mid Wales (see Figure 2). The first of was located at the summit of a sub-catchment area at an altitude of approximately 530 meters (asl). This wetland is ombrotrophic and vegetation was dominated by *Calluna vulgaris* with some bryophytes (*Sphagnum* sp and feather mosses) and cotton grasses (*Eriophorum sp*). The second wetland on the site was in a mineratrophic or riatrophic flush, with vegetation dominated by *Juncus effuses* and a mixture of grasses lying at an altitude of roughly 490 meters. The third wetland area was a riparian valley bottom system lting at the foot of the catchment next to a river at an altitude of 390 meters. Vegetation consisted primarily of *Molinea* sp grass.

Figure 2. Location of wetland catena within the Plynlimon catchment, including side elevation.



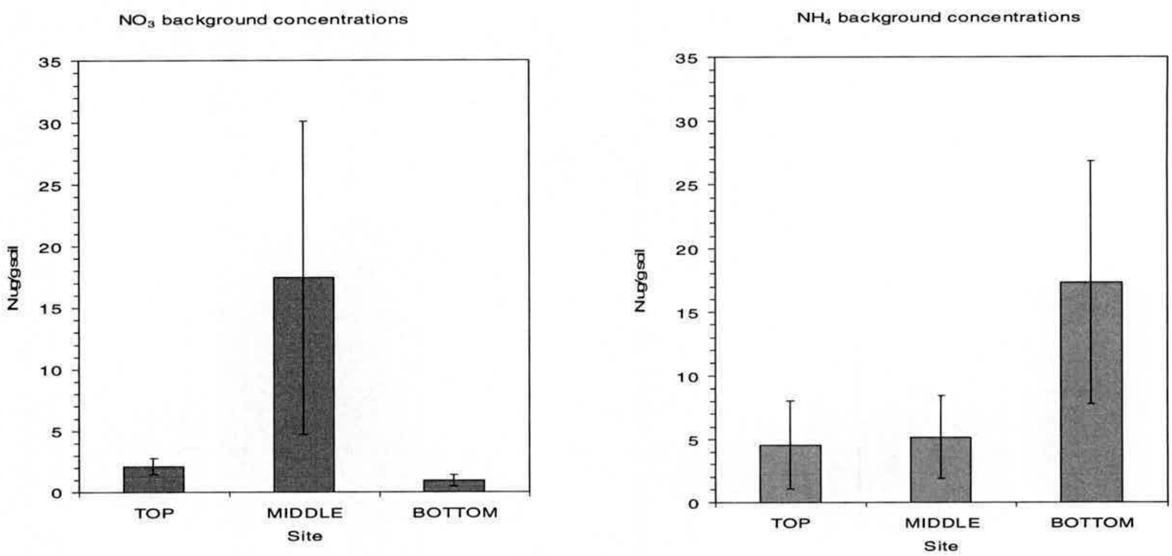
^{15}N 'pool dilution' : ^{15}N pool dilution assays were made at all of the three wetlands on 4 separate occasions between 2005-2007 to account for seasonal variation in biological functions. At each site, four plots were marked out along a transect. Five soil cores were taken at each of these plots, two for NH_4^+ analysis, two for NO_3^- analysis and a final control core (colorimetric analysis by segmented flow autoanalyser). One of the NH_4^+ and one NO_3^- core were returned to the lab for initial prelabelling analysis. The remaining ammonium core had 15.3 atom % $^{15}NH_4^+$ added, and the NO_3^- core had 15.3 atom % $^{15}NO_3^-$ added. These were then placed back into the ground from the plot that they were excavated from and covered with soil. The control cores were also placed back in the ground (T/Umbrella method of Hughes et al. 2004). The "initial" cores were injected with either ^{15}N atom % NH_4^+ or NO_3^- in the laboratory and were then KCl extracted. The resulting solutions were then steam

distilled, weighed, and then combusted using a CarloErba elemental analyser. The resultant N_2 from combustion/reduction was analysed for $^{15}N/^{14}N$ ratios using a Dennis Leigh technology IRMS (CEH Stable Isdtope Facility, Lancaster). After seven days incubation *in situ* the remaining cores were removed from the ground and the same procedure carried out. This whole process was repeated four times throughout the year to examine seasonal variations. Sampling times were April 2005, July 2005, October 2005 and January 2006. The Estimates of gross mineralization, ammonification, nitrification and immobilization fluxes were made using standard ^{15}N pool dilution mass balance calculations (e.g. Kirkham and Bartholomew 1954; Murphy et al., 1999).

3.3 Results 2005-2006

Preliminary results show highest nitrate concentrations in the rheotrophic flush and highest ammonium concentrations in the mineratrophic valley bog (Figure 3). Results from Wetland catena ^{15}N pool dilution experimentation are shown in Table 1.

Figure 3. Background wetland nitrate and ammonium concentrations from April 2005.



Initial analyses suggest that highest gross fluxes and nitrogen turnover were observed in the mineratrophic valley bottom wetland. Indeed the mineratrophic wetland nitrate pool was significantly reduced during months of highest ammonium immobilization. Immobilisation by microbes was also important in the ombrotrophic system and less so in the more anaerobic and the saturated rheotrophic ecosystem. This suggests that an interaction between the seasonality and the wetland type could be an important regulator of N dynamics.

Table 1. Calculated Gross and Net fluxes derived from mass balance ^{15}N pool dilution equations . where N>INFLOW is nitrification and ammonification and N<OUTFLOW is ammonification, nitrification and biological immobilization.

	Ombrotrophic		*Rheotrophic*		*Mineratrophic*	
N>INFLOW						
	NO_3^- *nitrification*	NH_4^+ *ammonif*	NO_3^- *nitrification*	NH_4^+ *ammonif*	NO_3^- *nitrification*	NH_4^+ *ammonif*
APRIL	2.156 ±0.20	3.485 ±1.87	0.684 ±1.70	2.661 ±0.30	4.482 ±1.32	10.127 ±4.23
JULY	5.588 ±1.35	4.086 ±1.72	3.467 ±1.83	3.536 ±1.70	0	18.248 ±7.20
OCTOBER	3.327 ±0.49	2.883 ±1.68	1.675 ±1.45	2.402 ±0.89	0	7.868 ±3.50
N<OUTFLOW						
	NO_3^- *immobilisation*	NH_4^+ *immobilization + nitrification*	NO_3^- *immobilisation*	NH_4^+ *immobilisation + nitrification*	NO_3^- *immobilisation*	NH_4^+ *immobilisation + nitrification*
APRIL	7.184 ±0.66	5.433 ±3.67	2.368 ±2.69	3.756 ±1.20	11.091 ±3.08	6.286 ±5.49
JULY	13.007 ±3.27	6.988 ±2.22	6.212 ±2.76	2.887 ±0.78	0	13.477 ±3.21
OCTOBER	9.901 ±1.64	5.163 ±3.86	4.371 ±3.86	2.345 ±0.89	0	9.738 ±4.26

4. Collaboration with Dynamic Modelling Umbrella and Freshwater Umbrella

Initial links have been made with other CEH modeling groups (Drs C. Evans and S. Smart) particularly prospect of using landscape data for integration into upland MAGIC/SMART model runs.

5. Key Findings

The ^{15}N pool dilution method was developed and tested for application in water saturated organic soils and peats. Early findings from the current T-Umbrella Task 13 'Wetlands as Nitrate Regulators' show that season and climate are strong drivers of soil N biofunction. Results also indicate that there is a strong influence of wetland type on gross fluxes. Progress is good and full data with AWS data will be completed over the following 12 months.

6. References

Hughes S., Grant H., Ostle N., Emmett B.A. and UKREATE. (2004). The controls on immobilisation of ammonium and nitrate and the link to the onset of N saturation.

Kirkham D. and Bartholomew W. V. (1954). Equations for following nutrient transformations in soil, utilizing tracer data. *Soil Science Society of America Proceedings* **18**: 33-34.

Murphy D.V., Bhogal A., Sheperd M., Goulding K.W.T., Jarvis S.C., Barraclough D. and Gaunt J.L. (1999). Comparison of ^{15}N labelling methods to measure gross nitrogen mineralization. *Soil Biology and Biochemistry* **31**: 2015-2024.

Schimel J. (1996). Assumptions and errors in the 15NH4+ pool dilution technique for measuring mineralization and immobilization. *Soil Biology and Biochemistry* **28**: 827-828.

Work Package 2:
Impacts, Recovery and Processes

Task 14:
Provision and Interpretation of Long Term Data to Refine Critical Loads for Forests, Modifications Required due to Climate Change and Review of Peat pH

M Broadmeadow and E Vanguelova

Forest Research

Task 14. – Provision and Interpretation of Long Term Data to Refine Critical Loads for Forests, Modifications Required due to Climate Change and Review of Peat pH

M Broadmeadow and E Vanguelova

Forest Research

1. Summary

The principal roles of Forest Research are to provide expert advice and to deliver a range of site specific environmental, soil and vegetation-related data-sets required for the development and evaluation of the Critical Loads methodology and the testing of dynamic ecosystem models including MAGIC and SAFE. The data-sets are based on measurements made in the UK plots of the EC and UNECE-ICP (Forests) Intensive Forest Health Monitoring (Level II) Network, which were established in 1994-5.

Key findings include the confirmation of recovery of soil solution pH from high pollution loading at the Ladybower site in the English Midlands and a downward trend in soil solution sulphate concentrations at most sites. The steady increase in nitrate in soil solution at Tummel, that was observed between 1996 and 2003, was reversed in 2004 and 2005, with annual mean concentrations in 2004 half the value observed in 2003 at both 10 and 50 cm depth. The soil solution nitrogen dynamics at this site cannot be interpreted by dynamic models of ecosystem chemistry and, instead, a biotic explanation is offered, linked to observed peaks of litterfall and data from the UK Forest Condition Survey. The reasons for observed trends in crown condition at the level II plots are also evaluated.

The simple mass balance equation for setting critical loads for acidity and nutrient nitrogen includes a growth uptake term for base cations and nitrogen, respectively. Progress is reported on a methodology for incorporating regional variation in productivity on the growth uptake term, together with an approach to represent the impacts of climate change. Progress is also reported on a review of peat pH values which is being conducted with the objective of providing a robust basis for confirming or amending the critical pH for organic soils in the UK .

2. Policy Relevance

The critical load approach is a key element of emissions reduction policy. Its continuous evaluation and development using current data are crucial for producing updated critical loads exceedance maps for the UK and thus targeting implementation policies towards effective ecosystem recovery. For woodland, the approach must be appropriate to the breadth of environmental conditions and woodland types present in the UK, and also representative of both unmanaged woodland and current practices employed across the managed forest estate. An ongoing analysis of the level II data-sets provides an assessment of ecosystem recovery as emissions control measures are implemented. Concerns over the effects of excess nitrogen deposition and ozone pollution have come to the fore alongside climate change, which is predicted to have both direct (e.g. drought, windthrow) and indirect (pest and disease outbreaks) impacts on forest ecosystems. Some evidence is already becoming apparent. The

Level II network, in this respect, provides an invaluable source of information on the condition of forests and their interaction with the wider environment.

Of particular interest at present to critical loads evaluation is the commitment made in the review of the UK Climate Change Programme (CCP06) to increase the utilisation of woodfuel from existing woodland (Defra 2006), together with the development by the Forestry Commission of a woodfuel strategy (DTI, 2006) in response to the Biomass Task Force (Anon, 2005). The realisation of these commitments will require a step-change in harvesting activity, primarily in private-sector woodland that has been under-managed over recent decades. It is also assumed that an increased utilisation of harvesting residues and other non-merchantable fractions will contribute to meeting the target, increasing nitrogen and base cation removal at harvest, effectively increasing nutrient nitrogen and decreasing acidity critical loads. CCP06 also commits Government to 'explore the potential for a market-based carbon trading the land management and forestry sector' – potentially raising the possibility of enhanced rates of woodland creation. Continuing critical loads development will enable an evaluation of how a range of land-use change scenarios will affect critical load exceedance, both from the perspective of reduced emissions and enhanced uptake.

By validating dynamic ecosystem models using observed spatial and temporal trends in response indicators (such as in Level II monitoring), uncertainty in the output of the models can be reduced. The models can then be used to predict the effectiveness of a range of emissions control policies on specific woodland ecosystems. A further benefit of the application of process-based dynamic models is their ability to account for predicted climate change and to distinguish between climate and pollution driven impacts. As well as the opportunity to account for the impacts of climate change, the application of dynamic models can predict the consequences of implementing policies aimed at climate change mitigation.

3. Project Update

1.1 Level II long term monitoring of forest ecosystems

Monitoring has now been carried out across the Level II network for more than ten years. Comprehensive data-sets have been obtained, providing the opportunity for a closer examination of long-term trends including, for example, both the input of acidifying components in precipitation, and a range of response variables on varying time scales. In addition, the long run of data provides the opportunity to interpret some of the trends and responses.

1.1.1 Case study – a biotic influence on soil solution chemistry at the Tummel Level II site

Of the three Sitka spruce Level II plots, high nitrate concentrations in soil solution are apparent at both Coalburn and Tummel, with an increasing trend evident at Tummel. At both sites, these high concentrations are at odds with those predicted by MAGIC, assuming the low total nitrogen deposition typical of both areas. One possible explanation for Coalburn is that they may reflect mineralisation of the peat, minimal drainage through the clay underlying the peat and effective uptake by sphagnum in the drains within the plot; nitrogen is thus not leaving the site, and this is confirmed by freshwater chemistry which does not show a significant loss of nitrogen. The high concentrations at Tummel are more difficult to explain, although the precipitation and throughfall chemistry, soil solution chemistry, litterfall and crown condition data-sets suggest that the steady increase in mean annual NO_3-N in soil solution at Tummel may be due to a biotic influence. The increase in mean annual soil solution NO_3-N for 2001, 2002 and 2003 corresponds to two distinct peaks in NO_3-N in throughfall each year, one in early spring and one in late summer, with the same delay prior to appearing in soil solution. However, there was no corresponding increase in bulk precipitation chemistry, excluding an additional source of pollution as a possible cause. Furthermore, data for 2004 to date indicate that soil solution NO_3-N concentrations have fallen to levels typical of the site prior to these 'episodes'. Litterfall has been monitored at the site since 2000, and it is now apparent that the peaks in throughfall and soil solution nitrate were followed by particularly heavy litterfall associated with attacks by the green spruce aphid, *Elatobium abietinum*. In winter and early summer the aphid feeds on mature needles which become discoloured and fall. The Forest Condition Survey results for the period 1993–2003 clearly show two occasions on which the crown density of Sitka spruce has deteriorated markedly (at a national level), which has been largely attributed to defoliation by *Elatobium* between 1996 and 1997, and between 2001 and 2002 (Hendry, 2005). Increased nitrate in throughfall soil solution corresponded with the earlier national outbreak, although litterfall data are not available to confirm that the site at Tummel was affected. It is also interesting to note that nitrate concentrations in soil solution correlated with [Ca] alone, which may be related to the aphid metabolism or the chemistry of their exudates.

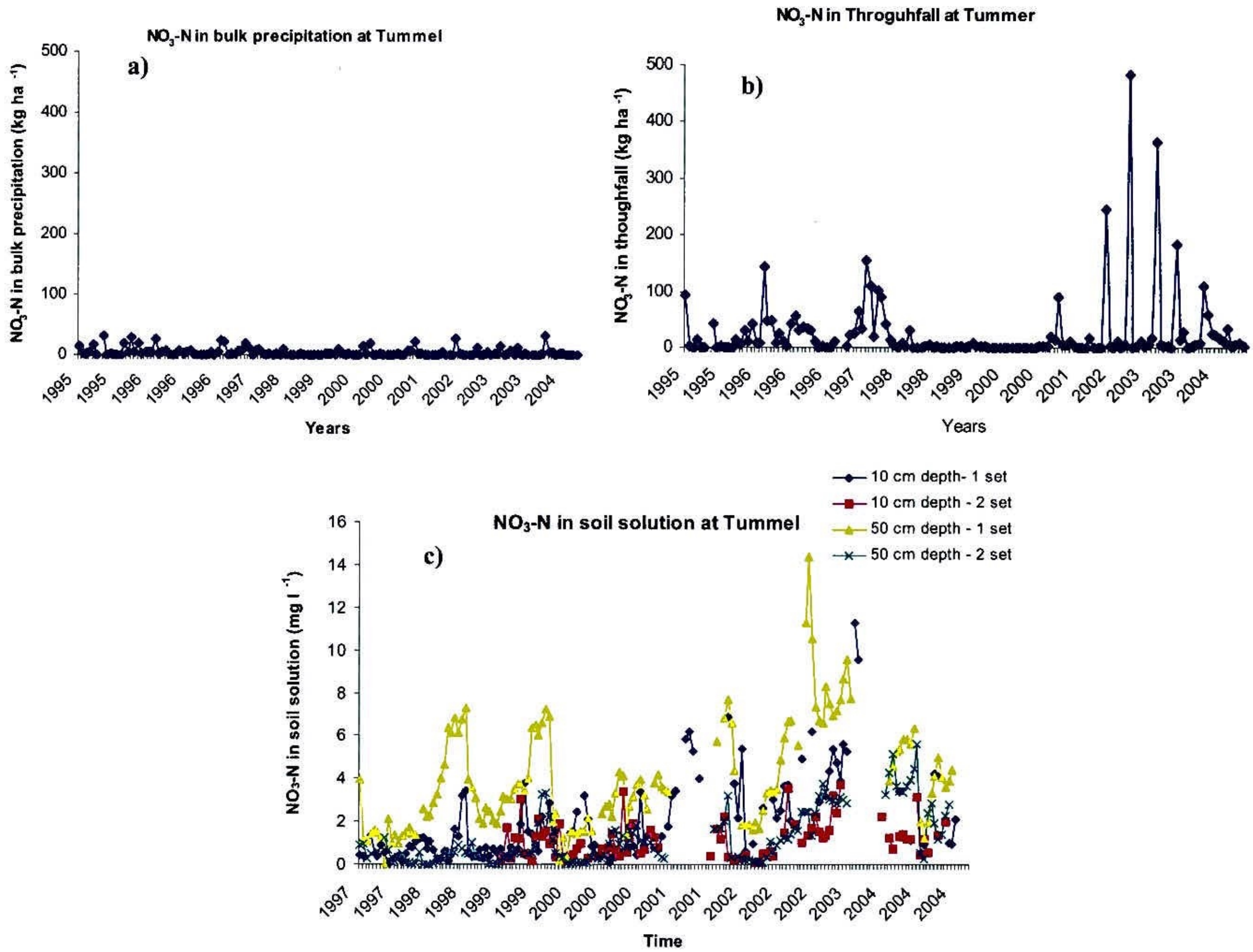


Figure 1. **NO_3-N in bulk precipitation (a), thoughfall (b) and soil solution (c) at the Tummel Level II site between 1995 and 2004.**

A range of insect pests are responsible for defoliation and other damage to forest trees. It is thought that many insect pests may become more damaging as a result of climate change, in part, driven by expectations that more frequent and severe summer droughts will make trees more susceptible to biotic agents. The integrated nature of monitoring across the level II network enables the relationships between climate, pollutant exposure, crown condition and insect populations, as estimated from litterfall (numbers of weevils, caterpillars and defoliating and gall-making insects) to be explored. To strengthen these evaluations, aphid populations will be estimated using insects bands.

1.1.2 Evaluation of trends in Level II crown condition

Crown condition is one of the principal measures used to assess tree responses to biotic and abiotic factors at Level II sites and has also been selected by the Ministerial Conference for the Protection of Forests in Europe (MCPFE) as an indicator for sustainable forest management. Crown condition is expressed as percentage reduction in crown density relative to either an 'ideal' tree. Trends in crown density of oak and Scots pine from the Level II sites (Figure 2) are in line with the general deterioration of crown condition reported for these species since 1993 that is evident in the more extensive Level I data-set, in both the UK and Europe (Hendry, 2005).

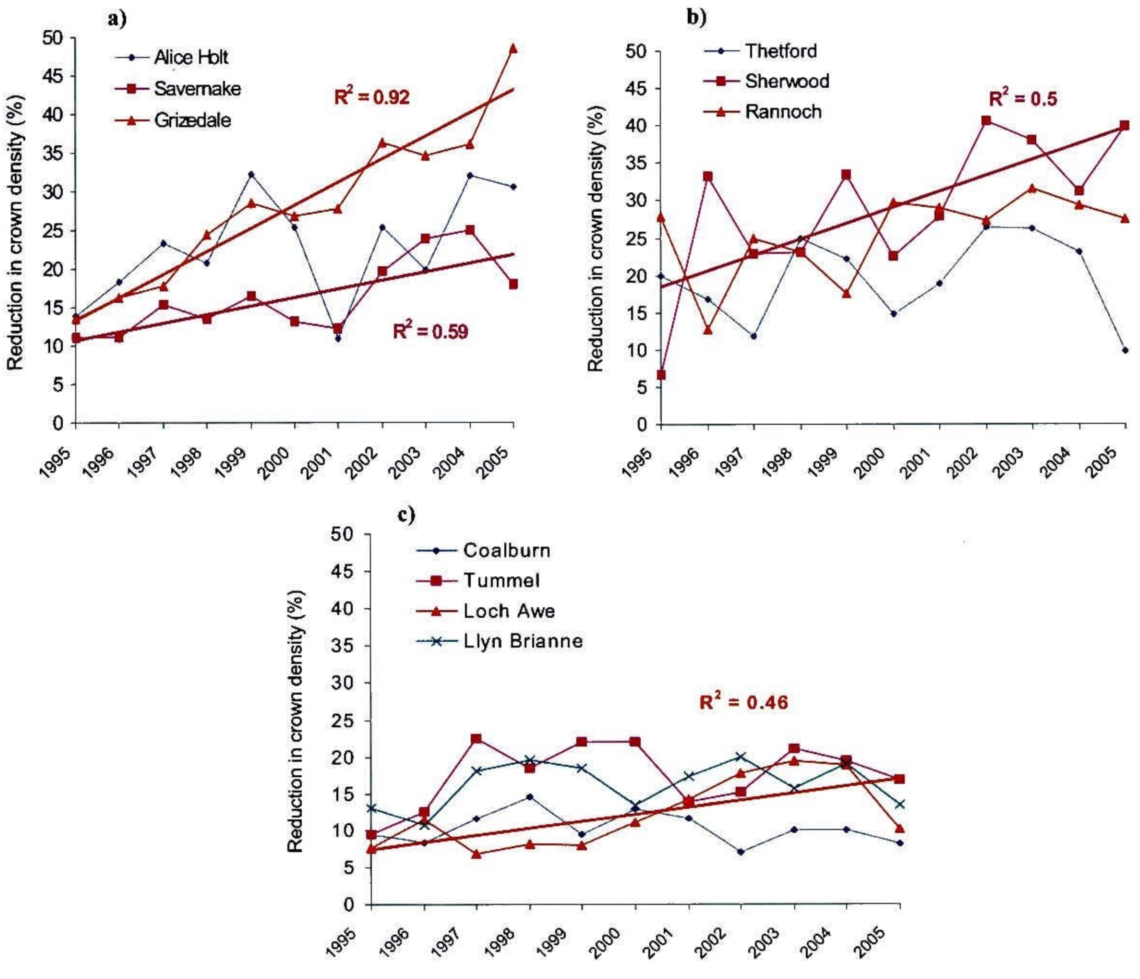


Figure 2. Crown condition of all Level II sites between 1995 and 2005.

Crown condition reflects a range of biotic and abiotic factors including insect defoliation, fungal infection, climate, air pollution and, frequently, to an interaction between these factors. A strong deterioration in mean crown density (from 14% in 1995 to 48 % in 2005) has been observed for oak at Grizedale (Figure 2a), corresponding to severe defoliation by Tortrix caterpillars, as indicated by the amount of frass and pupae numbers recorded during routine litterfall analysis. N and S deposition is relatively high compared to the other oak sites, while soil buffering is minimal due to the low soil base cation content. A similar deterioration in crown condition has been observed (from 7% in 1995 to 40% in 2005) at the Scots pine Level II site at Sherwood, situated in the previously highly polluted south Pennines (Figure 2b). Of the Sitka spruce sites, only Loch Awe showed a deterioration in crown density but the magnitude of this decline was much smaller than the oak and Scots pine sites (Figure2c). It could therefore be inferred that high N and S deposition can contribute to reduced crown density supporting the contention of Kennedy (2002), although no causal relationship can be established.

1.2 Soil peat pH review

Critical loads of acidity for woodland habitat on peat soils are based on a critical soil solution pH value of 4.4. Although a very important soil type for the UK, predominately Scotland and parts of Wales, supporting data for setting the critical pH of peat soils in the UK is limited. The current value of 4.4 is high in comparison to that proposed in the UNECE mapping manual, and also high compared to the majority of forest soils on which there is no evidence of pH related damage. A review is being undertaken to collate data for 'pristine peats' in the UK including information on the vegetation from which they were derived. Supporting data for setting the critical pH of peat soils in the UK is limited, and based upon a single experiment using Calluna peat. The current value of 4.4 is high in comparison to that proposed in the UNECE mapping manual, and also high compared to the majority of forest soils on which there is no evidence of pH related damage.

FR have identified sources and begun to collate data for 'pristine peats' in the UK including information on the vegetation from which they were derived. These data will be compared with non-UK data-sets (Ireland, North America, Fennoscandia, New Zealand, southern Chile) where similar peats exist, with the objective of providing a robust analysis of peat pH in functioning peatland ecosystems as a basis for a possible informed change to the critical peat pH adopted by the UK. Contact has been made with relevant experts in Finland (Raija Laiho and Teemu Tahvanainen) and Ireland (Ted Farrell) to source information published in the 'gray' literature. Data held in the Scottish Soils Knowledge and Information Database (SSKIB) will also be accessed. The review will be completed and submitted to the Department by the end of September.

1.3 Impact of climate change on the Critical Loads growth uptake term for woodland

Climate change is likely to affect tree growth differently in individual regions; for example, growth is likely to be reduced in southern England as a result of the increasing frequency and severity of summer droughts, contrasting with much of the north and west of the UK, where longer growing seasons and reduced cloud cover are likely to enhance tree growth (see Broadmeadow and Ray, 2005). In the final year of the contract, the impact of these changes (based on the UKCIP02 climate change scenarios: Hulme *et al.,* 2002) in forest productivity on the growth uptake term of the critical loads equation will be assessed, using output from the decision support system, Ecological site Classification ESC), to modify baseline growth rates. In addition, this process will provide the option of spatial variation in the growth uptake term, rather than the current assumption that all woodland is of average productivity class.

ESC has been updated over the past year: (1) the spatial representation of exposure (ie wind) limitation of tree growth has been improved; (2) the soils information underlying ESC-CC has been updated for England and Wales using data on available water capacity and depth to gleying of the NATMAP data-set of the Soil Survey of England and Wales. This process has not been possible for Scotland, to date. The benefits of this work are two-fold. First, predictions for species suitability in a given 5 km grid square are based on all mapped soil types making up that grid square, rather than the dominant soil type. More detailed analyses are therefore possible, including assessments of total areas where a given species is predicted to become less suitable, or the targeting of areas that will remain suitable. The second benefit is that it is now possible to couple the ESC SMR (soil moisture regime) factor to moisture deficit – ie accommodating climate change predictions in SMR, rather than solely in moisture deficit. The spatial representation of soil properties has also been improved, through incorporation of processed improving the ability to mode drought impacts; (3) the range of climate change scenarios covered has been extended to include the UKCIP02 2080s Low and High emissions scenarios; (4) the species suitability models have been extended to enable their application to the 2080s scenarios. The one outstanding area of development is the incorporation of a function to represent the CO_2 fertilisation effect. During the course of the next year, this will be implemented as the output from a process-based model (ForestGrowth) for 'generic' broadleaf and conifer species. The importance of this function is demonstrated in Figure 3, which shows a dendrochronological analysis of growth rate of oak over the past 180 years adjacent to one of the Level II plots (Alice Holt). The significant increase in diameter increment since 1980 is likely to have been driven by rising CO_2 levels in the atmosphere, corresponding to a growth stage when increment would be expected to be declining.

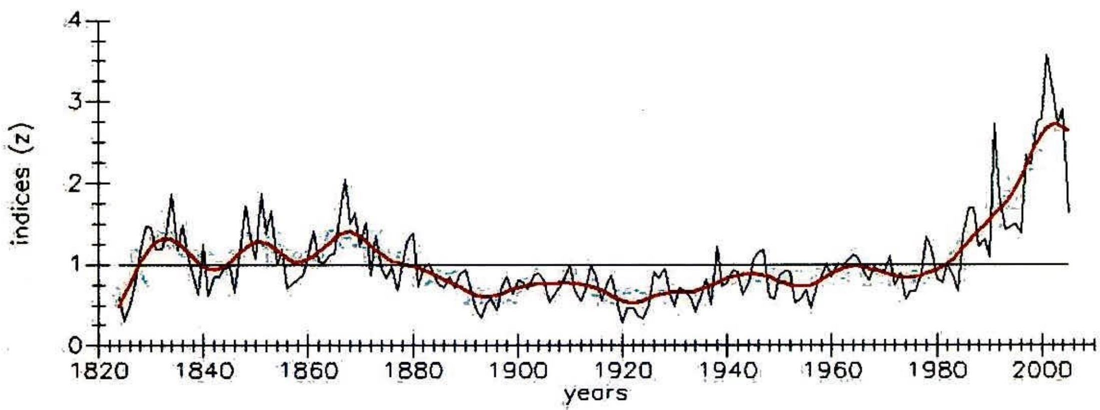


Figure 3. Tree-ring analysis of diameter increment of oak between 1820 and 2005 for 15 trees in Alice Holt forest. Raw data have been de-trended using a Hugershoff curve.

4. Collaboration with Dynamic Modelling Umbrella and Freshwater Umbrella

Collaboration over dynamic modelling has continued, using data from the Level II sites. For some sites, agreement between observed and modelled values are good. However, dynamic models do not simulate short-term variations in N cycling. Therefore the large, recent input in N at both Thetford and Savernake as a result of land use change/animal husbandry has resulted in significant discrepancies between modelled and observed concentrations of nitrogen compounds with a knock-on effect on acid neutralising capacity (ANC).

The very high nitrate concentrations in soil solution at Tummel and Coalburn have prevented any analysis. However, the interpretation of the variable nitrate concentrations given in section 1.1.1, together with the range of available input data provide the opportunity to test the hypothesis that the variation in soil solution nitrate concentrations is driven by insect pest induced defoliation and enhanced (biotic) N deposition. A possible explanation for the high soil solution nitrate concentrations at Coalburn has been proposed previously, summarised as a lack of drainage and retention of nitrate in vegetation. Although presenting more difficulties than at Tummel, it may also be possible to test this hypothesis using dynamic models, requiring a limited number of additional measurements for a robust analysis.

5. Key Findings

Evidence indicates that insect pest outbreaks in stands of Sitka spruce can significantly enhance nitrogen (and presumably other nutrient) input to forest ecosystems, producing a transient, but significant effect on soil solution chemistry. As a result of the breadth of data collected at the Level II sites, this hypothesis can be tested using dynamic ecosystem models, potentially enabling some of the discrepancies between modelled and observed data to be explained. If successful, this would have the benefit of (a) further developing the models and, (b) improving the confidence in model outputs. Other evidence from the Level II network confirms the continuing recovery at a site in the south Pennines from historically high levels of acid deposition, while there is also some evidence of recovery from a short-term increase in nitrogen deposition at the site at Thetford.

A clear signal of a CO_2-fertilisation effect has been demonstrated for old-growth oak woodland, indicating that accommodating climate change-driven changes in the growth uptake term of the critical load equation is a priority. A methodology to incorporate this, and other effects of climate change is under development.

6. References

Anon (2005). Biomass Task Force. Report to Government. (http://www.defra.gov.uk/farm/acu/energy/biomass-taskforce/).

Broadmeadow, M. and Ray, D. (2005). Climate change and British Woodland. Information Note 69. Forestry Commission, Edinburgh. (http://www.forestresearch.gov.uk/climatechange).

Defra (2006). Climate Change: the UK Programme 2006. The Stationery Office, Norwich. (http://www.defra.gov.uk/environment/ climatechange/uk/ukccp/index.htm).

DTI (2006). Government response to the Biomass Task Force. (http://www.defra.gov.uk/farm/acu/energy/biomass-taskforce/).

Hendry, S.J. (2005). Forest Condition 2004. Information Note 75. Forestry Commission, Edinburgh. (http://www.forestry.gov.uk/website/publications.nsf/).

Hulme, M., Jenkins, G., Lu, X. et al. (2002). Climate change scenarios for the United Kingdom: The UKCIP02 scientific report. Tyndall Centre, UEA, Norwich.

Kennedy, F. (2002). How extensive are the effects of nitrogen pollution in Great Britain's Forests? Forest Research Annual Report and Accounts 2001-2. (http://www.forestresearch.gov.uk/pdf/FR0102nitro.pdf/$FILE/FR0102nitro.pdf).

Work Package 2: Impacts, Recovery and Processes

Task 15: Analysis of monitoring data

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Task 15 - Analysis of monitoring data

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1 Summary

Understanding the increasing abundance of nutrient-demanding species in upland infertile habitats.

- Plant species less typical of upland vegetation types and more typical of lowland semi-improved grasslands, have increased in occupancy in Countryside Survey (CS) plots in upland Britain between 1978 and 1998. We sought to explain this pattern by correlating spatial and temporal changes in abundance of this species group with potential drivers including change in growing season length, change in sheep density between 1969 and 2000, sheep numbers in 2000, cover of intensive Broad Habitat in 1998, total N deposition in 1996 additionally broken down into NHx and NOy as separate explanatory variables.
- When variation due to other drivers was covaried out, the probability of at least one nutrient-demanding, lowland mesophyte being present in an upland grassland, heath or bog plot in Countryside Survey data in 1998 was best explained by long-term average annual temperature and total N deposition in 1996.
- However, the richness of nutrient-demanding, lowland mesophytes in CS plots in 1998 was positively correlated with sheep grazing intensity and cover of intensive Broad Habitat in each 1km square.
- A negative correlation between richness and NOy deposition remains difficult to explain.
- No significant correlations were detected between change in richness or presence of nutrient-demanding species over time (1978 to 1998 or 1990 to 1998).
- In conclusion, a warmer, more lowland climate plus high N deposition seem to track the incursion of these nutrient-demanding species into upland Britain but more intense agricultural activity, particularly sheep grazing, is best correlated with a greater richness of these species per square metre, and therefore with the most marked changes in local species composition and vegetation character.

Deriving botanical indicators of N deposition across upland, infertile vegetation types in Britain

- Previous work has shown that the large-scale eutrophication signal across Britain can be attributed to modelled N deposition at the level of individual plant species. This relies on correlating species changes with N deposition.
- Hitherto, signal attribution has not taken into account the competing effects of other human impacts including climate change and land-use.
- We sought to identify individual plant species as indicators of the unique signal of N deposition having covaried out relationships with these other drivers of ecological change.
- Despite the fact that the scale at which driving variables are resolved was much bigger than CS sampling plots, modeled NHx deposition displayed significant correlations

with 41% of the common higher plant species present in upland CS plots in the three survey years (total of 146 plant species tested).

- A smaller subset carried the strongest unique signal of NHx deposition.
- Overall more species-level differences in abundance between CS squares could be attributed to spatial differences in NHx deposition than to change in sheep density or growing season length.
- Finally, these results must be strongly qualified by the statement that they are correlative patterns that could be concealing other causal links, rather than causal relationships inferred more strongly from experimental manipulations.

2 Introduction

The most important drivers of vegetation change, in terms of their prevalence across upland Britain between 1978 and 1998, were increases in grazing pressure and nitrogen deposition (Smart et al 2005). Most recently, climate change has also increased in importance as a potential driver. During this period, clear signals of eutrophication have been detected in upland vegetation types involving a gradual increase in abundance of a small number of nutrient-demanding mesophytes more typical of semi-improved, lowland grasslands (Smart et al 2005, Smart et al 2003a,b). Given the operation of multiple drivers during the interval, a key question is to what extent observed species compositional changes uniquely reflect N deposition. This report takes two approaches to advance our understanding of the relative contribution of key human impacts in driving change in upland infertile vegetation types. In the first section, the joint abundance of a specific set of species in upland Countryside Survey (CS) plots was used to create three response variable; the proportion of plots occupied per 1km square in 1998, change in proportion of plots occupied over time, and species richness of the group per plot in 1998. The group of species used comprise those known to have increased in abundance across upland heath, bog and grassland between 1978 and 1998. These are listed in table 1.

Table 1: List of atypical species found to have increased in infertile upland habitats (heath, bog and upland grasslands) between 1978 and 1998 (see Smart et al 2005 and www.cs2000.org.uk/M01_tables/reports).

Agrostis stolonifera	*Urtica dioica*
Cardamine hirsuta/flexuosa	*Poa trivilais*
Cerastium fontanum	*Poa annua*
Cirsium arvense	*Arrhenatherum elatius*
Cirsium vulgare	*Plantago major*
Cynosurus cristatus	*Anthoxanthum odoratum*
Deschampsia cespitosa	*Ranunculus repens*
Festuca rubra agg.	*Poa trivilais/nemoralis*
Holcus lanatus	
Holcus mollis	
Lolium perenne	
Ranunculus ficaria	
Rumex acetosa	
Stellaria uliginosa	

The second section also focuses on upland habitats in GB but examines the signals of all species frequent enough to have shown significant change over the 1978 to 1998 period. The objective here was to search for botanical indicators that best reflected the unique impact of N deposition (both positive and negative) in the presence of other potential drivers.

2.1 Understanding the increasing abundance of nutrient-demanding species in upland infertile habitats.

Methods

The 1998 Countryside Survey data set was used to determine the spatial variation of atypical species in relation to a number of explanatory variables. The 1998 survey data set contains the largest number of plots (16 851) of different sizes and types. The number of plots containing at least one atypical species in a square was counted and the proportion of such plots in a CS 1 km square calculated. Plots were only selected from upland environmental zones 3,5 and 6 and in the upland vegetation types heath/bog and moorland grass/mosaics (Bunce et al 1999; Smart et al 2005). In addition, only areal and linear plot types most likely to be affected by grazing animals in the unenclosed uplands were included. Hence field boundary and roadverge plots were excluded. The mean richness of atypical species per plot per square was also calculated and used as an additional response variable.

Generalized linear models (proc genmod, SAS Insitute) were used to determine which of a range of potential driving variables were best correlated with the proportion of plots in each CS 1km square containing at least one member of the species group (Table 1). Two types of analysis were carried out: First, tests of the explanatory power of each driver were carried separately out for each driver. However because drivers can operate in a correlated fashion across GB, each driver was additionally tested by entering it last into a sequential model after all other variables (Type 1 tests). These analyses were then repeated using mean richness of nutrient-demanding species per plot per square. Explanatory variables are listed in Table 2. Change in the proportion of nutrient-demanding species over time in relation to environmental variables was also analysed using the above methods between 1990 and 1998, and between 1978 and 1998.

Table 2. Explanatory variables used to detect partial and unconditional correlations between botanical change in upland GB and gradients of the intensity of key human impacts.

Driver	Spatial scale	Temporal scale	Source
Sheep density	2km sqr	2000	MAFF census[1]
Change in sheep density	2km sqr	1969-2000	MAFF census[1]
Mean min Jan temp	5km sqr	1961-1999	UKCIP[2]
Mean max July temp	5km sqr	1961-1999	UKCIP[2]
Change in annual growing season length	5km sqr	1961-1999	UKCIP[2]
Atmospheric N	5km sqr	1996	Models and

[1] Downloaded for each 5km sqr containing each CS 1km sqr from the EDINA AgCensus database at www.edina.ac.uk.

[2] Long term annual average for the 5km sqr containing each CS 1km sqr. Downloaded from www.met-office.gov.uk/research/hadleycentre/obsdata/ukcip/index.html.

deposition			measurements[3]
Intensive Broad Habitat cover	1km sqr	1998	Countryside Survey field maps[4]

Results

Spatial analysis

When tested individually, therefore with no other covariates present, all explanatory variables were significantly correlated with proportion of plots occupied by nutrient-demanding species. However, this could be because drivers were correlated in their operation across the sampling domain and so shared explanatory power because their signals overlapped. We know, for example, that both increased sheep grazing and heightened N deposition can favour nutrient demanding grasses but in CS surveillance data these 'treatment' effects could well be spatially confounded (termed multi-collinearity). Hence, an apparent signal due to N deposition could be due to the fact that the N deposition gradient is strongly correlated with the sheep grazing intensity gradient and vice versa. The type 1 tests quantify the partial explanatory power of each driver and therefore exclude any overlapping variance beween drivers. The discrepancy between the two sets of results gives an indication of the extent of this spatial overlap among drivers. When the type 1 test results are examined (Table 3) the number of significant partial predictors drops dramatically. NOy rather than NHx now emerges as the only unique N deposition predictor part from Total Nitrogen. This probably reflects the strong correlation between change in sheep grazing intensity and NHx deposition but low correlation between change in sheep grazing and NOy deposition.

Table 3. Results from analysis of plots from aggregate classes 7 and 8, environmental zone 3,5,6 (uplands), area plots (X,Y,U, SW) with proportion of plots in a square containing a atypical species as the response variable and climatic, N deposition and grazing variables as explanatory. Both tests incorporating all variables (type 1 tests) and individual tests are included.

	Type 1 tests		Direction of correlation	Individual tests		Direction of correlation
	Chi sqr	p		chisqr	p	
Mean min jan	2.83	0.09		0.73	0.4	
Total Nitrogen	4.08	< 0.05	+	51.82	<0.001	+
NHx	0.15	0.69		46.42	<0.001	+
NOy	5.5	< 0.05	+	39.87	<0.001	+
Change in sheep numbers	0.6	0.43		17.73	<0.001	+
Total sheep 2000	0.14	0.71		12	<0.001	+
Growing season change	1.35	0.25		14.09	<0.001	-
Average temperature	4.8	< 0.05	+	42.86	<0.001	+
Amount of intensive	0.71	0.39		5.13	<0.05	+

[3] See NEGTAP (2001) available at www.nbu.ac.uk/negtap/finalreport.htm.
[4] Sum of the percentage of improved grassland+arable+urban/built in each 1km sqr.

Broad Habitat in square						

When mean species richness per plot per square in 1998 was used as a response variable instead of the proportion of plots occupied, the results were rather different (Table 4). Fewer explanatory variables showed significant relationships. Interestingly, agricultural disturbance in the form of either cover of intensive Broad Habitat per square or sheep density were associated with a higher richness per plot as expected given the habitat affinities of the species group. NOy deposition had a negative relationship; an enigmatic result that requires further investigation.

Table 4. Results from analysis of plots from aggregate classes 7 and 8, environmental zone 3,5,6 (uplands), area and stream plots (X,Y,U, SW) with <u>mean richness</u> of atypical species per square as the response variable and climatic, N deposition and grazing variables as explanatory. Both tests incorporating all variables (type 1 tests) and individual tests are included.

	Type 1 tests		Direction of correlation	Individual tests		Direction of correlation
	Chi sqr	p		chisqr	p	
Mean min jan	3.32	0.07		0.01	0.94	
Total Nitrogen	0.98	0.32		0.12	0.72	
NHx	3.57	0.06		1.34	0.24	
NOy	1.55	0.21		4.33	<0.05	-
Change in sheep numbers	0.02	0.89		5.75	<0.05	+
Total sheep 2000	0.07	0.79		4.89	<0.05	+
Growing season change	0.16	0.69		0.01	0.93	
Average temperature	8.41	<0.01	-	0.45	0.48	
Amount of intensive Broad Habitat in square	4.54	<0.05	+	15.69	<0.001	+

Temporal analysis

Results

There were no significant relationships between the proportion of plots containing an atypical species versus nitrogen deposition, climatic and grazing variables for either the time period 1978 to 1998 or 1990 and 1998 in upland habitats.

Table 5. Results from analysis of change in proportion of atypical species between 1990 and 1998 as the response variable in upland habitat types (aggregate classes 7 and 8, environmental zones 3,5,6 and X,Y,U, SW plots) and climatic, N deposition and grazing variables as explanatory. Both tests incorporating all variables (type 1 tests) and individual tests are included.

	Type 1 tests		Individual tests	
	chisqr	p	chisqr	p
Mean min jan	0.94	0.33	3.12	0.08
Total Nitrogen	0.93	0.33	2.07	0.15
NHx	0.68	0.41	2.47	0.12
NOy	0.04	0.84	0.19	0.66
Change in sheep numbers	0.38	0.53	2.19	0.14
Total sheep 2000	0.55	0.56	2.35	0.12
Growing season change	0.17	0.68	0.33	0.56
Average temperature	0.11	0.74	3.31	0.07

Table 6. Results from analysis of change in proportion of atypical species between 1978 and 1998 as the response variable in upland habitat types (aggregate classes 7 and 8, environmental zones 3,5,6 and X,Y,U, SW plots) and climatic, N deposition and grazing variables as explanatory. Both tests incorporating all variables (type 1 tests) and individual tests are included.

	Type 1 tests		Individual tests	
	chisqr	p	chisqr	p
Mean min jan	1.59	0.2	0.89	0.34
Total Nitrogen	0.08	0.78	0.4	0.5
NHx	1.89	0.17	0.19	0.7
NOy	2.43	0.12	1.19	0.27
Change in sheep numbers	1.6	0.2	1.99	0.16
Total sheep 2000	1.81	0.18	2.09	0.15
Growing season change	0.46	0.5	0.04	0.84
Average temperature	0.05	0.82	1.08	0.3

Conclusions

Strong correlations were detected between spatial differences in the abundance of nutrient-demanding species in upland plots in 1998 and all potential driving variables. These undoubtedly reflected similarly strong patterns of spatial correlation between driving variables in the way they have impacted British upland ecosystems in the last quarter of the 20^{th} century. This is because one of the defining features of ecological surveillance data is that responses will reflect impacts whose variation in type, intensity and prevalence are beyond the control of the observer. Hence, spatial confounding is always a strong possibility either by bad luck or because drivers are mechanistically correlated such as NHx deposition and numbers of grazing sheep. In such cases, unique signals maybe analytically not attributable or point the way for further attribution studies such as those reported in section 2 of this report.

Despite these inherent design problems. Two important results have emerged from these analyses. Firstly, when other covariates were taken into account, the strongest correlate of presence of an atypical species in upland vegetation was average annual temperature followed by total nitrogen deposition (Table 3). However when richness of atypical species was analysed, rather than just presence of any one member of the group, there was a clear positive link with agricultural disturbance, of which sheep grazing activity has clearly been the most important recent driver in the British uplands.

These results are interesting in that different impacts are associated with different aspects of the incursion of the species group into upland vegetation types. The presence of any species in the group appears best explained by climate and N deposition. Hence, warmer, lower altitude survey squares and those with higher N deposition are more likely to be invaded. However, the richness of the group per plot is better explained by agricultural activity. This is understandable since, grazing and improvement really typify the kinds of conditions favoured by this more characteristically lowland, grassland species group.

2.2 Deriving botanical indicators of N deposition across upland, infertile vegetation types in Britain

Methods

146 higher plant species were frequent enough for testing in upland vegetation types in repeat plots recorded in the Countryside Surveys of 1978, 1990 and 1998. Analyses were constrained to plots located in heath, bog and moorland grass/mosaics in 1978 (Bunce et al. 1999). Relationships between botanical change and potential drivers were analysed using generalized linear models (proc genmod, SAS Insitute) and generalized linear mixed models (proc glimmix, SAS Insitute). The plot data were structured as sets of species presences within fixed vegetation plots nested in 1km survey squares and recorded on three occasions. For each 1km square, one value of each explanatory variable was available. Thus the intensity of the driver did not change at each successive survey. For drivers such as change in growing season length (1961-1999) and change in sheep grazing intensity (1969-2000), the explanatory variable was a rate of change, however this still applied to the 1km square and did not change with survey year. The CS square id was treated as a random, class variable. All other covariates were continuous including year of survey.

Botanical indicators of the unique effect of N deposition were sought by quantifying the strength of the correlations between spatial and temporal changes in species frequency in CS plots versus a large-scale gradient of N deposition, before and after fitting relationships with other covariates. Hypothesis tests were as follows:

Hypothesis tests
H1: Which species changed in plot occupancy over time?
Test the main effect of survey year for each species.

H2: Which drivers were significantly positively or negatively correlated with species abundance in squares irrespective of survey year.
Test the main effects of each driver against average abundance across all years of survey.

H3: Did changes in species abundance over time depend on differences in the intensity of different drivers between CS squares.
Test the interaction terms between year of survey and each driver after fitting all main effects.

H4: Which species showed the strongest correlations with N deposition when the effects of other drivers were factored out?
Test the main effect of NHx deposition against average abundance across all years of survey having first fitted all other main effects as covariates.

Covariate selection and design problems: Here we examine design and collinearity issues as they impact signal attribution in 1978 data for upland vegetation types only. These issues refer to the lack of control over the ways drivers are arranged across the sampling domain. High levels of correlation can mean that some signals cannot be attributed uniquely to drivers or that unique signals are small and omit fractions of observed variation that are shared between drivers because they have operated in spatially correlated ways. As highlighted in section 1, this can reflect drivers that are mechanistically correlated such as NHx deposition and numbers of grazing sheep, or independent yet accidentally confounded in space. Taking account of such patterns before anlaysis can highlight drivers that could be excluded or, if included, then expected to present foreseeable and hence understandable problems in isolating unique signals.

A number of predictably strong spatial correlations existed between drivers (Table 8, Figure 1). Sheep density in 2000, change in sheep density from 1969 to 2000 and cover of intensive agriculture were all strongly intercorrelated. Therefore, only change in sheep density was retained for further analysis, because this represents the best explanatory variable for pervasive agricultural land-use impacts in upland Britain (Fuller & Gough 1999). Change in growing season length was correlated with mean minimum January temperature and mean maximum July temperature reflecting the fact that the uplands have seen the largest recent increases in growing season length. Change in growing season length change was retained as the sole climate variable because it was the most direct correlate of climate change as an ecological driver. NHx deposition was retained as the sole explanatory variable for atmospheric nitrogen deposition because of its emergence as a more important correlate of botanical change in British infertile plant communities than modelled NOy deposition (Smart et al 2004).

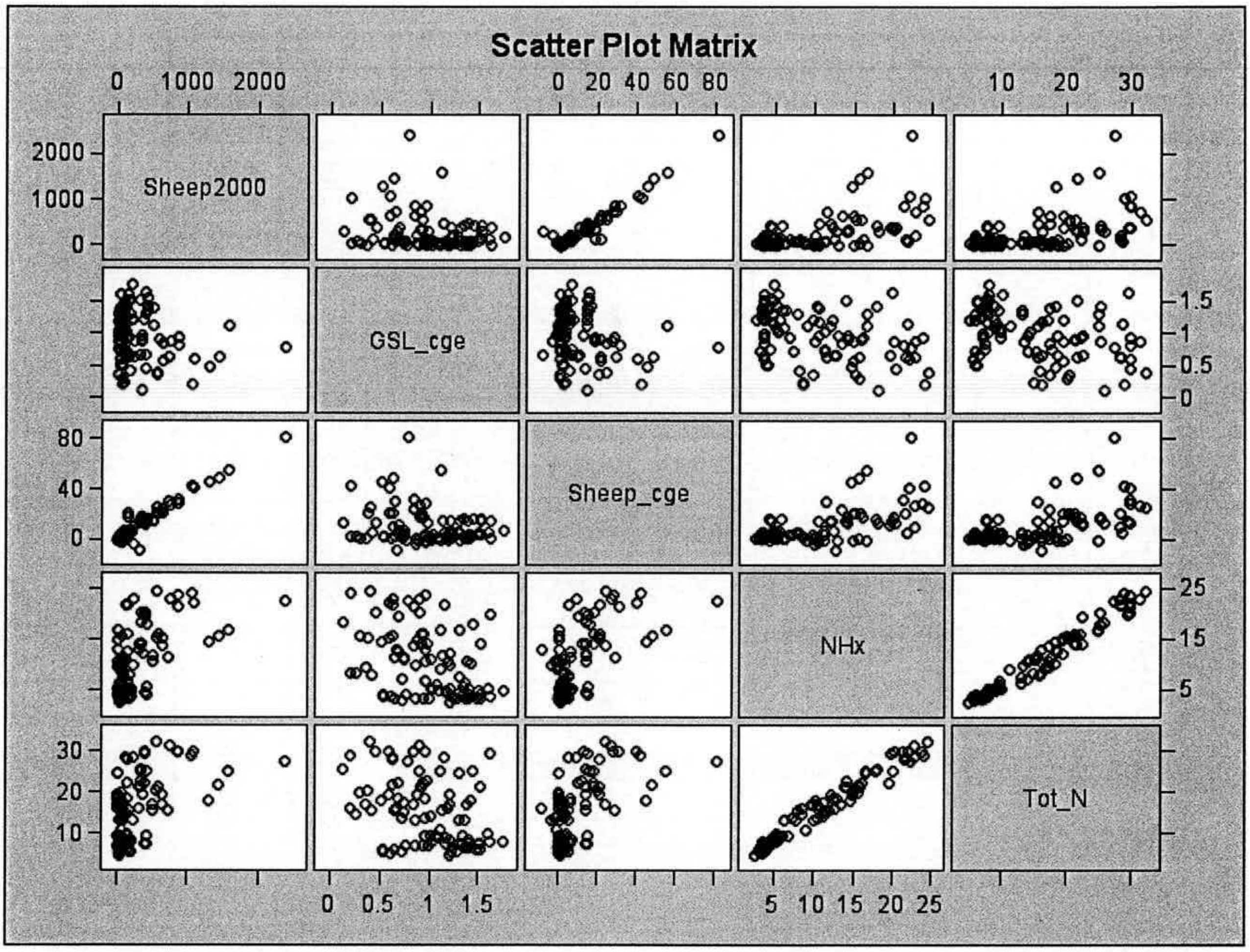
Table 7. Descriptive statistics for drivers of upland species change. Based on repeat squares recorded in 1978, '90 and '98 in upland environmental zones 3,5 and 6. See table 2 for description of drivers.

Variable	N	Mean	Std Dev	Sum	Minimum	Maximum
S2000	89	290.3	401.75	25836	0	2390
GSL	89	1.0	0.39	87	0.1	1.8
S_cge	89	10.6	14.61	941	-8.7	82.0
NHx	89	10.6	6.65	945	2.5	24.6
Tot_N	89	15.6	8.09	1390	4.9	32.5
BHint	89	15.8	27.35	1407	0	89.0
mja	89	-4.1	1.96	-362	-7.5	0.4
mju	89	19.2	2.03	1707	15.2	24.3

Table 8. Correlation matrix between drivers of upland species change.

	S2000	GSL	S_cge	NHx	Tot_N	BHint	mja	mju
S2000	1.000	-0.221 0.0372	0.966 <.0001	0.574 <.0001	0.536 <.0001	0.437 <.0001	-0.200 0.0601	0.340 0.0011
GSL		1.000	-0.231 0.0294	-0.410 <.0001	-0.379 0.0003	-0.436 <.0001	0.328 0.0017	-0.594 <.0001
S_cge			1.000	0.614 <.0001	0.587 <.0001	0.415 <.0001	-0.279 0.0081	0.418 <.0001
NHx				1.000	0.981 <.0001	0.573 <.0001	-0.489 <.0001	0.737 <.0001
Tot_N					1.000	0.466 <.0001	-0.548 <.0001	0.690 <.0001
BHint						1.00000	-0.074 0.4927	0.576 <.0001
mja							1.000	-0.484 <.0001
mju								1.000

Figure 1. Matrix plot showing patterns of correlation between potential drivers of upland botanical change. See table 2 for descriptions of driving variables.



Results

Out of 146 species tested in upland vegetation, 61 showed significant correlations between abundance in plots averaged over all three survey years, and modeled NHx deposition in 1996. 23 of these were more likely to be present at higher deposition (positive indicators) and 38 were negative indicators (Appendix 1). 19 of the negative indicators and 6 of the positive indicators can be considered to carry the strongest, unique signal of NHx deposition impacts since they showed significant relationships with NHx even after covarying out the unique or shared signal attributable to grazing and climate change (Appendix 1). Thus the best positive indicators include *Rumex acetosa, Deschampsia flexuosa* and *Juncus effusus*. The best negative indicators include *Calluna vulgaris, Erica tetralix, Carex echinata, Pinguicula vulgaris* and *Erica cinerea* (Appendix 1) The plausibility of the indicators is supported to some extent by the fact that positive indicators generally have higher Ellenberg N values and the negative group lower Ellenberg N values (Figure 2). The identification of these indicators was based on many multiple tests and an important caveat is that some significant results may have occurred by chance purely because many tests were carried out.

Links between NHx deposition and changes in abundance between surveys were explored by first testing for overall change and then determining whether any of the observed change between 1978 to '90 to '98 was explainable by NHx deposition ie. did species increase more or decrease more where NHx deposition was higher? Overall significant changes up or down are indicated in column 6 in Appendix 1. If change was additionally explained by NHx deposition then this is indicated by the presence of an estimate of the size of the effect in the Type I Yr*NHx estimate column. The results indicate that very few significant temporal changes in indicator abundance were explained by NHx deposition. This is not surprising since the power of such tests is low. It also implies that much of the botanical change correlated with spatial differences in NHx deposition had happened before 1978 as well as during the 20 year survey interval.

By analyzing relationships between individual species abundance and each potential driver having covaried out the shared effects of other drivers, we can tell which driver appeared to generate the strongest unique signals in upland GB based on the number of species that had significant partial correlations with each driver. Our tentative conclusion is that NHx deposition generated more species specific signals than either change in growing season length or change in sheep grazing intensity (Figure 3). This could simply reflect the fact that the gradient of NHx deposition was longer and better replicated than the others across Britain. However, descriptive statistics and correlations between drivers (Tables 7, 8 and Figure 1) suggest that the other gradients were not appreciably shorter or less variable.

Conclusions

Despite the fact that the scale at which driving variables are resolved was much bigger than CS sampling plots, modeled NHx deposition displayed significant correlations with 41% of the common higher plant species present in upland CS plots in the three survey years. A smaller subset (17%) carried the strongest unique signal of NHx deposition. Overall, more species-level differences in abundance between CS squares could be attributed to spatial differences in NHx deposition than to change in sheep density or growing season length.

Figure 2. Count of plant species whose abundance in CS plots in 1978,'90 and '98 showed either a negative or positive correlation with modeled NHx deposition in 1996. Indicator species have been grouped by their Ellenberg N value.

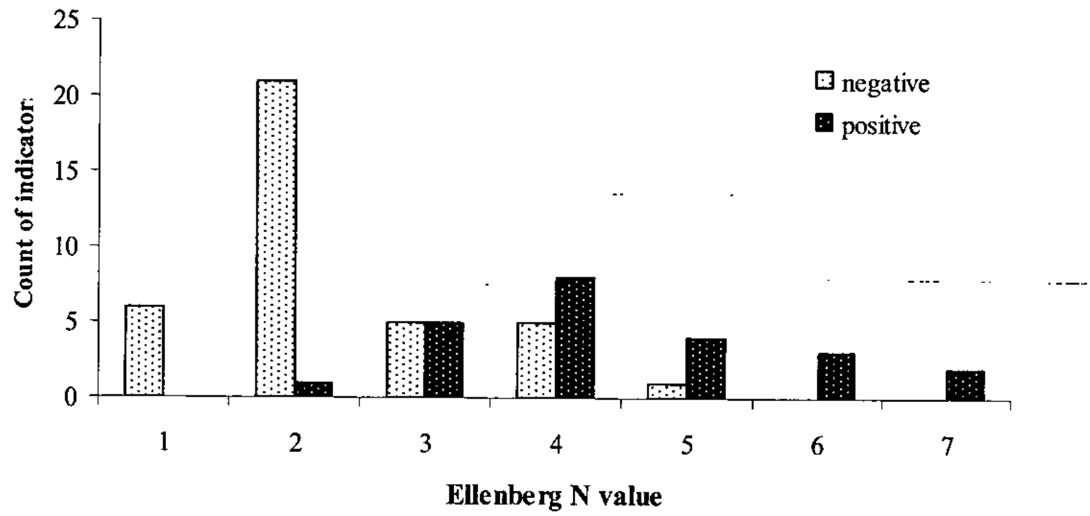
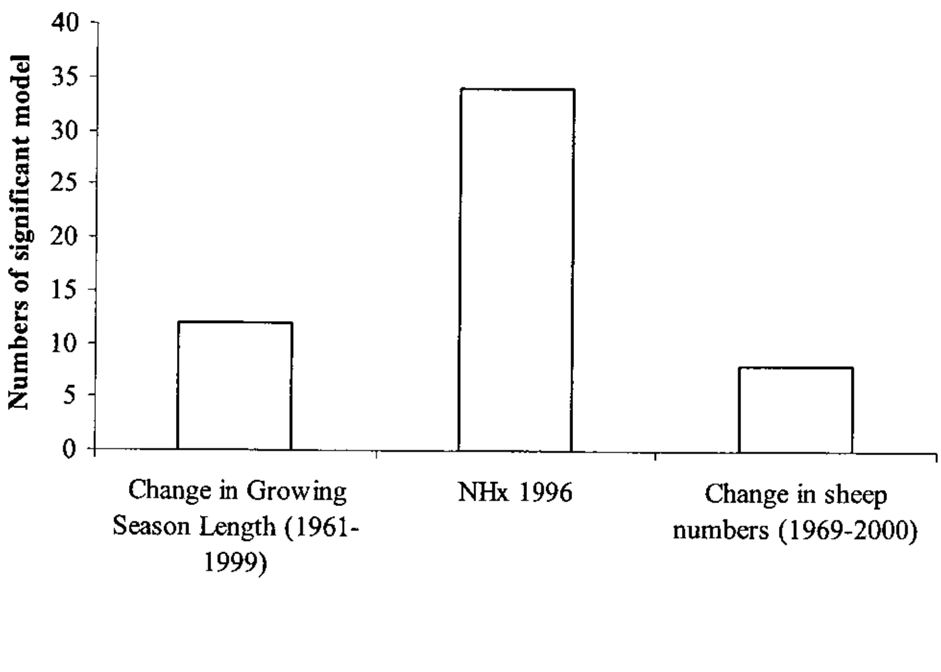


Figure 3. Numbers of species out of 146 tested, whose average abundance in CS plots in 1978, '90 and '98 showed significant partial correlations with three key drivers of upland botanical change.



3 References

Bunce, R.G.H., Barr, C.J., Gillespie, M.K., Howard, D.C., Scott, R.A., Smart, S.M., van de Poll, H.M. and Watkins, J.W. (1999) *Vegetation of the British countryside – the Countryside Vegetation System.* Department of the Environment, Transport and the Regions, London.

Fuller, R.J., Gough, S.J., 1999. Changes in sheep numbers in Britain: implications for bird populations. *Biological Conservation* 91, 73-89.

NEGTAP 2001. *National Expert Group on Transboundary Air Pollution: Acidification, Eutrophication and Ground Level Ozone in the UK. 1st report.* On behalf of the UK Department of the Environment, Transport and the regions and the devolved administrations. On-line at: www.nbu.ac.uk/negtap/finalreport.htm

Smart, S.M., Robinson, J.C., Shield, E.J., van de Poll, H.M. 2003a. Locating eutrophication effects across British vegetation between 1990 and 1998. *Global Change Biology* 9, 1763-1774.

Smart, S.M., Bunce, R.G.H., Marrs, R.H., LeDuc, M., Firbank, L.G., Maskell, L.C., Scott, W.A., Thompson, K., Walker, K.J. 2005. Large-scale changes in the abundance of common higher plant species across Britain between 1978, 1990 and 1998 as a consequence of human activity: tests of hypothesised changes in trait representation. *Biological Conservation* 124, 355-371.

Smart, S.M., Ashmore, M.R., Hornung, M., Scott, W.A., Fowler, D.A., Dragosits, U., Howard, D.C., Sutton, M.A., Famulari, D. 2004. Detecting the signal of atmospheric N deposition in recent national-scale vegetation change across Britain. *Water, Air, Soil Poll. Focus* 4: 269-278.

Smith, R.I., Fowler, D., Sutton, M.A., Flechard, C., Coyle, M. 2000. Regional estimation of pollutant gas dry deposition in the UK: model description, sensitivity analyses and outputs. *Atmos. Env.* 34, 3757-3777.

Appendix 1. Indicator species of NHx deposition in GB upland heath, bog and grasslands. Grey shading indicates significant change in plot occupancy between 1978 and 1998. The Type 1 and Type III estimates indicate the direction and relative strength of the correlation between spatial or temporal changes in species abundance and NHx deposition. Species with Type III estimates are those that showed a significant correlation with NHx deposition even after fitting other covarying potential drivers of botanical change. Hence these species (in bold) carry the strongest <u>unique</u> signal of NHx deposition impacts across the British uplands as represented in CS data.

Type III estimate	BRC names	Type I estimate	Pos/neg indicators	Yr estimate	Direction of change '78 to '98	Eb N	Type I Yr*NHx estimate
-0.714	***Drosera intermedia***	**-0.925**	**neg**	**0.276**		**1**	
	Anemone nemorosa	-0.680	neg	0.285		4	
	Carex dioica	-0.548	neg	0.000		2	
	Schoenus nigricans	-0.459	neg	0.415		2	
	Plantago coronopus	-0.417	neg	0.000		4	
	Pilosella officinarum	-0.347	neg	-1.300	down	2	0.203
-0.309	***Pedicularis sylvatica***	**-0.339**	**neg**	**0.022**		**2**	**0.099**
	Plantago maritima	-0.316	neg	0.050		4	
-0.492	***Myrica gale***	**-0.315**	**neg**	**-0.071**		**2**	
-0.436	***Pinguicula vulgaris***	**-0.287**	**neg**	**-0.036**		**2**	
	Primula vulgaris	-0.281	neg	-0.068		4	
	Salix repens agg.	-0.249	neg	0.368		3	
	Selaginella selaginoides	-0.228	neg	-0.157		2	
-0.277	***Drosera rotundifolia***	**-0.213**	**neg**	**-0.007**		**1**	
-0.169	***Carex pulicaris/serotina***	**-0.211**	**neg**	**-0.058**		**2**	**0.083**
	Hypericum pulchrum	-0.194	neg	-0.325	down	3	
-0.286	***Erica cinerea***	**-0.175**	**neg**	**-0.027**		**2**	
-0.207	***Polygala vulgaris/serpyllifolia***	**-0.173**	**neg**	**-0.063**		**2**	**0.046**
	Oreopteris limbosperma	-0.168	neg	-0.189		3	0.106
	Carex viridula subsp.brachyrrhyncha	-0.164	neg	0.186	up	2	
-0.347	***Trichophorum cespitosum***	**-0.164**	**neg**	**-0.007**		**1**	
	Blechnum spicant	-0.154	neg	-0.061		3	
-0.311	***Narthecium ossifragum***	**-0.151**	**neg**	**-0.004**		**1**	
	Carex bigelowii	-0.127	neg	-1.154	down	2	
-0.279	***Erica tetralix***	**-0.126**	**neg**	**-0.093**	down	**1**	
-0.195	***Succisa pratensis***	**-0.125**	**neg**	**0.116**		**2**	
	Armeria maritima	-0.123	neg	0.767	up	5	
	Festuca vivipara	-0.121	neg	0.034		2	
-0.215	***Carex panicea***	**-0.104**	**neg**	**0.244**	up	**2**	
	Alchemilla alpina	-0.097	neg	-0.384		3	
-0.190	***Eriophorum angustifolium***	**-0.092**	**neg**	**-0.007**		**1**	
	Thymus praecox	-0.088	neg	0.000		2	
-0.137	***Viola riviniana/reichenbiana***	**-0.087**	**neg**	**-0.192**	down	**4**	
-0.094	***Carex echinata***	**-0.085**	**neg**	**-0.119**		**2**	
	Carex binervis	-0.082	neg	0.187	up	2	
-0.216	***Molinia caerulea***	**-0.065**	**neg**	**-0.035**		**2**	
-0.218	***Calluna vulgaris***	**-0.055**	**neg**	**-0.022**		**2**	
-0.244	***Potentilla erecta***	**-0.044**	**neg**	**0.025**		**2**	

	Galium saxatile	0.034	pos	-0.021		3	
	Agrostis capillaris	0.039	pos	-0.029		4	
	Festuca ovina agg.	0.046	pos	-0.166	down	2	
0.110	***Deschampsia flexuosa***	**0.053**	**pos**	**-0.066**		**3**	
	Ranunculus repens	0.056	pos	0.148		7	
	Cirsium palustre	0.067	pos ·	0.152		4	
0.108	***Rumex acetosa***	**0.068**	**pos**	**0.247**	up	**4**	
	Stellaria media	0.069	pos	0.000		7	
	Juncus articulatus/acutiflora	0.073	pos	-0.012		3	-0.025
0.150	***Juncus effusus***	**0.073**	**pos**	**0.036**		**4**	
	Chrysosplenium oppositifolium	0.074	pos	-0.151		5	0.084
	Cardamine pratensis	0.078	pos	0.177		4	
	Cirsium vulgare	0.087	pos	0.372	up	6	
	Poa trivialis	0.096	pos	0.233		6	
	Deschampsia cespitosa	0.096	pos	0.216	up	4	
	Galium palustre	0.098	pos	-0.137		4	
0.138	***Stellaria uliginosa***	**0.103**	**pos**	**0.313**		**5**	
	Veronica chamaedrys	0.107	pos	0.216		5	-0.055
	Cynosurus cristatus	0.115	pos	0.316	up	4	
	Montia fontana	0.126	pos	0.421		3	
0.148	***Holcus mollis***	**0.149**	**pos**	**0.139**		**3**	**-0.054**
0.121	***Cardamine hirsuta/flexuosa***	**0.161**	**pos**	**0.606**	up	**5**	
	Ranunculus ficaria	0.207	pos	1.386	up	6	-0.201

Work Package 3:
Dynamic modelling and vegetation response

Task 17:
Vegetation module for dynamic modeling

Simon Smart[1], Simon Wright[1], Ibby Moy[2], James Bullock[2], Ed Rowe[3], Chris Evans[3], David Roy[4], Bridget Emmett[3]

[1] CEH Lancaster
[2] CEH Dorset
3 CEH Bangor
[4] CEH Monks wood

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1. Summary

Work completed as part of the last Terrestrial Umbrella showed much potential for linking soil (MAGIC) and vegetation models (GB_MOVE) to produce a modelling capability for testing scenarios of pollutant deposition change against expected change in abundance of individual plant species. Model testing revealed shortcomings in the current plants species response surfaces and these have continued to be developed together with an approach to add realism to species-level predictions using an informatics based approach to select species-pools for each square. The approach combines two existing databases;

1. Biological Records Centre species lists for each 10km square in Britain.
2. The satellite land cover map for 2000 (LCM2000) which provides Broad Habitat coverage at $25m^2$ resolution for the UK.

In order to downscale from the 10km square list to the 1km square, the 10km square species list will be weighted according to the coverage of each Broad Habitat in each 1km square combined with a preference index that gives, for each, species an indication of the Broad Habitat in which each species is most likely to be found. Firstly, species will be selected that have a high preference index for the target habitat. Secondly, species will be further ranked in their likely importance in the surrounding species pool given the coverage of surrounding habitats with which they maybe more or less associated . In essence, the method used to define species-pools for site-based predictions can be simplified for 1km squares. If we can identify where in each 1km square the sensitive habitat is, then exactly the same method could be used with species pools reflecting the Broad Habitat composition in a buffer zone around each habitat area. The simplest method, requiring the least processing would be based solely on the % cover of habitats in each 1km square irrespective of location with in the square.

Our intention is to focus initially on Common Standards Monitoring indicators since changes in these species have an acknowledged impact on the conservation status of the habitat concerned.

2.Policy relevance

A key objective of the Umbrella work program is to generate model-based predictions of change in soil conditions and key plants species for sensitive habitats and soils in British 1km squares. This Task is strongly linked to the Defra Dynamic Modelling Umbrella coordindated by Chris Evans of CEH Bangor. This work is considered a high priority as a call for dynamic modelling output for terrestrial systems is expected in 2007/2008.

3.Key findings

Steps to estimate species pool of potential immigrants based on dispersal traits, species' broad habitat preferences and local broad habitat composition have been developed. These include:

1. Determine $10km^2$ species pool using BRC data or use site species lists where available.
2. Determine broad habitat extent in site (s), in 1500m buffer (b1) around site and in further 1500m buffer (b2). This step uses LCM 2000 and therefore assumes it is accurate at least in the identity and rank abundance of Broad Habitats.
3. Derive an approximate abundance weighting for each species in each zone (s, b1 and b2) using its preference index for each broad habitat as published by Hill & Preston (2003) in combination with the abundance of each broad habitat ie. high indices will reflect a large extent of the broad habitat with which each species is most associated. The index is worked out for species *j* as the sum of the products of multiplying each broad habitat proportion (a value between 0 and 1) by the preference index for the species and the broad habitat. This gives a maximum value of 4. So the index is rescaled by dividing this sum by 4.
4. Multiply the pool abundance index from 3 by the species' dispersal index.
5. Sort the CSM indicator table in descending order of the index in 4.
6. Interpret the table on the assumption that each component of the above index is reliable and realistic at the particular site, and that the dispersal ranking is an accurate reflection of real dispersal potential if appropriate vectors are in place.
7. Hence, species with high indices are expected to be most likely to disperse into the monitored patch. Establishment is then hypothesised to be favoured by increased habitat suitability predicted by MAGIC/VSD linked to GBMOVE.

<u>Index construction for Purple Moor-grass at Budworth common</u>

Broad Habitat composition in site with preference indices in brackets

Bog 3% (4)
Dwarf Shrub Heath 80% (2)
Acid grassland 5% (1)

Sum of proportions * preference

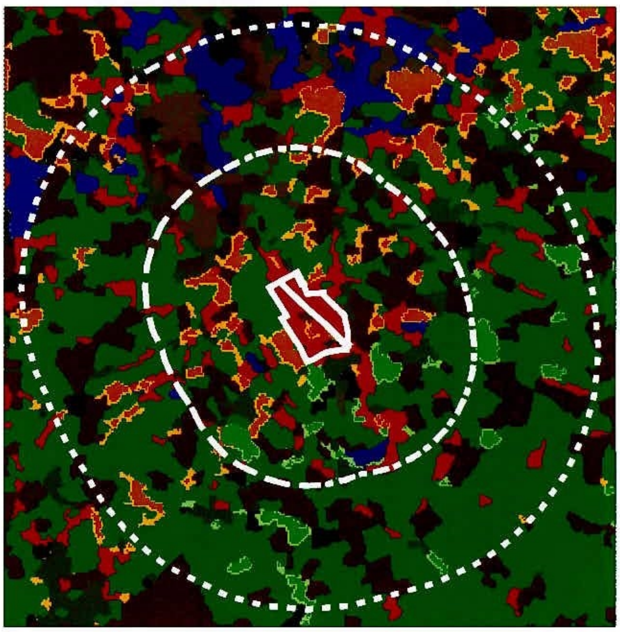
= (0.03 x 4) + (0.8 x 2) + (0.05 x 1)

= 1.77

Index = 1.77/ 4 = 0.44

Then multiply by dispersal index (high = more easily dispersed)

0.44 x 0.43 = **0.19**



This will be repeated for all CSM indicators NOT recorded in the monitored vegetation, then ordered table by index in descending order.

Appendices

First Draft of Mini-Syntheses

The impacts of acid and nitrogen deposition on:

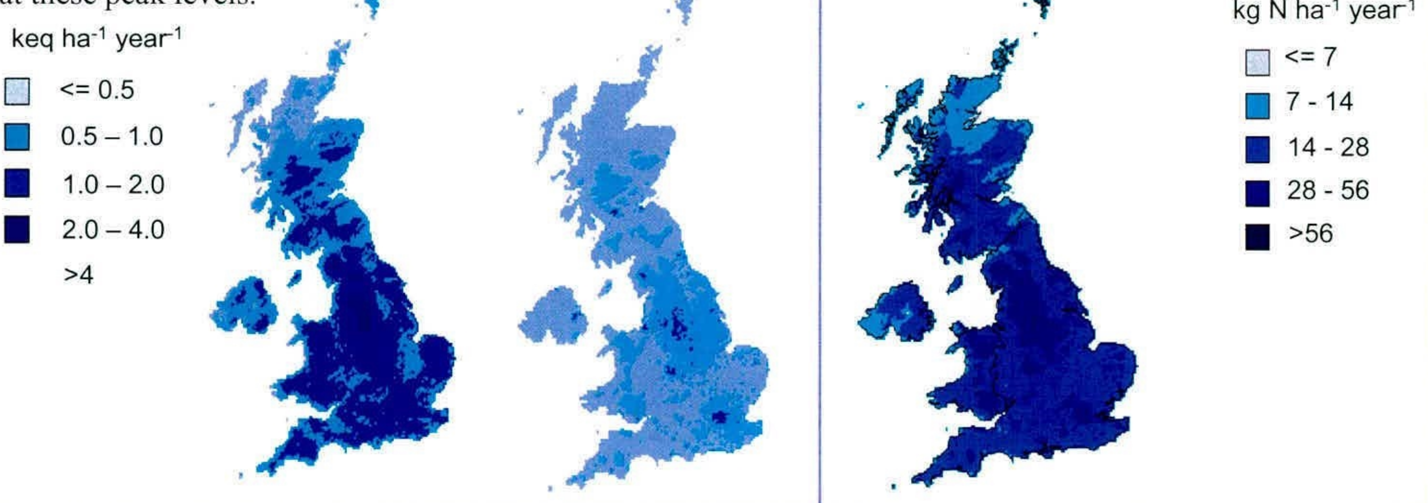
Sand dune habitats

Sand dune habitats are one of the most natural remaining vegetation types in the UK. They support over 70 nationally rare or red-data book species, and are a refuge for many species lost due to agricultural improvement of other lowland habitats. The open dune habitats in particular are important for a range of species: plants which are intolerant of competition; insects which require some bare soil for burrowing; and for threatened reptiles and amphibians such as sand lizard, natterjack toad and great-crested newt.

Sand dunes are sensitive to many pressures, including: habitat loss, sea-level rise, climate change, agricultural improvement, tourist pressure, lack of management, and over-stabilisation. Nitrogen deposition is thought to be a major contributor to over-stabilisation in UK dune systems.

The distribution of inputs of acidity and nitrogen across the UK

Energy production through the combustion of fossil fuels results in the emission of nitrogen oxides and sulphur dioxides into the atmosphere. Food production results in the emission of ammonia into the atmosphere due to emissions from farm animal units and nitrogen oxides emissions linked to intensive fertiliser use. These pollutants are transported in the atmosphere affecting the quality of the rain and air quality across the UK. This has resulted in acidification of soils and waters in acid-sensitive areas such as many upland habitats and has also contributed to nitrogen enrichment of semi-natural areas. Reductions in emissions due to policy control measures have achieved a reduction in the quantity of sulphur and nitrogen oxides falling on different habitats but unfortunately due to increases in emissions from shipping recovery has not been as fast as hoped for. Ammonia emissions increased sharply from 1950s to 2000 but currently remain at these peak levels.



Nitrogen (NOx + NHx) deposition measured 2001-2003

Sulphur deposition measured 2001-2003

Total acid deposition (S + NOx + NHx) measured 1999-2001

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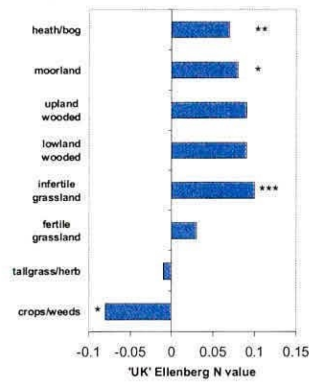
Evidence of acidification and N-enrichment effects at the national scale

There are various sources of information which indicate vegetation, soils and waters have been affected by acidic and nitrogen deposition. A review of the evidence for the UK was brought together by the National Expert Group on Transboundary Air Pollution (http://www.nbu.ac.uk/negtap/home.html). The evidence for nitrogen enrichment of vegetation includes two national monitoring programmes – the Countryside Survey and the New Plant Atlas for the UK – which identified shifts in species composition towards more nutrient-demanding species in the latter half of the 20th century (Preston et al. 2002, Haines-Young et al. 2003) (e.g. Figure 1).



The change in UK Ellenberg N score for different habitats between 1990 and 1998 taken from the most recent UK Countryside Survey (redrawn from Haines-Young et al. 2003). A low value is associated with low N fertility and high value with high N fertility. Statistically significant changes are marked as * $P<0.05$, ** $P<0.01$, *** $P<0.001$.

Bron will improve on figure box – my attempt at the moment

Figure showing trend in N and acid deposition during 20th C with embedded figure of recovery plot perhaps

Changes in the acidity of soils and waters have also been reported with declines in acidity recorded in some areas perhaps reflecting the success of emission policies to reduce levels of acid deposition in the environment although this variable across the country with less recovery in the north and west due to increases in emissions from shipping.

Why does air pollution affect our soils, vegetation and waters?

Text on why changes in species composition (competitive balance, increase in biotic/abiotic stresses etc), acidification processes and transfers etc. I can fill in but if you feel like helping….

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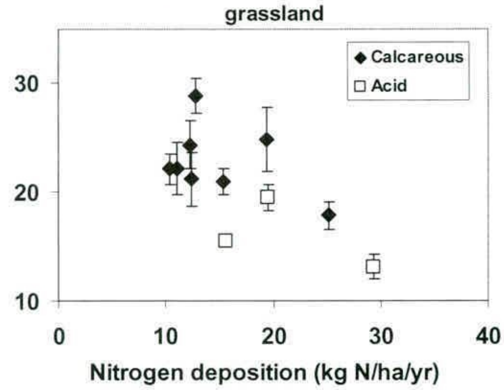
Current evidence for air pollution effects in sand dunes

Although physical processes, hydrology and succession have been extensively studied in sand dunes, research into air pollution effects in sand dunes is limited. A review on the effects of nitrogen deposition on sand dune habitats concluded that they were likely to be at risk of eutrophication. The greatest impact is likely to be on the early successional communities which are important for many of the sand dune rarities.

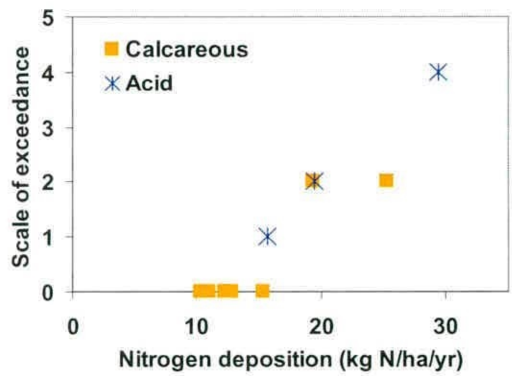
A survey approach of UK sand dunes along a gradient of nitrogen deposition showed significant correlations of vegetation, soil and groundwater parameters with increasing nitrogen deposition (Jones et al. 2004). In semi-fixed (open) dune habitats, cover of marram grass (*Ammophila arenaria*) and total biomass increased. In fixed dune grasslands, plant species diversity decreased (Figure 1) and biomass increased. Soil parameters showed surprisingly that the C:N ratio increased and available nitrogen decreased. Dissolved organic nitrogen (DON) concentrations in groundwater also increased. Experimental evidence from sand dune mesocosms in the Netherlands has shown an increase in graminoid cover and a decline in herbaceous species with increased nitrogen deposition (van den Berg et al. 2005). As a result of UK and Dutch studies, the suggested critical load range for sand dunes is 10 – 20 kg N ha^{-1} yr^{-1} (Figure 2).

Fig 1. Species richness, fixed dune grassland



Nitrogen retention in sand dune soils is poor, due to the low levels of organic matter. Nitrogen addition experiments in the Netherlands showed that leaching of nitrogen varied between 0 - 70 % of inputs. Leaching was highest in young calcareous soils and lowest in acidic soils with grazed vegetation (ten Harkel et al. 1998). A UK study under ambient nitrogen deposition, showed that total leaching losses of nitrogen varied from 15 – 65 % of inputs, usually dominated by DON. Losses were lowest in ungrazed vegetation on calcareous soils but, in contrast to the Netherlands, were highest in rabbit grazed vegetation (Jones et al. 2005).

Fig 2. Critical load exceedance



In UK dunes, the main impact of nitrogen is that of eutrophication. Acid deposition has relatively little impact because sand dune soils in the UK are generally well-buffered, with the exception of a few acidic dune systems.

However, in the Netherlands where the sand usually has a lower initial carbonate content, both acidification and eutrophication have resulted in a decline of rare basiphilous pioneer species in dune slacks (Sival & Strijkstra-Kalk, 1999). Furthermore, in the UK, acid dune systems appear to be more sensitive to nitrogen deposition (Figure 2). Phosphorus limitation may reduce vegetation responses to excess nitrogen deposition. However, as soil pH approaches pH 5, phosphorus becomes more available to plants, thus increasing the likelihood of adverse impacts of nitrogen deposition.

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How will climate change and management affect the impacts of air pollution?

Management can be used to mitigate the effects of nitrogen deposition. Both mowing and grazing tend to increase species diversity, and mowing (with removal of cuttings) removes nitrogen from the system. As mowing dune vegetation is impractical on a large scale, managed grazing is used to reduce the vigour of competitive species, and to retain species of conservation interest. However, the net removal of nitrogen by grazers is almost negligible. Grazing does, however, reduce the rate of nitrogen accumulation and, over an extended period of time, rabbit grazing effectively reduces the soil nitrogen pool by slowing accumulation compared with ungrazed habitats. Natural grazers such as rabbits play a key role in controlling vegetation growth, but need to be managed in conjunction with conventional grazing to allow for natural fluctuations in rabbit populations. More extreme management options such as turf-stripping or topsoil inversion can be used to remove the eutrophic surface layers and return dune habitats to an earlier successional stage.

The effects of climate change and the interactions with air pollution are uncertain. In many areas, sea-level rise will result in loss of sand-dune habitat, and the corresponding rise in water table will increase the extent of dune slacks. Early successional habitats, already impacted by nitrogen deposition, are most at risk. Climate change may increase or decrease mobility of dune systems, depending on the balance of rainfall, storminess, wind direction and wind speeds, and rising temperatures may affect the ranges of key species.

UK actions being taken to help reduce air pollution

Text on emission control policies – UK and European. Basic desciption of critical loads etc. Success to date in reductions. Challenges ahead (NHy etc). I will do but feel free to suggest text….

Further Information

- NEGTAP (National Expert Group on Transboundary Air Pollution) 2001: Transboundary Air Pollution: Acidification, Eutrophication and Ground-Level Ozone in the UK. ISBN 1 870393 61 9. Available online at http://www.nbu.ac.uk/negtap/
- Countryside Survey 2000 - http://www.cs2000.org.uk/
- New Atlas of the British and Irish Flora, (2002). *Edited by C. D. Preston, D. A. Pearman and T. D. Dines.* ISBN: 0198510675
- Air Pollution Information System-- http://www.apis.ac.uk/

Key references

Jones, M.L.M. et al., 2004. Changes in vegetation and soil characteristics in coastal sand dunes along a gradient of atmospheric nitrogen deposition. Plant Biology 6 (5), 598-605.

Jones, M.L.M. et al.., 2005. Determining a nitrogen budget for Merthyr Mawr sand dune system. Final Report for CCW.

Sival, F.P., Strijkstra-Kalk, M., 1999. Atmospheric deposition of acidifying and eutrophicating substances in dune slacks. Water Air and Soil Pollution 116, 461-477.

ten Harkel, M.J. et al.., 1998. Water and solute fluxes in dry coastal dune grasslands: the effects of grazing and increased nitrogen deposition. Plant and Soil 202 (1), 1-13.

van den Berg, L.J.L. et al.., 2005. Effects of nitrogen enrichment on coastal dune grassland: A mesocosm study. Environmental Pollution 138 (1), 77-85.

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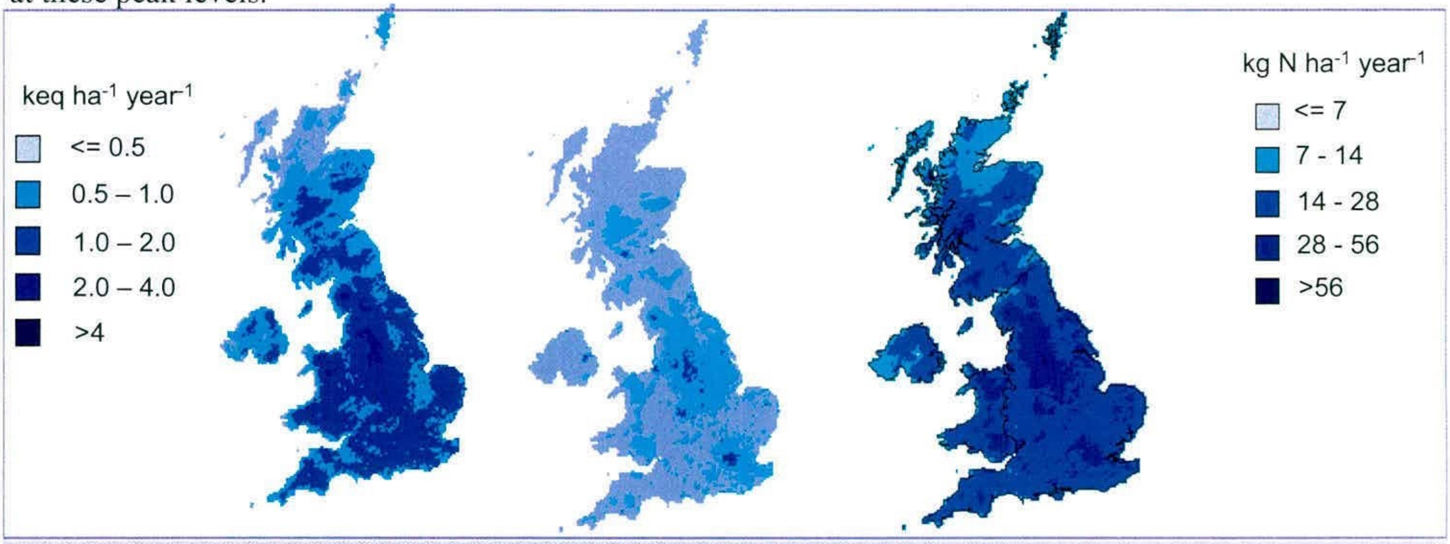
The impacts of acid and nitrogen deposition on:

Lowland Heath

The UK currently has approximately 58,000 ha of lowland heath, representing 20% of the international total for this habitat. Lowland heathland occurs at altitudes of less than 300m and is typically associated with nutrient-poor, often sandy, soils. Both wet and dry heathlands are priority habitats for nature conservation and, in addition to supporting a diverse flora and fauna, heathlands also have a high amenity value. Lowland heaths are managed systems, requiring the regular export of nutrients to maintain nutrient-poor conditions. However, since the late 19^{th} Century, a decline in traditional management practices, together with changes in land use and increasing urbanisation, have resulted in the loss of large areas of heathland. More recently, increased rates of atmospheric nitrogen deposition are believed to have contributed to heathland decline throughout Europe.

The distribution of inputs of acidity and nitrogen across the UK

Energy production through the combustion of fossil fuels results in the emission of nitrogen oxides and sulphur dioxides into the atmosphere. Food production results in the emission of ammonia into the atmosphere due to emissions from farm animal units and nitrogen oxides emissions linked to intensive fertiliser use. These pollutants are transported in the atmosphere affecting the quality of the rain and air quality across the UK. This has resulted in acidification of soils and waters in acid-sensitive areas such as many upland habitats and has also contributed to nitrogen enrichment of semi-natural areas. Reductions in emissions due to policy control measures have achieved a reduction in the quantity of sulphur and nitrogen oxides falling on different habitats but unfortunately due to increases in emissions from shipping recovery has not been as fast as hoped for. Ammonia emissions increased sharply from 1950s to 2000 but currently remain at these peak levels.



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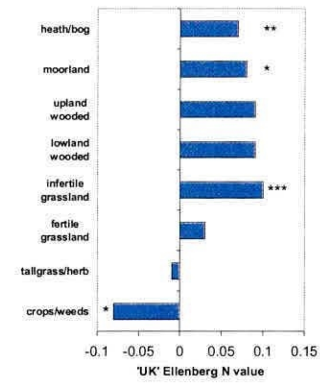

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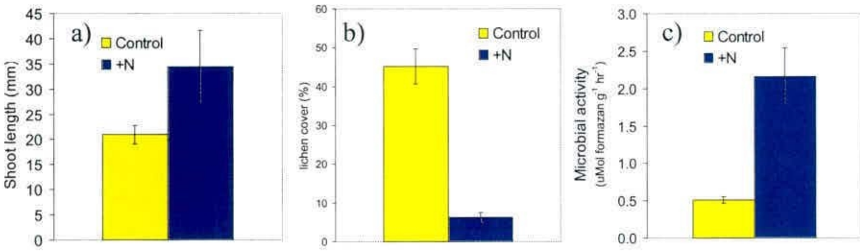
Current evidence for air pollution effects on lowland heaths

Research in the Netherlands, Denmark and the UK has demonstrated a variety of effects of nitrogen deposition and, to a lesser extent, acidity on the growth, chemistry and species composition of heathland vegetation, and on nutrient cycling within heathland ecosystems. Results from long term nitrogen-addition experiments at lowland heaths in north-west and southern England (right) have shown that even relatively small increases in nitrogen inputs result in increased heather growth, earlier bud burst and higher shoot nitrogen concentrations. These changes are, however, associated with increased sensitivity to drought and frost, and faster growth of the heather beetle, an insect which feeds exclusively on heather and which is responsible for wide scale damage to heather plants.

Experimental nitrogen addition at Thursley Common NNR

In the Netherlands, prolonged increases in nitrogen deposition are associated with increased dominance of heathland grasses, such as wavy hair grass (*Deschampsia flexuosa*) and purple moor grass (*Molinea caerulea*), at the expense of heather (*Calluna vulgaris*) (Aerts & Heil, 1993). A recent nationwide survey has shown that the occurrence of the dominant heathland species (*C. vulgaris* and *Erica cinerea*), and heathland habitat, has decreased significantly in the UK during the past 20 years (CS2000). However, whilst there is some evidence linking nitrogen deposition with a change in heathland community composition in the UK, field experiments have shown only a transient increase in grass species in response to nitrogen addition (Wilson, 2004; Barker *et al*., 2004).



Effects of nitrogen addition (30 kg ha^{-1} yr^{-1}) on a) Calluna *growth, b) lichen % cover and c) soil microbial activity at a lowland heathland in southern England.*

Lichens and bryophytes are considered to be the most sensitive components of most ecosystems and there is evidence of negative effects of nitrogen on sensitive heathland species, and associated changes in the lower plant community (Haworth, 2005; Carroll *et al.,* 1999).

Although nitrogen deposition is the major pollutant issue for lowland heathland, acidity and high concentrations of ammonium ions have also been shown to have direct toxic effects on sensitive herbaceous species in Dutch heathlands (de Graaf *et al*., 1998). Whilst the effects of increased nitrogen deposition are most clearly seen above-ground, changes below-ground are also observed, with important consequences for plant nutrient and water relations. Typical below-ground responses include a reduction in *Calluna* root growth, a build up of soil nitrogen stores and changes in the rate of nutrient cycling.

Measuring the fate of added nitrogen in heathland soils

The evidence to date indicates that a large proportion of nitrogen inputs is immobilised in the soil microbial biomass. Changes in microbial community composition and an increase in microbial activity are associated with accelerated rates of decomposition and faster nutrient cycling, factors which may favour the growth of more nutrient-demanding species, such as grasses, over slower growing dwarf shrubs. Current and future legislation will reduce emissions of nitrogenous pollutants, but little is known about the ability of semi-natural ecosystems to recover from the effects of eutrophication. Ongoing work at a lowland heath in Surrey indicates that recovery will be a slow process, with the effects of earlier nitrogen inputs persisting for many years after additions cease.

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How will climate change and management affect the impacts of air pollution?

Air pollution is not the only driver of ecosystem change; climate change is likely to have detrimental effects on heathland vegetation and alter nutrient cycling. Research has shown that nitrogen addition increases the sensitivity of heather to drought; climate change may result in even greater levels of drought injury in the field, particularly in combination with elevated nitrogen deposition. Another important issue is that temperature is frequently a limiting factor for insect and microbial performance; warmer temperatures are therefore likely to result in increased herbivory and, potentially, faster nutrient cycling. Lowland dry heathlands typically occur on well drained, sandy soils, with relatively limited water holding capacity. Predicted changes in summer rainfall and temperature may result in a greater frequency of uncontrolled, summer fires, with detrimental effects on soil structure and seed bank, as well as heathland fauna.

Habitat management, in the form of controlled (low temperature) burning, turf cutting, mowing or grazing, is used as a tool to maintain low nutrient levels in lowland heaths. However, recent results from both experiments and modelling studies indicate that frequent, intensive management (for example, turf cutting or mowing with litter removal) is needed to retain nutrient-limited conditions at many heathland sites under current levels of nitrogen deposition.

Management burn of an experimental heathland plot

UK actions being taken to help reduce air pollution

Text on emission control policies – UK and European. Basic desciption of critical loads etc. Success to date in reductions. Challenges ahead (NHy etc). I will do but feel free to suggest text….

Further Information

- NEGTAP (National Expert Group on Transboundary Air Pollution) 2001: Transboundary Air Pollution: Acidification, Eutrophication and Ground-Level Ozone in the UK. ISBN 1 870393 61 9. Available online at http://www.nbu.ac.uk/negtap/
- Countryside Survey 2000 - http://www.cs2000.org.uk/
- New Atlas of the British and Irish Flora, (2002). *Edited by C. D. Preston, D. A. Pearman and T. D. Dines.* ISBN: 0198510675
- Air Pollution Information System-- http://www.apis.ac.uk/

Key references

- Aerts, R. & Heil, G.W. (Eds.)(1993) *Heathlands: patterns and processes in a changing environment.* Kluwer, Dordrecht
- Barker, C.G., Power, S.A., Bell, J.N.B. and Orme, D. (2004) Effects of habitat management on heathland response to atmospheric nitrogen deposition. *Biological Conservation*, **120**, 41-52.
- Carroll, J.A., Caporn, S.J.M., Cawley, L., Read, D.J. & Lee, J.A., 1999. The effect of increased deposition of atmospheric nitrogen on *Calluna vulgaris* in upland Britain. *New Phytologist*, **141**, 423-431.
- Haworth, B. (2004). TITLE?? PhD thesis, University of Bradford.

- De Graaf, M. C. C.,Verbeek, P. J. M.Bobbink, R., Roelofs, J. G. M. (1998) Restoration of species-rich dry heaths: the importance of appropriate soil conditions. *Acta Botanica Neerlandica*, **47**, 89-111.
- Wilson, D. (2003). *Effect of nitrogen enrichment on the ecology and nutrient cycling of a lowland heath.* Ph.D. thesis, Manchester Metropolitan University, UK.

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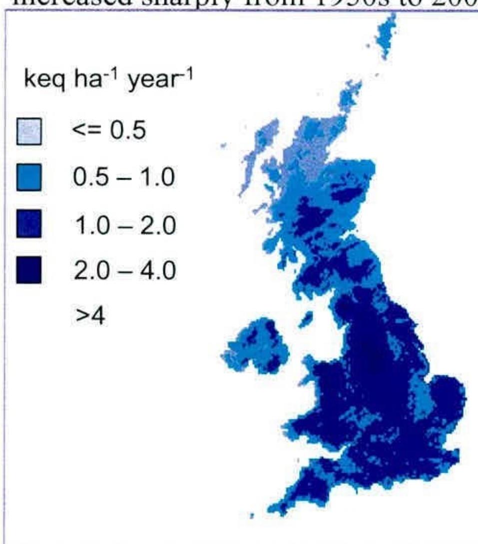
The impacts of acid and nitrogen deposition on:

Upland Heath

Upland heath, a priority UKBAP habitat, covers extensive areas of moorland between enclosed farmland and montane regions in the north and west of the UK. The dwarf shrub vegetation is a feature of the cool climate and the nutrient poor, acidic surface soils (usually thin peats or peaty podzols). A product mainly of burning and grazing management, its plant community is dominated by heather (*Calluna vulgaris*), and other shrubs including bilberry (*Vaccinium myrtillus*) and crowberry (*Empetrum nigrum*). The economically and ecologically important vegetation provides grazing for grouse, sheep and deer, and habitat for diverse bryophytes, lichens, invertebrates and raptor birds. Considerable areas were lost in the 20th century due to changing land use, over-grazing and management neglect. In parts of Britain, high levels of sulphur dioxide and acid rain have damaged the soils and vegetation of upland heath and related moorland ecosystems. More recently, nitrogen pollution has become recognised as a real, continuing threat to the structure and function of these nutrient-poor communities.

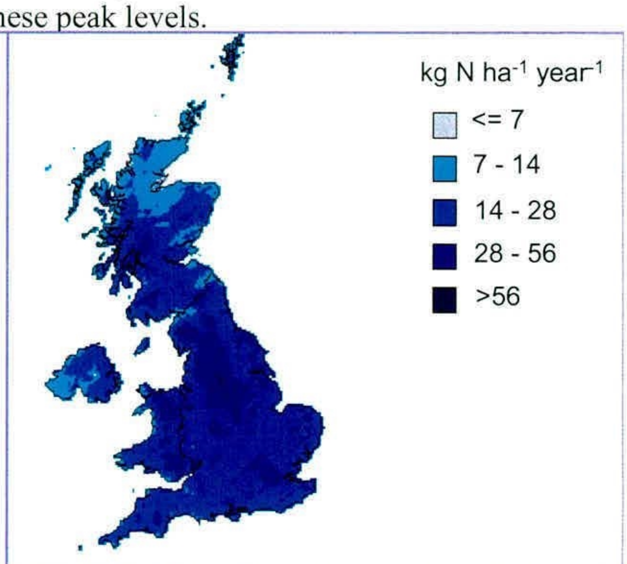
The distribution of inputs of acidity and nitrogen across the UK

Energy production through the combustion of fossil fuels results in emission of nitrogen oxides and sulphur dioxides into the atmosphere. Food production results in the emission of ammonia into the atmosphere due to releases from farm animal units and nitrogen oxides emissions from soils linked to intensive fertiliser use. The pollutants are transported in the atmosphere affecting the quality of the rain and air quality across the UK. This has acidified soils and waters in acid-sensitive areas such as many upland habitats and also contributed to nitrogen enrichment of semi-natural areas. Reductions in emissions due to policy control measures have achieved a reduction in the quantity of sulphur and nitrogen oxides falling on different habitats but due to increases in emissions from shipping recovery has not been as fast as hoped for. Ammonia emissions increased sharply from 1950s to 2000 but currently remain at these peak levels.



Nitrogen (NOx + NHx) deposition measured 2001-2003

Sulphur deposition measured 2001-2003



Total acid deposition (S + NOx + NHx) measured 1999-2001

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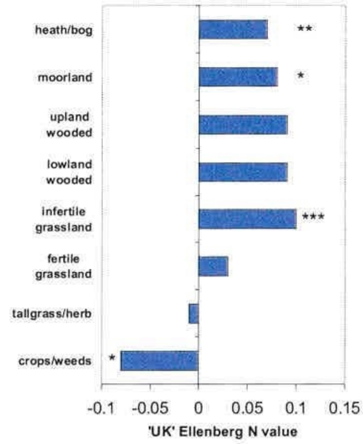
Evidence of acidification and N-enrichment effects at the national scale

Various sources of information indicate that vegetation, soils and waters have been affected by acidic and nitrogen deposition. The evidence for the UK was reviewed by the National Expert Group on Transboundary Air Pollution (http://www.nbu.ac.uk/negtap/home.html).
The evidence for effects of nitrogen enrichment on vegetation includes two national monitoring programmes – the Countryside Survey and the New Plant Atlas for the UK – which identified shifts in species composition towards more nutrient-demanding species in the latter half of the 20th century (Preston et al. 2002, Haines-Young et al. 2003) (e.g. Figure 1).
Bron will improve on figure box – my attempt at the moment



Change in UK Ellenberg N score for habitats between 1990 and 1998 taken from the UK Countryside Survey (redrawn from Haines-Young et al. 2003). A low value is associated with low N fertility and high value with high N fertility. Statistically significant changes are marked as * $P<0.05$, ** $P<0.01$, *** $P<0.001$.

Why do inputs of acidity and nitrogen across the UK matter to upland heaths?

The acidity of soils supporting upland heaths depends upon a delicate balance between inputs of base cations (mainly Ca^{2+}, Mg^{2+}, K^+ and Na^+) and acidity (hydrogen ions) from the atmosphere and base cations from weathering of soil minerals. Defra-funded research showed that for many upland heaths the latter inputs are small compared to inputs in rain and dust. Soil acidity then depends upon atmospheric inputs of acidity and base cations – the more acid the rain, the more acid the soil, and more acid the water draining from soil to streams and lakes. Peat soils are purely organic, and may be regarded as the most acidification-sensitive unless fed by water draining from mineral soils up-slope as they have no weathering minerals to release any base cations.
Should we worry if already acidic soils get a bit more acidic? We should, because the plant community has evolved to tolerate a certain level of soil acidity. Soil acidification slows the growth rate of plants, some more than others, so communities change. Research showed that decomposition of plant litter slows down, substantially at heavily polluted sites, and this is not sustainable in the longer term. Peat acidification can reduce the population of enchytraeid worms, which are important to the first stage of litter decomposition, and can reduce substantially the colonization of heather roots by beneficial mychorizal fungi. These biological effects are important to ecosystem sustainability. Equally of concern is the acidification impact upon waters draining into streams. As soils acidify from the top down, water draining from near-surface soils during heavy storms may make acid episodes at high discharge more severe. The link between peat acidification and acid rain across Scotland was established in 1989 (Skiba *et al.* 1989). Simulation experiments with artificial acid rain (possible for peat because mineral weathering rates are not an issue) allowed models to be developed to predict the acidity of upland peats anywhere in Britain. These were subsequently validated by direct measurement along pollution gradients. This allowed critical loads to be set for acidifying pollutants, the assumption being made that acidification only within the uncertainty of the model prediction would be insignificant, and therefore tolerable.

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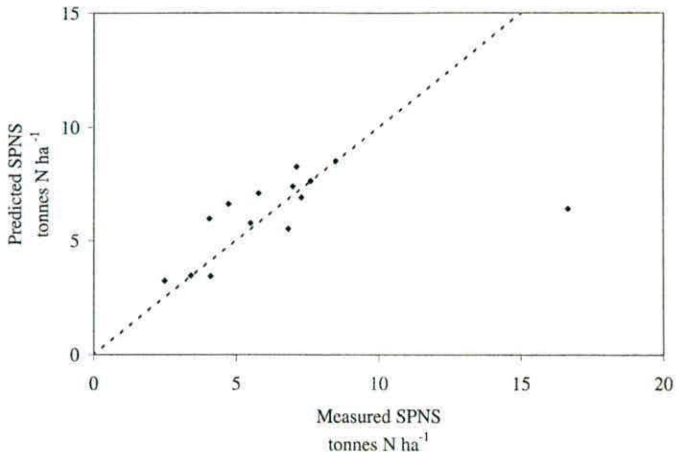
More recently DEFRA research at York showed that the acidity of soil solution in peat and the effective acidity of rainfall (corrected for dry deposited acidifying gases and the concentrating effects of evaporation and transpiration) were identical, and this concept is now used to set critical loads of acidity for peat soils. However it would be over-precautionary to try to protect acidity of soil solution above the limit set by the presence of naturally occurring organic acidity in soils, so this is taken as a naturally safe limit.

Simulation experiments have been performed over almost a decade have shown that *Calluna vulgaris* itself is very tolerant of acidity as sulphuric acid. The pots to the far left with no decomposition of residual *Calluna* litter and no regeneration are receiving very acidic simulated rain (with an effective pH of 2.15), but adjacent pots at pH 2.65 have strong regeneration and a diverse heath community. Similar experiments with simulated ammonium deposition show the community is much less tolerant of high nitrogen pollution.

A regional survey of upland heath podzols across Scotland, where samples were selected within selected altitude and slope constraints, clearly suggested a strong link between the soil profile nitrogen storage (SPNS) and the amount of nitrogen deposition at each site. In the graph on the right, SPNS has been predicted from N deposition flux and precipitation at 14 sites. But does this accumulation of soil nitrogen, almost all in the soil organic matter, really matter? Note that one point does not fit on the line. This is the most grossly polluted site. Beyond a certain critical N input, nitrate and ammonium may be leached, rather than retained in the soil.



Why does air pollution with ammonia and nitrate affect our soils, vegetation and waters?

When rainwater composition is compared with water dripping through heathland plant canopies, a large part of each nitrogen species is removed, taken up either by foliage or by leaf micro-flora. Thus deposition of nitrogen species may affect vegetation directly, quite apart from any effect that it has upon soil or soil solution. However, it also impacts upon soils and their associated drainage waters. Ammonium positively charged cations reaching soil are held initially on negatively charged cation exchange sites. Unless ammonium is taken up by plants or micro-organisms, or converted to nitrate, eventually the precipitation (rain, etc.) and drainage water will have similar ammonium concentrations. This happens at grossly nitrogen-polluted sites such as part of the South Pennines. Nitrate anions are more mobile in soil, but are taken up by plants and/or soil micro-organisms. However at heavily nitrogen-polluted sites, plant requirements may be exceeded, and nitrate leaching occurs, even in summer when plant uptake is at a maximum. Defra research at York has shown that nitrate leaching from upland heathlands may be modelled from nitrogen deposition and catchment maximum altitude and slope characteristics. The reduction in lower plant cover in the moorlands of the most polluted regions also means that this vegetation, especially the moss layer, is less able to absorb and retain the deposited nitrogen (Curtis et al, 2005).

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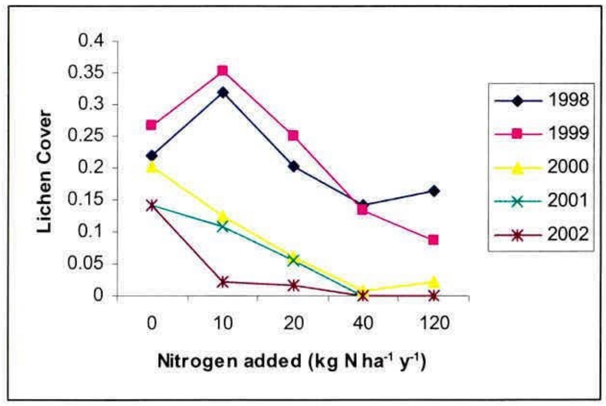
Evidence for air pollution effects on upland heaths – now and the future

The long term effects of N pollution on heather (*Calluna vulgaris*) are uncertain. Some pot experiments found adverse effects on roots and mycorrhizas, yet studies around the country found heather grew well even in the most polluted regions (Milne *et al* 2002). In one field experiment, inputs of N over 17 years have not substantially changed the condition of the dominant heather plants. Rather than the higher plants, it is the lichen, liverwort and mosses that provide the main above-ground biodiversity in upland heath and the same experiment in North Wales found these 'cryptogams' to be highly sensitive to nitrogen pollution (right).



Damage to the dominant shrubs may come indirectly, as episodic environmental stresses, such as freezing, exposed conditions or heather beetle, can interact with N to cause increased injury in heather shoots (Carroll *et al*, 1999). Probably the key to maintenance of good heather condition in N-polluted regions is very regular management. Experimental work indicates that N inputs advance the development of heather, therefore, burning or cutting need to be practised more often. Overgrazing by sheep or deer should be prevented; Mitchell and Hartley (2005) treated experimental plots in the Cairngorms to N and found that heather suffered significant decline, with an increase in grasses, only when grazing was allowed.

UK actions being taken to help reduce air pollution

Further Information

•NEGTAP (National Expert Group on Transboundary Air Pollution) 2001: Transboundary Air Pollution: Acidification, Eutrophication and Ground-Level Ozone in the UK. ISBN 1 870393 61 9. Available online at http://www.nbu.ac.uk/negtap/
•Countryside Survey 2000 - http://www.cs2000.org.uk/
•New Atlas of the British and Irish Flora, (2002). *Edited by C. D. Preston, D. A. Pearman and T. D. Dines*. ISBN: 0198510675
•Air Pollution Information System-- http://www.apis.ac.uk/

Key references

•Skiba, U. et al., (1989) Peat acidification in Scotland. *Nature*, **337**, 68-89.

•Curtis et al 2005

•Mitchell & Hartley

•Carroll et al 1999

•Pilkington et al 2005

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The impacts of acid and nitrogen deposition on:

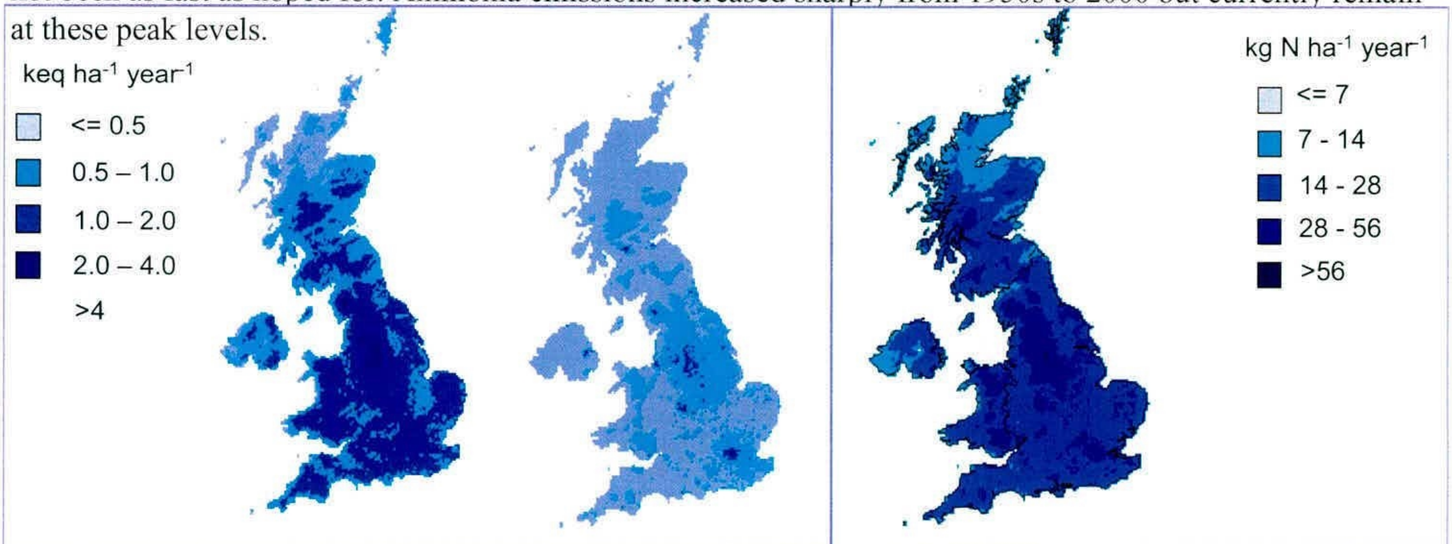
Montane Heath

Montane heathlands comprise a range of dwarf-shrub, moss and lichen dominated communities occurring above the potential tree-line. They represent the most extensive remaining areas of near-natural habitat in the UK and are highly valued for the unique range of arctic and alpine species which they support. Montane areas also form the headwaters of many river systems and play an important role in the regulation of water supply and quality for downstream uses such as drinking water and fisheries.

Montane moss and dwarf-shrub heaths have a restricted distribution south of the Scottish Highlands and are thought to have declined over the last 50 years. There is also evidence of a loss of characteristic mosses and lichens and an increase in grasses. This is thought to be a result of past and current overgrazing combined with the effects of nitrogen (N) deposition and, increasingly, climate change (Thompson & Brown 1992).

The distribution of inputs of acidity and nitrogen across the UK

Energy production through the combustion of fossil fuels results in the emission of nitrogen oxides and sulphur dioxides into the atmosphere. Food production results in the emission of ammonia into the atmosphere due to emissions from farm animal units and nitrogen oxides emissions linked to intensive fertiliser use. These pollutants are transported in the atmosphere affecting the quality of the rain and air quality across the UK. This has resulted in acidification of soils and waters in acid-sensitive areas such as many upland habitats and has also contributed to nitrogen enrichment of semi-natural areas. Reductions in emissions due to policy control measures have achieved a reduction in the quantity of sulphur and nitrogen oxides falling on different habitats but unfortunately due to increases in emissions from shipping recovery has not been as fast as hoped for. Ammonia emissions increased sharply from 1950s to 2000 but currently remain at these peak levels.



Nitrogen (NOx + NHx) deposition measured 2001-2003

Sulphur deposition measured 2001-2003

Total acid deposition (S + NOx + NHx) measured 1999-2001

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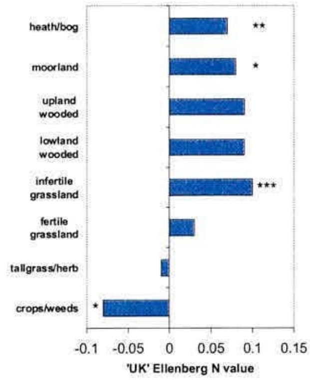
Evidence of acidification and N-enrichment effects at the national scale

There are various sources of information which indicate vegetation, soils and waters have been affected by acidic and nitrogen deposition. A review of the evidence for the UK was brought together by the National Expert Group on Transboundary Air Pollution (http://www.nbu.ac.uk/negtap/home.html). The evidence for nitrogen enrichment of vegetation includes two national monitoring programmes – the Countryside Survey and the New Plant Atlas for the UK – which identified shifts in species composition towards more nutrient-demanding species in the latter half of the 20th century (Preston et al. 2002, Haines-Young et al. 2003) (e.g. Figure 1).



The change in UK Ellenberg N score for different habitats between 1990 and 1998 taken from the most recent UK Countryside Survey (redrawn from Haines-Young et al. 2003). A low value is associated with low N fertility and high value with high N fertility. Statistically significant changes are marked as * $P<0.05$, ** $P<0.01$, *** $P<0.001$.

Bron will improve on figure box – my attempt at the moment

Figure showing trend in N and acid deposition during 20th C with embedded figure of recovery plot perhaps

Changes in the acidity of soils and waters have also been reported with declines in acidity recorded in some areas perhaps reflecting the success of emission policies to reduce levels of acid deposition in the environment although this variable across the country with less recovery in the north and west due to increases in emissions from shipping.

Why does air pollution affect our soils, vegetation and waters?

Text on why changes in species composition (competitive balance, increase in biotic/abiotic stresses etc), acidification processes and transfers etc. I can fill in but if you feel like helping….

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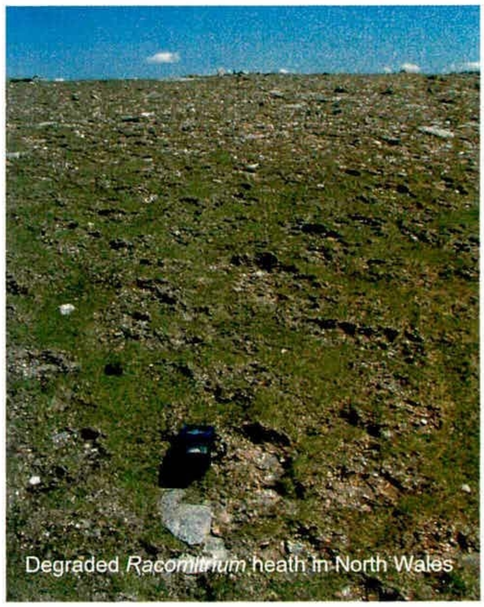
Current evidence for air pollution effects in montane heaths

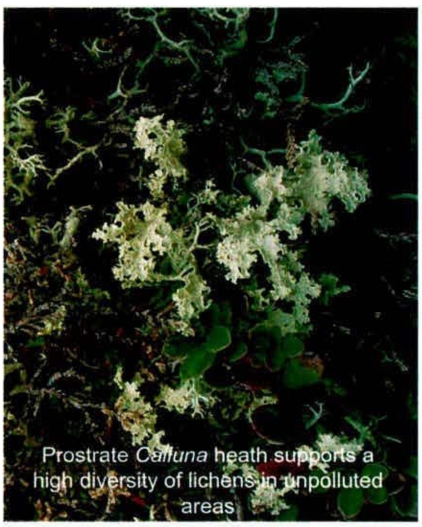
The few studies which have been carried out in montane heathland ecosystems have focussed on two distinct communities, prostrate *Calluna vulgaris-Cladonia arbuscula* heath (NVC H13) and *Carex bigelowii-Racomitrium lanuginosum* moss heath (NVC U10). The former is dominated by a carpet of prostrate dwarf shrubs and contains a high diversity of lichens, especially *Cladonia* species, while the latter is dominated by the montane moss *Racomitrium lanuginosum* (woolly fringe moss) with a range of grasses and sedges. Both are climatically-determined 'climax communities' and occupy large areas (sometimes several km^2) on exposed mountain summits and ridges in the UK.



Degraded *Racomitrium* heath in North Wales

The present-day distribution of montane heathland communities is restricted south of the Scottish Highlands where they are absent from many areas where they were recorded in the 1950s. Current summit vegetation in the Southern Uplands, northern England and north Wales often consists of impoverished variants of these communities, with a reduced cover of mosses and lichens and dwarf shrubs and a high prevalence of grasses. This degradation is thought to result from a combination of increasing N deposition (which currently reaches 56 kg N ha^{-1} y^{-1} in montane areas of the UK) and high grazing pressures, both of which occurred over the period 1950-present. Separating the effects of these two factors is not easy and interactions between the two may substantially increase their impacts. Degradation of the ecosystem is not limited to effects on the plant community composition; studies on the Carneddau plateau in north Wales (Britton et al 2005) have also shown reduced soil C:N ratios, loss of soil carbon from the most degraded areas and a high N content of both soils and plant tissues when compared with 'clean' sites in northern Scotland.

Both *Racomitrium* and *Calluna* have been shown to respond to N deposition by accumulating N in their tissues. *Racomitrium* in particular is adapted to take up nutrients deposited on its surface by rainfall as it has no roots and, hence, its N content closely corresponds to the ambient N deposition. Below 10 kg N ha^{-1} y^{-1} the moss is able to use additional N supplies for growth but, as the N deposition rate increases, excess N is accumulated in the tissues damaging the cell membranes of the moss and making them 'leaky', resulting in decreased growth and eventually shoot death (Pearce et al 2003). The result is a reduced cover and depth of the moss mat and increased N availability in the soil as nutrients are released from decaying tissue. Reduction in moss cover may lead to soil erosion or allow expansion of grasses and sedges able to use the increased N supply, this in turn may decrease light availability to the moss and further reduce growth. Loss of the moss carpet and the thick organic layer below which acts as both a sponge and a filter may also have important consequences for hydrology and water quality downstream.



Prostrate *Calluna* heath supports a high diversity of lichens in unpolluted areas

In prostrate dwarf-shrub heaths in the UK and Scandinavia, nitrogen deposition impacts have been demonstrated on several aspects of community structure and function (Britton In prep, Fremstad et al 2005). Nitrogen deposition as low as 10 kg N ha^{-1} y^{-1} causes a reduction in the cover and species richness of the diverse lichen community associated with this habitat. *Calluna,* however, responds by increasing its shoot growth as long as phosphorus is not limiting and there is evidence of increased enzyme production by the roots to allow more phosphate to be taken up to meet the increased demand. This community does not show the dramatic shift to grass dominance seen in lower altitude heaths, probably because of the limited amount of grasses present initially. Nitrogen addition also impacts the soil and soil waters causing rapid acidification of the soil and loss of essential plant nutrients such as calcium, magnesium and potassium into the soil water. Increased acidity also mobilises toxic ions such as aluminium and heavy metals which are poisonous to aquatic life.

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How will climate change and management affect the impacts of air pollution?


Winter-injury in *Calluna vulgaris* subject to high N load

The effects of air pollution on vegetation, soils and waters are not independent of other environmental change factors. Factors such as climate change and management may influence the sensitivity of ecosystems to air pollution. In some cases the effect of these drivers of change will be to reduce damage but in other cases they may act to amplify effects. In montane heathlands two issues have been highlighted as being of major concern. These are the impact of domestic and wild herbivore populations and that of climate change. Studies in *Racomitrium* heath (van der Wal et al 2003) have shown how grazing can enhance the negative effects of nitrogen deposition through trampling of the moss and deposition of dung, further enhancing grass growth. This interaction may be responsible for the degraded state of montane heaths south of the Highlands. Predicted future increases in drought frequency may also amplify nitrogen impacts. Montane habitats are exposed to extreme climates including the drying effects of high wind speeds both summer and winter and nitrogen deposition has been shown to increase drought sensitivity of key species. When also exposed to high levels of nitrogen deposition this can lead to increased winter injury in *Calluna* and reduced growth in *Racomitrium* (Jones et al 2002).

UK actions being taken to help reduce air pollution

Text on emission control policies – UK and European. Basic desciption of critical loads etc. Success to date in reductions. Challenges ahead (NHy etc). I will do but feel free to suggest text….

Further Information

•NEGTAP (National Expert Group on Transboundary Air Pollution) 2001: Transboundary Air Pollution: Acidification, Eutrophication and Ground-Level Ozone in the UK. ISBN 1 870393 61 9. Available online at http://www.nbu.ac.uk/negtap/
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Key references

•Biodiversity in montane Britain: habitat variation, vegetation diversity and some objectives for conservation. (1992) Thompson, D.B.A. & Brown, A. *Biodiversity and Conservation* **1**, 179-208.

•Interplay between nitrogen deposition and grazing causes habitat degradation. (2003) Van der Wal, R. et al. *Ecology Letters* **6**, 141-146.

•The influence of nitrogen deposition, competition and desiccation on growth and regeneration of *Racomitrium lanuginosum* (Hedw.) Brid. (2002) Jones, M.L.M. et al. *Environmental Pollution* **120**, 371-378.

•Impacts of grazing on montane heath vegetation in Wales and implications for the restoration of montane areas. (2005) Britton, A.J. et al. *Biological Conservation* **125**, 515-524.

•Physiological and growth responses of the montane bryophyte *Racomitrium lanuginosum* to atmospheric nitrogen deposition. (2003) Pearce I.S.K. et al. *New Phytologist* **160**, 145-155.

•Impacts of increased nitrogen supply on Norwegian lichen-rich alpine communities: a 10-year experiment. (2005) Fremstad E. et al. *Journal of Ecology* **93**, 471-481.

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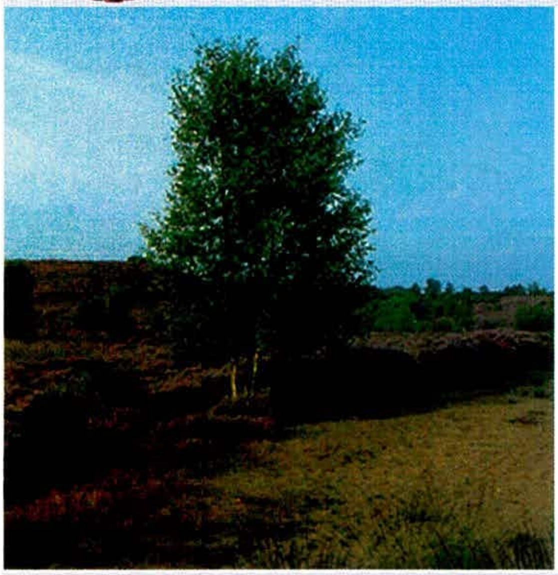
The impacts of acid and nitrogen deposition on:

Blanket and raised bogs

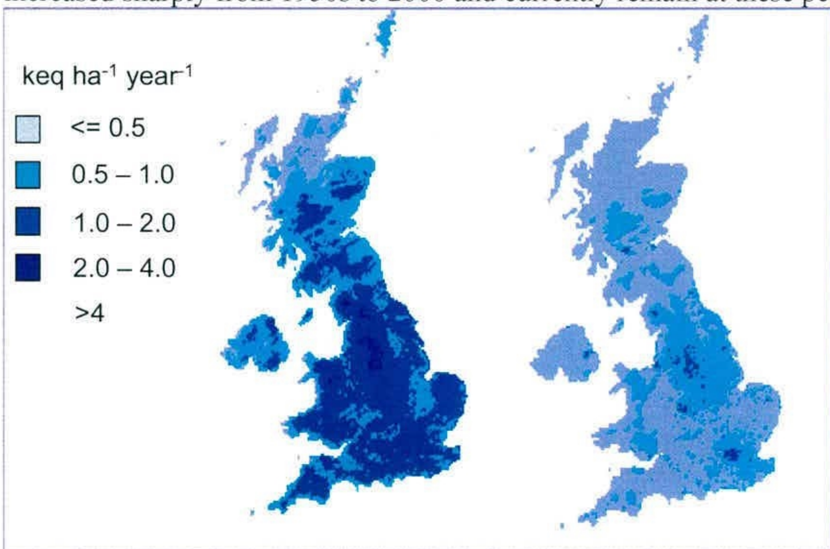


Blanket and raised bogs are peat based ecosystems with a restricted world distribution. In Great Britain, bogs cover around 1.5 million hectares being widespread north of the Wash especially in Scotland and Ireland and represent an internationally important conservation resource. Peat based bogs may represent thousands of years of organic matter accumulation making them amongst Britain's most ancient natural/semi-natural ecosystems. Bogs are valued for their specialised plant and bird communities and their ability to act as a sink for carbon. Today the state of British and many European bogs gives cause for concern. The Countryside Survey has shown a significant decline in two of the major species of NV classification type M19, *Calluna vulgaris* and *Eriophorum vaginatum* with no data for the keystone species *Sphagnum spp*. Bogs support specialised plant communities adapted and restricted to nutrient limited conditions sustained by the wet, often anoxic, acidic conditions. These conditions, which restrict decomposition, are generated by the unique properties of the keystone species *Sphagnum spp.* or 'bog moss'.

Today, apart from reclamation, drainage, for other land uses, one of the major threats to the sustainability of bogs comes from the enhanced deposition of reactive nitrogen. *Sphagnum* mosses, in addition to their naturally low decomposition rates are efficient scavengers of inorganic nutrients such as nitrogen. By 'locking up', filtering mineral N *Sphagnum* mosses help to exclude faster growing plant species with higher transpiration rates and the potential to lower the water-table. Without *Sphagnum* many bogs would be transformed into drier grass/tree dominated habitats, at the expense of all the specialized species. Unfortunately this ability to remove mineral nitrogen heightens their sensitivity to nitrogen, putting at risk the sustainability of these highly specialised bog communities

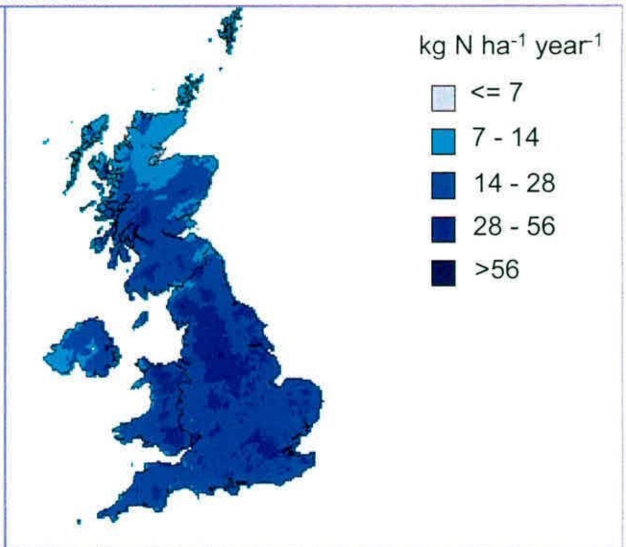
The distribution of inputs of acidity and nitrogen across the UK

Atmospheric reactive nitrogen comes from different sources. The combustion of fossil fuels, to provide energy, has released large amounts of nitrogen and sulphur oxides into the atmosphere and enhanced N and S deposition to ecosystems. Food production, especially via intensive livestock units and fertilizer use has further added to the problem of increasing N deposition in the form of reduced nitrogen. These pollutants can be deposited locally (ammonia) or transported for varying distances in the atmosphere to affect air and rainfall quality across the UK. This has resulted in acidification of soils and waters in acid-sensitive areas such as many upland habitats and has also contributed to nitrogen enrichment of semi-natural areas. Reductions in emissions have been achieved due to policy control measures. However, reductions in the quantity of sulphur and nitrogen oxides falling on semi-natural habitats, remote from sources, partly due to increases in emissions from shipping recovery, has not been as fast as hoped for. Ammonia emissions increased sharply from 1950s to 2000 and currently remain at these peak levels.



Nitrogen (NOx + NHx) deposition measured 2001-2003

Sulphur deposition measured 2001-2003



Total acid deposition (S + NOx + NHx) measured 1999-2001

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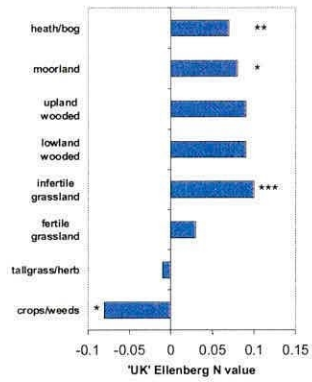
Evidence of acidification and N-enrichment effects at the national scale

There are various sources of information which indicate vegetation, soils and waters have been affected by acidic and nitrogen deposition. A review of the evidence for the UK was brought together by the National Expert Group on Transboundary Air Pollution (http://www.nbu.ac.uk/negtap/home.html). The evidence for nitrogen enrichment of vegetation includes two national monitoring programmes – the Countryside Survey and the New Plant Atlas for the UK – which identified shifts in species composition towards more nutrient-demanding species in the latter half of the 20th century (Preston et al. 2002, Haines-Young et al. 2003) (e.g. Figure 1).



The change in UK Ellenberg N score for different habitats between 1990 and 1998 taken from the most recent UK Countryside Survey (redrawn from Haines-Young et al. 2003). A low value is associated with low N fertility and high value with high N fertility. Statistically significant changes are marked as * $P<0.05$, ** $P<0.01$, *** $P<0.001$.

Bron will improve on figure box – my attempt at the moment

Figure showing trend in N and acid deposition during 20th C with embedded figure of recovery plot perhaps

Changes in the acidity of soils and waters have also been reported with declines in acidity recorded in some areas perhaps reflecting the success of emission policies to reduce levels of acid deposition in the environment although this variable across the country with less recovery in the north and west due to increases in emissions from shipping.

Why does air pollution affect our soils, vegetation and waters?

Air pollutants affect the natural balance between living organisms and converts closed ecosystems into open ones with a consequent loss of control over what enters and leaves. While many air pollutants are nutrients they usually deposit individually, so that their use is usually restricted by the availability of other nutrients supplied by slower natural processes. Plants have evolved to optimise nutrient uptake and avoid excess uptake however the supply of nutrients from the atmosphere circumvents many of these control mechanisms leading to excessive uptake and the potential for toxicity.

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Current evidence for air pollution effects on bogs

The plant species composition of British bogs was surveyed and well documented in the latter half of the 20th century. Changes to both the bog resource and its associated plants since these surveys are now a priority issue for conservation bodies under the European Habitats Directive. In Scotland the area of bog declined by 21% between 1940 and 1980, mostly due to afforestation. Detrimental effects of air pollutants, primarily sulphate and nitrate deposition, have been confined to the Pennines and north Wales, areas in close proximity to large urban conurbations. The loss of *Sphagnum,* which now seems to be re-establishing in places, was most closely linked to elevated sulphur and nitrogen deposition although overgrazing and burning also played a role. Evidence of nitrogen deposition threatening the sustainability of bogs comes mainly from Europe and experimental studies. In Europe, especially The Netherlands and Denmark, many bogs occur in regions of intensive livestock farming and here the detrimental effects of reduced nitrogen, ammonia, are clear for all to see: grassy plains where once there were bogs. The majority of British bogs by comparison do not occur in areas of high nitrogen deposition, although most sites already receive the Critical Nitrogen Load of 5-10 kg N ha-1 y-1.

Nitrogen impacts on bogs have mostly been studied with respect to *Sphagnum* species. Once inputs of nitrogen exceed the CL a chain of events is set in motion: For a time, depending on the level of N input, *Sphagnum* sequesters/filters the reactive N without too much cause for concern up to twice the CL. Nitrogen loads in excess of this exceed the capacity for 'safe' storage as nitrate or amino acids and the increasing concentration of ammonium becomes toxic, attacking membranes and releasing reactive nitrogen into the bog water. This increase in reactive nitrogen favours fast growing, nitrogen demanding species which previously were excluded. Dutch work based on nitrogen addition experiments and surveying *Sphagnum* in bogs from sites with known nitrogen deposition suggest that 1.2 % is a critical value for the nitrogen content of *Sphagnum*. However, work here in the UK using more realistic nitrogen deposition scenarios suggest N status alone will not necessarily reveal the health status of *Sphagnum* or indicate its vitality. After four years of enhanced wet deposition, at up to six times the CL, and despite detrimental effects being observed within one year and significant accumulations of amino acids being measured, the actual cover of vital *Sphagnum* has not declined over the first four years.

Often detrimental effects of nitrogen are indirect or require the presence of another agent(s), such as drought, shading, pests and pathogens. The impacts of elevated nitrogen have often been fatal under conditions of sustained drought, which impacts on the water table and the biogeochemistry, and the accumulation of amino acids will increase the likelihood of pest and pathogen outbreaks. Likewise the availability of phosphorus and potassium especially, but all nutrients that impact on growth, will influence whether reactive nitrogen is used or accumulates in *Sphagnum*, and also the rate at which and type of species that will replace *Sphagnum* and transform the ecosystem.

Sphagnum is not the only bog species that is sensitive to nitrogen. Lichens such as the common matt forming *Cladonia portentosa* can be eradicated by high ammonia concentrations, such as found close to intensive animal units. *Calluna*, likewise, has died out near such units, partly due to increased vulnerability to desiccating winter winds, pests and pathogens. These species appear like Sphagnum to be less affected by wet deposited nitrogen applied frequently at low concentrations.

How will climate change and management affect the impacts of air pollution?

Climate change is likely to exert a large influence on bogs if the climate becomes drier, cloudbursts increase in intensity and frequency and mean temperatures increase or rainfall decreases. *Sphagnum* species can be very sensitive to desiccation stress, and too much nitrogen will exacerbate the problem, and vice versa. Both a reduction in rainfall and increasing temperatures will influence the chemistry of the soil, increasing the rate at which nutrients become more available to faster growing nitrogen loving species. This sets up a positive feedback increasing the competitive ability of faster growing species that use more water and lower the water table further and also shade out many of the under-storey mosses and lichens. One positive effect of peat drying out is the increase in oxidation processes and nitrification that tend to acidify the peat, helping to exclude the faster growing species and favouring *Sphagnum* species. Maintaining a low pH also helps limit the production of methane an important greenhouse gas.

Management intervention by increasing the amount of water draining into bogs, providing the chemical inputs are very low, and damming up drainage outflows could help offset a falling water table. The addition of phosphorus and or potassium can reduce the accumulation of nitrogen in *Sphagnum*, but whether this can provide a long – term solution may then depend on the availability of carbon dioxide, sufficient water and maintaining an acid pH. On bogs where *Racomitrium lanuginosum*, woolly hair moss is dominant reducing the grazing measure may help offset effects of climate change and elevated nitrogen inputs. The problem of invasive species may to some extent be controlled by limiting the type of plants grown in close proximity to bogs.

UK actions being taken to help reduce air pollution

Text on emission control policies – UK and European. Basic desciption of critical loads etc. Success to date in reductions. Challenges ahead (NHy etc). I will do but feel free to suggest text….

Further Information

- •NEGTAP (National Expert Group on Transboundary Air Pollution) 2001: Transboundary Air Pollution: Acidification, Eutrophication and Ground-Level Ozone in the UK. ISBN 1 870393 61 9. Available online at http://www.nbu.ac.uk/negtap/
- •Countryside Survey 2000 - http://www.cs2000.org.uk/
- •New Atlas of the British and Irish Flora, (2002). *Edited by C. D. Preston, D. A. Pearman and T. D. Dines.* ISBN: 0198510675
- •Air Pollution Information System-- http://www.apis.ac.uk/

Key references

Lamers LPM, Bobbink R, Roelofs JGM 2000 Natural nitrogen filter fails in polluted raised bogs. Global Change Biology 6, 583-586.

Van Breeman N 1995. How Sphagnum bogs down other plants. Trends in Ecology and Evolution 10, 270-275.

Tallis JH, Meade R, Hulme PD. 1997 (eds) Blanket Mire Degradation Causes, Cosequences and Challenges. Proceedings. Mires Research Group British Ecology Society. ISBN 0 7084 0597 5

Charman D, 2002 Peatlands and Environmental Change. John Wiley and Sons.

Limpens J 2003 Prospects for Sphagnum bogs subject to high nitrogen deposition. PhD thesis University of Wageningen.

Tomassen, HBM. 2004 Revival of Dutch Sphagnum bogs: a reasonable perspective PhD thesis University of Nijmegen.

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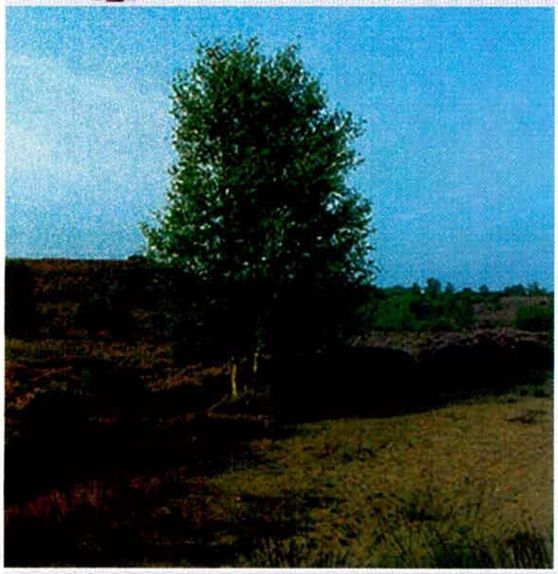
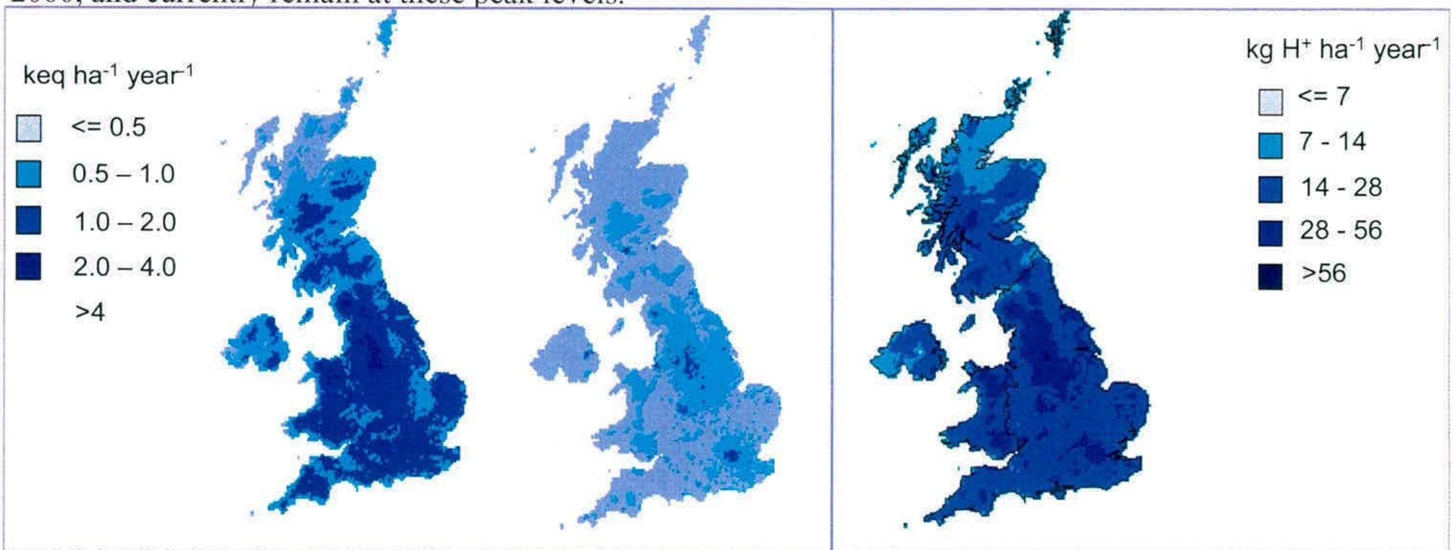
The impacts of sulphur and nitrogen deposition on: woodlands



Woodlands are valued for their high conservation status, contribution to the landscape, ability to store carbon and provision of public access and recreation opportunities. Recent surveys have shown no evidence of widespread damage from air pollution to forest trees, but excess nitrogen deposition has been implicated in observed changes in the composition of woodland plant communities. Recent plant surveys indicate that nitrogen demanding species such as bramble are becoming more abundant, while others, such as bilberry, are declining. It is also thought that a lack of regeneration due to under-management in recent decades and browsing by deer has also contributed to a degradation of ecological condition in some woodlands.

The distribution of inputs of acidity and nitrogen across the UK

The combustion of fossil fuels results in the emission of oxides of sulphur and nitrogen into the atmosphere. Other sources include the emission of ammonia from farm animal units and nitrogen oxides emissions linked to intensive fertiliser use. These pollutants are transported across the UK in the atmosphere, affecting rain water chemistry and air quality. This has resulted in acidification of soils and waters in acid-sensitive areas subject to high rainfall such as parts of central and southwest Scotland, Cumbria, the Pennines, Wales and the Mourne Mountains in Northern Ireland . Reductions in emissions due to policy control measures are achieving a decline in sulphur and nitrogen deposition but, unfortunately, due to increases in emissions from shipping, recovery has not been as fast as hoped for. Ammonia emissions increased sharply from the 1950s to 2000, and currently remain at these peak levels.



Nitrogen (NOx + NHx) deposition measured 2001-2003

Sulphur deposition measured 2001-2003

Total acid deposition (S + NOx + NHx) measured 1999-2001

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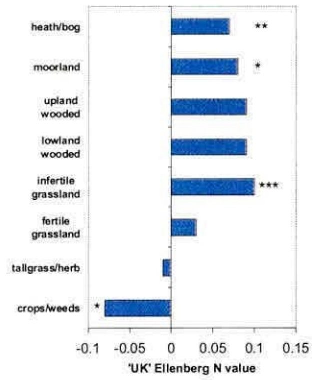
Evidence of acidification and N-enrichment effects on semi-natural habitats

There are various sources of information which indicate vegetation, soils and waters have been affected by sulphur and nitrogen deposition. A review of the evidence for the UK was undertaken by the National Expert Group on Transboundary Air Pollution (http://www.nbu.ac.uk/negtap/home.html). The evidence for nitrogen enrichment of vegetation includes two national monitoring programmes – the Countryside Survey and the New Plant Atlas for the UK – which identified shifts in species composition towards more nutrient-demanding species in the latter half of the 20th century (Preston *et al.* 2002, Haines-Young *et al.* 2003) (e.g. Figure 1).



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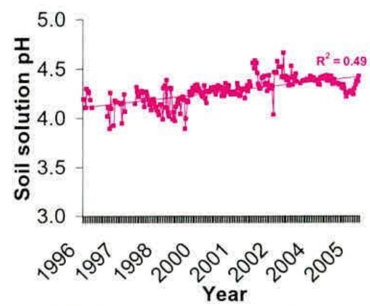

Although there is limited evidence of biological recovery from past emissions, there is some evidence of chemical recovery in soils and waters. A decline in acidity has been reported in some regions, perhaps reflecting the success of emission control policies to reduce levels of acid deposition in the environment. There is an expectation that, in time, this improvement will be reflected in biological recovery.

How does air pollution affect our waters, soils and vegetation?

Although rainfall is naturally acidic, the additional acidity introduced by sulphur dioxide and the oxides of nitrogen has affected waters, soils and vegetation in the UK. The pH of lakes and rivers fell, in turn affecting populations of fish, invertebrates and aquatic plant communities. Soils also became more acidic, affecting organic matter decomposition rates and soil nutrient balance. Soil acidification also increases the solubility of some elements such aluminium in the soil solution, which can be toxic to the fine roots of trees at high concentrations. Pollutants are also deposited to vegetation directly as gases, aerosols and in fogs and mists. Deposition rates are higher to forests than other land covers because they are taller, generally have a larger leaf area and are aerodynamically 'rougher'. The effects of acid deposition, particularly on freshwaters, have been greater in forests, not because the trees are acidifying in their own right, but because they are simply more efficient at removing pollutants from the atmosphere. This 'scavenging' effect is an important issue in parts of Scotland, for example, where forestry is identified as a specific pressure on water quality.

Excess nitrogen deposition can also be damaging in its own right due to its contribution to nutrient enrichment or eutrophication. Although excess nitrogen deposition generally promotes tree growth, it can also have negative effects, including nutrient imbalances, increased susceptibility to frost damage and higher levels of insect damage. However, the area of greatest concern is its impact on the wider plant community of woodland ecosystems; competition between species is affected, with the result that nitrogen demanding species such as nettle and bramble dominate at the expense of other herbaceous species that contribute so much to the character of our woodlands. Populations of sensitive lichen and moss species are also at risk.

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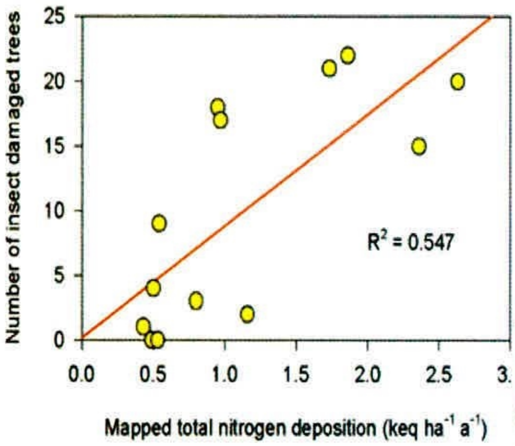
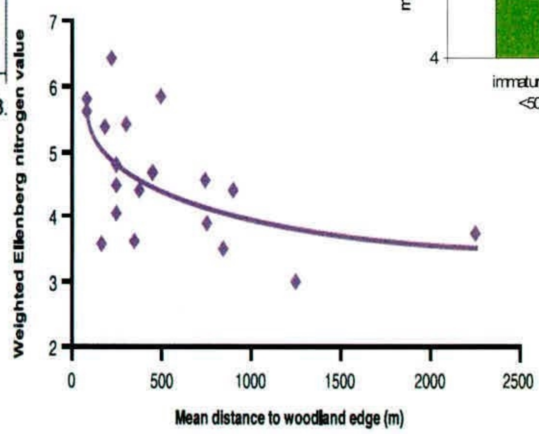
Current evidence for air pollution effects in British woodland

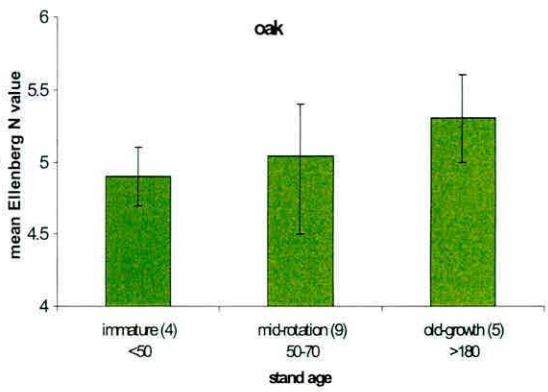
Long-term monitoring of forests in the UK indicates that there is no widespread damage to forests and trees as a result of atmospheric pollution. The network of 350 Forest Condition Survey Plots, on which diameter increment and crown condition have been measured since 1987, show no link between soil acidification and tree vitality. However, there is a possibility that soil function is being affected, leading to more subtle effects. For example, the Forest Condition Survey has identified a link between nitrogen deposition and the level of insect damage to Scots pine. An in-depth analysis of the ground flora associated with beech woodlands across the network has also revealed a relationship between the distribution of nitrogen demanding species and distance to woodland edge - a clear signal of the effects of nitrogen deposition. This observation is supported by recent surveys of plant communities, including Countryside Survey 2000, the New Plant Atlas and a re-analysis of ecological condition of over 100 semi-natural woodlands first surveyed in1971 (Kirby *et al.*, 2005), as well as by studies of the effects of points sources of nitrogen pollution (Pitcairn *et al.*, 2002). However, there are caveats that should be applied to all of these surveys when interpreting observed trends as a clear signal of the effects of nitrogen deposition. First, the level of woodland management has declined over the past two to three decades as a result of the poor economic climate of the forestry sector, reducing light levels on the forest floor thus favouring shade-tolerant species. Second, the woodlands have aged, which similarly affects the composition of the ground flora.



Relationship between observed number of insect damaged trees and mapped total nitrogen deposition for Forest Condition Survey Scots pine plots. (NEGTAP, 2001).



Changes in the nitrogen demand of beech woodland ground flora with proximity to the woodland edge. (Kennedy, 2003).



Variation in nitrogen demand of ground flora of oak woodland with stand age. (Pitman, 2006).

Trend analysis of soil solution chemistry from the Intensive Forest Monitoring Network has indicated some recovery, over the past ten years, from high historical pollution loading as a result of emission control policies. The most dramatic observation is the downward trend in tree foliar sulphur concentrations, corresponding to a reduction in soil solution sulphate concentrations across the network. Indeed, it is likely that sulphur deficiency may become an issue in the near future in some regions. There is little evidence to indicate a reduction in nitrogen deposition or its impacts; to the contrary, there is evidence of the effects of high nitrogen deposition, particularly in areas dominated by intensive agriculture, such as East Anglia. At one site in Thetford forest, where nitrogen deposition in throughfall has been measured as 25 kg ha^{-1} yr^{-1}, soil solution pH has fallen by 2 units since monitoring began in 1995 and nitrate concentrations in soil solution as high as 150 mg l^{-1} have been measured. Nettle, a species with high nitrogen demand, has also increased its dominance of the ground flora over this period.

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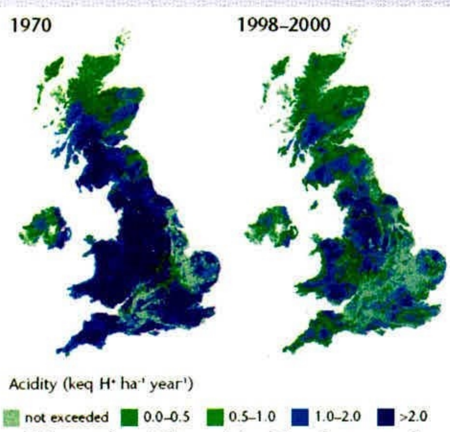
How will climate change and woodland affect the impacts of air pollution?

The predicted changes to the climate of the UK are likely to have significant impacts on woodland plant communities. Limited evidence suggests that rising CO_2 levels may favour ruderal species, at the expense of slower growing species, possibly compounding the impacts of nitrogen deposition. Where water is not limiting, the combined effects of nitrogen deposition and rising CO_2 levels are likely to lead to significant increases in forest growth. There is already some evidence for this in old-growth oak in southern England. Climate change will also affect soil processes, leading to interactions between growth, nutrient cycling and pollutant inputs. For example, mineralisation rates will increase thus affecting nutrient availability, while nutrient uptake and leaching may be affected by changing rainfall patterns and a longer growing season.

Conventional forest management increases the input of nitrogen that a forest ecosystem can withstand, by removing a proportion in timber. At the same time, base cations are removed reducing the ability of the ecosystem to withstand acid deposition, particularly woodlands established on base cation-poor soils. The future level of woodland management, particularly in response to an increased utilisation of woodfuel for climate change mitigation objectives, will affect how woodlands respond to the continuing effects of air pollution and it will be important to ensure that this interaction is accommodated in developing new climate change policy and forest management practices.

UK actions being taken to help reduce air pollution



Mapped acidity critical loads exceedance for woodland in 1970 and 2000

Critical loads methodologies have been developed to help policy makers target effective emissions control policies. A critical load is defined as the level of pollution input below which damage to the tree, soil or ecosystem is thought unlikely. Where a critical load is exceeded, it does not always mean that damage will occur immediately. In the UK, 98% of woodlands currently exceed the critical load for nutrient nitrogen and 75% for acidity. It should, however, be understood that critical loads are set to protect the most sensitive element of an ecosystem in the long-term; this extent of exceedance does not mean that the trees are at immediate risk and that any effects may take many decades to become apparent. Indications based on future emissions projections are that the extent of exceedance will fall for acidity, but that nitrogen will continue to be a significant threat.

Further Information

•NEGTAP (National Expert Group on Transboundary Air Pollution) 2001: Transboundary Air Pollution: Acidification, Eutrophication and Ground-Level Ozone in the UK. ISBN 1 870393 61 9. Available online at http://www.nbu.ac.uk/negtap/

•Countryside Survey 2000 - http://www.cs2000.org.uk/

•New Atlas of the British and Irish Flora, (2002). *Edited by C. D. Preston, D. A. Pearman and T. D. Dines.* ISBN: 0198510675

•Air Pollution Information System-- http://www.apis.ac.uk/

Key references

•Kirby et al. (2005). Long-term ecological change in British woodland. EN Research Reports 653. http://www.englishnature.gov.uk/pubs/publication/PDF/653.pdf

•Kennedy (2002). How extensive are the effects of nitrogen pollution in Great Britain's Forests? Forest Research Annual Report and Accounts 2001-2. http://www.forestresearch.gov.uk/pdf/FR0102nitro.pdf/$FILE/FR0102nitro.pdf

•Bradley et al.(2005). The impact of climate change on soil functions. Final report SP0538 to Defra.

•Broadmeadow M (2004). Woodland and our changing environment. Sustainable Forestry Briefing Note, Forestry Commission, Edinburgh. http://www.forestry.gov.uk/pdf/fcib001.pdf/$file/fcib001.pdf

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check as it was complicated.
Not got a pdf yet so
will let you know later

Jackie

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