Direct and indirect effects of nitrogen deposition on species composition change in calcareous grasslands

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Abstract

Atmospheric nitrogen (N) deposition has been identified as a major threat to biodiversity, but field surveys of its effects have rarely focussed on sites which are actively managed to maintain characteristic species. We analysed permanent quadrat data from 106 plots in nature reserves on calcareous grassland sites in the United Kingdom collected during a survey between 1990 and 1993 and compared the data with the results from resurvey of 48 of these plots between 2006 and 2009. N deposition showed no significant spatial association with species richness, species diversity, or the frequency of species adapted to low nutrient conditions in the 1990–1993 dataset. However, temporal analysis showed that N deposition was significantly associated with changes in Shannon diversity and evenness. In plots with high rates of N deposition, there was a decrease in species diversity and evenness, a decline in the frequency of characteristic calcareous grassland species, and a lower number of rare and scarce species. As all sites had active management to maintain a high diversity and characteristic species, our results imply that even focussed management on nature conservation objectives cannot prevent adverse effects of high rates of N deposition. Structural equation modelling was used to compare different causal mechanisms to explain the observed effects. For change in Shannon diversity, direct effects of N deposition were the dominant mechanism and there was an independent effect of change in grazing intensity. In contrast, for change in herb species number, indirect effects on soil acidity, linked to both N and S deposition, were more important than direct effects of N deposition.

Keywords: Festuco brometalia grasslands, management, nitrogen deposition, soil acidity, species diversity

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Introduction

Atmospheric deposition of nitrogen (N) and sulphur (S) compounds increased dramatically in Europe in the second half of the 20th century, mainly as a result of increased fossil fuel combustion and intensive agriculture (Galloway, 1995). Despite a strong decline in S emissions and deposition in most parts of Europe since the 1990s (Vestreng et al., 2007), N deposition has not declined to the same extent. Both N and S deposition have continued to increase in other parts of the world, especially in parts of India and China (Galloway & Cowling, 2002; Dentener et al., 2006).

N deposition can result in severe eutrophication via the direct input of nutrients but also acidification via indirect processes such as nitrification (Galloway, 1995; Bobbink et al., 1998). N deposition has been identified as one of the most important global drivers of biodiversity change in terrestrial ecosystems (Vitousek et al., 1997; Sala et al., 2000; Phoenix et al., 2006), and has been linked to changes in plant species composition in a wide range of terrestrial biomes from the Arctic to the tropics (e.g. Bobbink et al., 2010). Some of the most affected terrestrial habitats are nutrient-poor seminatural habitats such as heathlands and grasslands. Calcareous grasslands are floristically among the most species-rich communities of North-Western Europe, containing many rare and endangered plant species (e.g. Willems, 1990). Their biodiversity is threatened by changes in management (e.g. Tansley, 1922; Willems, 1978), habitat fragmentation (e.g. Fischer & Stöcklin, 1997), and disturbance (e.g. Smart et al., 2003a), among other factors.
Experimental studies in Dutch calcareous grasslands have shown that increased N deposition results in the loss of characteristic plant species and an increase of highly competitive grass species such as *Brachypodium pinnatum* (e.g. Willems, 1990; Bobbink et al., 1998). In the United Kingdom, changes in species composition, a reduction in flowering, and a reduction in the forb/grass ratio have been shown in a long-term N-addition experiment (Carroll et al., 2003), but encroachment by *B. pinnatum* or other highly competitive grasses such as *Bromus erectus* was not observed, possibly due to colimitation by phosphorus (Morecroft et al., 1994; Wilson et al., 1995; Carroll et al., 2003; Phoenix et al., 2003).

Stevens et al. (2004) found a significant negative spatial relationship between N deposition and species richness in a national survey of one acidic grassland community in the United Kingdom. More recently, analysis of data from a national survey covering all plant communities in the United Kingdom corroborated these findings for both heathlands and acid grasslands, but failed to find a significant relationship between N deposition and species richness in calcareous grasslands, even when interacting environmental factors such as rainfall and altitude were included in the models (Maskell et al., 2010). However, significantly more species characteristic of fertile soil conditions, and a higher grass/forb ratio, were found in calcareous grassland plots with higher N deposition. These findings suggest that the responses of calcareous grasslands to increased N deposition differ from those of other seminatural ecosystems, such as acid grasslands and heathlands. This may be due to several factors, including (a) the importance of colimitation by phosphorus, (b) active management and conservation to protect characteristic plant species, and (c) the occurrence on shallow well-buffered soils that are less sensitive to acidification caused by N and S deposition.

In addition, the limitations of the studies undertaken to date need to be considered. Experimental studies measure change over a relatively short period of time with N deposition as the only variable factor. Field surveys which explore the spatial associations between N deposition and species richness, or other response variables, do not allow an assessment of whether these are due to long-term historical deposition or recent changes in deposition, such as those simulated in experimental studies. Although long-term vegetation change in calcareous grasslands in the United Kingdom since 1950 has been described by Bennie et al. (2006), their analysis was not designed to assess the effect of N deposition, and there are no field surveys relating recent changes in species composition or species numbers to atmospheric N deposition, using plots that can be relocated with confidence. A further limitation to the interpretation of field survey data is the multicollinearity between predictor variables such as N deposition and climate variables, and earlier studies have highlighted the limitations that multiple regression or covariance analysis have in predicting the relative effects of the explanatory variables on dependent variables such as species richness (e.g. Stevens et al., 2004; Maskell et al., 2010).

We investigated the direct, indirect, and interacting effects of atmospheric N deposition, other confounding environmental variables, and site management on species richness and species diversity in calcareous grasslands in the United Kingdom. We based our study on a network of 128 permanently marked plots in areas of calcareous grassland spread across the United Kingdom that were specifically established in the early 1990s, with the aim of monitoring long-term impacts of climate change and air pollution, by John Rodwell and colleagues (Rich et al., 1993). Between 2006 and 2009, 48 plots were revisited and additional soil measurements were made. This allowed us to assess the effect of N deposition and other factors on both spatial variation and recent temporal changes in species richness and other relevant variables.

In order to test if changes in species richness and composition are significantly affected by atmospheric N deposition in a matrix of effects of factors such as management, climate, S deposition, and soil conditions, we analysed the data using both mixed effect models and structural equation modelling (SEM). The SEM models allow explanatory variables to be placed in ‘series’ rather than ‘parallel’, using sequential arrangements that reflect hypothetical causal chains linking the effects. Such models provide insights that are not possible with standard covariance analyses into the relative importance of different direct and indirect effects of N deposition.

We tested three major hypotheses:

1. N deposition is a significant explanatory driver of spatial variation in species richness, species diversity, and the frequency of individual species in calcareous grasslands.
2. N deposition is a significant explanatory driver of change over the last 15 years in species richness, species diversity, and the frequency of individual species in calcareous grasslands.
3. Recent changes in species richness, species diversity, and the frequency of individual species in calcareous grasslands can be explained by a combination of the effects of management, direct effects of N deposition, and indirect effects of S and N deposition via soil acidification.
Materials and methods

Calcareous grassland survey: 1990–1993 survey

A large national survey of calcareous grasslands was carried out in the United Kingdom between 1990 and 1993 (Rich et al., 1993). The survey method was specifically designed to be repeatable with a minimum of observer bias. All the selected sites were nature reserves with active management to maintain species diversity and characteristic species; 56 sites were selected, with one to six plots on each site, resulting in a total of 128 plots. Sites were chosen to cover a wide geographical range (Fig. 1), a large climatic gradient, a wide range in deposition of air pollutants and all types of calcareous grasslands known in the United Kingdom, using the National Vegetation Classification (NVC) (Rodwell, 1998). On most sites, plots were chosen on both south- and north-facing slopes to account for effects of aspect. Each plot covered an area of 144 m² (typically measuring 12 × 12 m or more rarely 6 × 24 m) and was established in a floristically homogeneous stand of calcareous grassland. Individual plots were permanently marked at unique places with buried copper coils. Detailed field notes, describing the exact location of the plots, photographs, sketch maps and measurements from fixed features were also recorded, to ensure successful relocation of each plot in future surveys. The presence/absence of all vascular plants was recorded in thirty-six 50 × 50 cm quadrats positioned within each permanent plot. Soil pH of the top 10 cm was determined in the field by mixing soil into a slurry with distilled water (1:1 vol:vol), equilibrating for 5 min, and measuring using a calibrated pH meter. Soil type, soil depth, slope, aspect, and sward height were recorded for each plot. For each site, the size, its protection status and detailed information on local management (grazing intensity) were noted. Data from the original reports could not be recovered for seven plots (four sites), resulting in available data for 121 plots.

2006–2009 survey

In 2006, 35 out of the 56 sites were selected for resurvey (48 plots), covering a wide range of N deposition. Sites were selected to reduce the effects of spatial autocorrelation in atmospheric N deposition (e.g. by selecting high and low N deposition sites at similar latitudes). Sites and plots with NVC communities CG1, CG2, CG10, CG11, and CG12 were targeted where possible to reduce variation due to initial vegetation composition. Only six of the plots selected for resurvey were identified as supporting different NVC communities (CG3, CG8, CG9), mainly due to the presence of grasses such as B. erectus (CG3) or Sesleria albicans (CG8 and 9). In broad terms, plots in the lowlands of Wales and England focussed on stands of CG1 and CG2, while those in the uplands of Wales and Scotland focussed on stands of CG10, CG11, or CG12. There was no significant correlation between the different vegetation types and total N deposition ($r = 0.23$, $P = 0.14$).

Metal detectors were used for relocation of the plots. In each plot, 36 quadrats were positioned in exactly the same pattern as in 1990–1993 and in each quadrat the presence/absence of all vascular plants was recorded. One soil sample, composed of five subsamples, was taken from the top 10 cm of each plot using an auger (3 cm diameter). Soil samples were stored at 4 °C until further analysis. Sward height and soil depth were recorded at each plot and detailed information on the local management [specifically, grazing intensity in livestock units (LU) per hectare per year] for the past 15 years was obtained from local managers at the plot level, for comparison with 1990–1993.

Soil analysis

Soil samples from the 2006 to 2009 survey were thoroughly mixed before extraction. Extractions were performed on fresh soil and corrected for moisture content after drying for 24 h at 105 °C. A portion of 35 g of fresh soil was mixed with 100 mL bi-distilled water for water-soluble extraction or 100 mL 0.2 M NaCl for the exchangeable fraction. Extracts were shaken for 1 h at 100 rpm after which the pH was measured in the water fraction with a Radiometer type PHM 82 pH meter. Soil pH was not measured in the field as in 1990–1993. Extracts were filtered (0.2 μm) and stored at 4 °C until analysis. Plant available P was measured after Olsen-P extraction (Olsen et al.,...
and stored at 4°C until analysis. NO$_3^-$ and NH$_4^+$ were analysed on both extracts using a Bran and Luebbe Autoanlyser 3 (Bran and Luebbe, Norderstedt, Germany), but only results from the NaCl extracts are discussed in this paper as this represents the plant available fraction. Concentrations of base cations (Ca$^{2+}$, Na$^+$, Mg$^{2+}$, and K$^+$) and Al$^{3+}$ and plant available P were determined in the NaCl and Olsen extracts, respectively, using an ICP Spectrometer (IRIS Intrepid II, Thermo Electron Corporation, Franklin, MA, USA). Organic matter contents of the soils were determined by loss on ignition (4 h, 550°C).

**Plant attributes**

The names of all plant species in the 1990–1993 survey were checked and if necessary updated to enable a comparison with the recent 2006–2009 survey. All data were then linked to the attribute list PLANTATT (Hill et al., 2004). Stace (1997) was followed for nomenclature. We used Ellenberg indicator values modified for the United Kingdom (Hill et al., 2004) to characterize plant species. For each plot the average, frequency weighted, Ellenberg value was calculated in order to characterize plots according to the species they contained. Species were grouped into eutrophic (Ellenberg N $>$ 5), oligotrophic (Ellenberg N $<$ 3), acidophilic (Ellenberg R $>$ 5), and acidophillic (Ellenberg R $<$ 3) species. Significant responses to N deposition were not expected for intermediate species (Ellenberg R = 1, 2, 3, 4, and 5); this was confirmed by our statistical analyses (results not shown).

The total number of scarce and rare species, as denoted in the PLANTATT database, was recorded for each plot. Species richness (S) was recorded as the total number of vascular plant species in each plot. For each plot, the species diversity was calculated as the Shannon index ($H'$): $H' = - \sum (p_i \ln p_i)$, where $p_i$ is the number of occurrences of the $i$th species in the 36 quadrats expressed as a proportion of the total number of occurrences of all species in those quadrats (Magurran, 2004). In addition, Shannon’s equitability or the Evenness ($E_{99}$) was calculated as $E_{99} = H'/\ln (S)$. For each species the frequency of occurrence was calculated as $n_i/36$, where $n_i$ is the number of quadrats in which the species occurred in plot $i$.

**Survey data selection**

In the soils of the plots revisited in 2006–2009, pH varied widely, ranging from shallow rendzinas of pH 8.5 to peaty soils overlaying calcareous rock of pH 4.2. The six plots with the lowest pH had in general very different vegetation (e.g. more acidophilic species such as Calluna vulgaris or Juncus effusus), a very high soil organic matter and low base cation content. In order to relate atmospheric N deposition and other environmental parameters to species composition and richness in calcareous grasslands, which share similar characteristics and to reduce confounding effects, only those plots with a soil pH ($pH_{1990–1993}$) $>$ 6.0 were included in the analysis. This resulted in 106 plots from the 1990 to 1993 data and 42 plots which could be used for the comparison between 1990–1993 and 2006–2009. For analysis at the individual species level, changes in species frequency were analysed on a subset of the data; only those species which occurred in at least 11 (25%) or more of the 42 plots in both surveys were included. This resulted in a dataset for only 52 species. Grass to herb ratio was calculated as the ratio between the numbers of grass and herb species at the plot level.

**Plot and site-specific parameters**

Modeled N and S deposition data for each site were obtained from the Centre of Ecology and Hydrology (CEH) for two periods: 1993–1995 and 2006–2008 for the surveys of 1990–1993 and 2006–2009, respectively. The N deposition ranged from 7.3 to 40.7 kg N ha$^{-1}$ yr$^{-1}$ whereas S deposition ranged from 8.0 to 27.6 kg S ha$^{-1}$ yr$^{-1}$. Climate data for each site were obtained from the 5 km gridded climate database held by the UK Meteorological Office (http://www.metoffice.co.uk). Ten-year averaged data for the periods 1981–1991 and 1997–2006, respectively, were used for the 1990–1993 and 2006–2009 surveys, including mean minimum winter temperature, mean maximum summer temperature, mean annual temperature, and mean annual rainfall. The aspect of each plot was converted to a 0–1 scale using sin($\pi a/(360/\pi)$), where $a$ is the aspect in degrees. This resulted in a gradient from North = 0 to South = 1, with East and West both at 0.5. Where information on stocking was available from local site managers, grazing intensity for each plot was calculated in terms of LU per hectare per year using the conversion factors provided in Attwood & Heavey (1964). For the nine plots where detailed stocking was not provided, we estimated the grazing intensity from the sward height, based on the significant correlation ($r^2$ = 0.58) between sward height and LU obtained from the other plots.

**Statistical analysis**

Principal component analysis (PCA) in CANOCO (Ter Braak, 1988) was used to determine the variables explaining most of the variation between the plots. For each survey, we selected the strongest, noncorrelated, variables for further analysis.

Possible combinations of predictor variables were explored to assess the effects of N deposition, soil conditions, climate data and grazing pressure on species richness, and species composition. Data from the 1990 to 1993 survey were used in the spatial analysis, while data from both surveys were used in the temporal analysis. All models were explicitly tested for spatial autocorrelation in the response variable and residuals by inspection of semivariograms. In the presence of spatial autocorrelation, generalized linear mixed-effect models were used to control for this spatial autocorrelation. In these models, a correlation structure was added to correct for spatial auto-correlation. Correlation structures such as corExp and corSpher were used with the ‘form = ~ Easting + Northing’ argument in the correlation option to calculate the Euclidean distances (using Pythagoras theorem) between sites with coordinates given by Easting and Northing. Predictor variables

were omitted after ranking the models according to the Akaike Information Criterion (AIC; Sakamoto et al., 1986). Only the models and corresponding correlation structure with the lowest AIC, representing the 'best fitting models,' were selected, allowing nonsignificant variables to enter the models if they improved the fit. When no spatial autocorrelation was observed, multiple linear regression models were used. Again, models were ranked according to AIC and only the models with the lowest AIC were selected. To test for multicollinearity, variance inflation factors (VIF) were computed for each predictor variable in the model. Predictor variables that were highly correlated (VIF > 10; Gujarati, 1995) were not analysed in the same model; in those situations only the predictor variable that explained most of the variance was selected.

The ecological significance of the change in Shannon diversity $H'$, was quantified after dividing the data into three groups: one with N deposition below the lowest value, one with N deposition within the range, and one with N deposition above the highest value, of the critical load range of N deposition for this vegetation type (<15 and >25 kg N ha$^{-1}$ yr$^{-1}$, respectively, Bobbink et al., 2003). This division was partly supported by a regression tree model of change in $H'$, which showed that the first split in the data was at 25.3 kg N ha$^{-1}$ yr$^{-1}$, very similar to the higher critical load. The second split in the tree model was found at a level of 10.1 kg N ha$^{-1}$ yr$^{-1}$, below the lower critical load value. However, as the number of plots below 10 kg N ha$^{-1}$ yr$^{-1}$ was limited, a value of 15 kg N ha$^{-1}$ yr$^{-1}$, corresponding to the lower end of the empirical critical load range, was adopted.

The Cohen $d$-value was then calculated according to:

$$d = \frac{\bar{x}_1 - \bar{x}_2}{\sqrt{\frac{s_1^2 + s_2^2}{2}}}.$$

where $\bar{x}_1$ and $\bar{x}_2$ are the means of the data for sites with high and low N deposition, respectively, and $s_1$ and $s_2$ are the standard deviations in these groups. The calculated value of $d$ was checked against Cohen's classification in which $d > 0.8$ implies a large effect (Cohen, 1988). We also compared the predicted change in $H'$ from our models at sites with N deposition above 25 kg ha$^{-1}$ yr$^{-1}$ with the value of $H'$ in a reference dataset representing an highly undesirable next step in trophic status that could be realized given continued high N deposition when grazing and/or cutting is maintained. We located plots of the Countryside Survey (CS) of Great Britain which independent land cover mapping showed could be referenced to the Neutral Grassland Broad Habitat (Jackson, 2000). The change in $H'$ from 1990–1993 to 2006–2009 in the high N deposition plots was then expressed as a percentage of the difference between the 1990 and 1993 value and that of the reference dataset, an approach similar to that used by Smart et al. (2003b). The analyses were done in the ‘M’ (version 2.9.0) statistical and programming environment (R Development Core Team, 2008).

SEM(Byrne, 2010) were used to further examine the relationships between N deposition, soil conditions, climate data and grazing pressure on species richness, and species composition. The software package AMOS 17 was used to design the SEM models and calculate path coefficients, squared multiple correlations, direct and indirect effects and model fit. A significant $\chi^2$-statistic indicates that the SEM model provides a poor fit. $\chi^2$-statistics and four non-$\chi^2$-distributed fitting functions (CFI, Comparative Fit Index; NFI, Normed Fit Index; TLI, Tucker-Lewis Index; IFI, Incremental Fit Index) were compared in order to test the goodness-of-fit of the models. In the SEM models, soil nutrient concentrations were represented by the sum of the NaCl extractable NH$_4^+$ and NO$_3^-$ concentrations, subsequently referred to as 'mineral N.'

Results

Analysis of 1990–1993 survey

Maximum summer temperature and soil depth were the most important variables explaining variation in species composition and species richness in the 1990–1993 dataset (Table 1a). Spatial variation in species richness, the number of eutrophic species, the number of herb species, and $H'$ could be explained solely by soil depth and maximum summer temperature, with positive relationships with both variables in all cases.

Linear regression between the total number of vascular plant species and N deposition did not reveal a significant relationship in the 1990–1993 dataset (Fig. 2a), but N deposition did show a near significant positive relationship with the grass to herb ratio (Fig. 2b). Almost no scarce and rare species were observed at plots with N deposition above 25 kg N ha$^{-1}$ yr$^{-1}$ (Fig. 2c); the average number of scarce and rare species was 1.6 in the 77 plots with deposition below 25 kg N ha$^{-1}$ yr$^{-1}$ compared with 0.3 in the 29 plots with deposition above 25 kg N ha$^{-1}$ yr$^{-1}$ (ANOVA, $df = 1$, $F = 13.62$, $P < 0.001$).

Minimum winter temperature and maximum summer temperature were highly correlated; only maximum summer temperature was included in the model, as this was the stronger predictor according to the PCA. For similar reasons, organic content of the soil, Olsen P, and slope (which were all strongly correlated with pH) and mean annual rainfall (which was strongly correlated with maximum summer temperature) were excluded from the analysis. Grazing pressure entered the analysis, but was not a significant explanatory variable for any of the dependent parameters (Table 1a).

N deposition did not have a significant relationship with species richness or with the number of eutrophic or oligotrophic species. The frequency weighted average Ellenberg N-value was also not significantly related to N deposition, but was best explained by the maximum summer temperature. The only variable for which N deposition entered the final model was the grass to herb ratio, for which a positive effect, albeit at
### Table 1  Effect sizes of the predictor variables (expressed as the coefficient ± SD), for the subset of predictors that were included in the model with the lowest AIC for each response variable, for (a) the survey in 1990–1993 and (b) the change between the surveys in 2006–2009 and 1990–1993

(a) 1990–1993

<table>
<thead>
<tr>
<th>Predictor variable</th>
<th>pH</th>
<th>Soil depth</th>
<th>Aspect</th>
<th>N deposition</th>
<th>Max temp summer</th>
<th>Grazing</th>
<th>df&lt;sub&gt;1,2&lt;/sub&gt;</th>
<th>AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Species richness†</td>
<td>0.338 ± 0.121*</td>
<td>0.274 ± 0.093***</td>
<td>1.442 ± 0.616*</td>
<td>1.530 ± 0.467***</td>
<td>1.715 ± 1.218</td>
<td>1, 103</td>
<td>770</td>
<td></td>
</tr>
<tr>
<td>Eutrophic species†</td>
<td>0.274 ± 0.093***</td>
<td>1.000 ± 0.000**</td>
<td>0.121*</td>
<td>1.442 ± 0.616*</td>
<td>1.530 ± 0.467***</td>
<td>1, 103</td>
<td>715</td>
<td></td>
</tr>
<tr>
<td>Oligotrophic species</td>
<td>-0.507 ± 0.123***</td>
<td>1.000 ± 0.000**</td>
<td>1.000 ± 0.000**</td>
<td>1.000 ± 0.000**</td>
<td>1.000 ± 0.000**</td>
<td>1, 103</td>
<td>715</td>
<td></td>
</tr>
<tr>
<td>Acidophilic species</td>
<td>1.715 ± 1.218</td>
<td>1.442 ± 0.616*</td>
<td>1.530 ± 0.467***</td>
<td>1.000 ± 0.000**</td>
<td>0.121*</td>
<td>1, 103</td>
<td>715</td>
<td></td>
</tr>
<tr>
<td>Acidophobic species</td>
<td>0.266 ± 0.097**</td>
<td>0.121*</td>
<td>1.442 ± 0.616*</td>
<td>1.530 ± 0.467***</td>
<td>1.000 ± 0.000**</td>
<td>1, 103</td>
<td>715</td>
<td></td>
</tr>
<tr>
<td>Herb species†</td>
<td>0.001 ± 0.000***</td>
<td>0.001 ± 0.000***</td>
<td>0.001 ± 0.000***</td>
<td>0.001 ± 0.000***</td>
<td>0.001 ± 0.000***</td>
<td>1, 103</td>
<td>715</td>
<td></td>
</tr>
<tr>
<td>Grass : herb ratio†</td>
<td>0.266 ± 0.097**</td>
<td>0.121*</td>
<td>1.442 ± 0.616*</td>
<td>1.530 ± 0.467***</td>
<td>1.000 ± 0.000**</td>
<td>1, 103</td>
<td>715</td>
<td></td>
</tr>
<tr>
<td>WA Ellenberg N*</td>
<td>0.028 ± 0.009**</td>
<td>0.121*</td>
<td>1.442 ± 0.616*</td>
<td>1.530 ± 0.467***</td>
<td>1.000 ± 0.000**</td>
<td>1, 103</td>
<td>715</td>
<td></td>
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<tr>
<td>WA Ellenberg R*</td>
<td>0.213 ± 0.073***</td>
<td>0.121*</td>
<td>1.442 ± 0.616*</td>
<td>1.530 ± 0.467***</td>
<td>1.000 ± 0.000**</td>
<td>1, 103</td>
<td>715</td>
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<tr>
<td>Shannon index</td>
<td>0.0133 ± 0.004***</td>
<td>0.001 ± 0.000***</td>
<td>0.001 ± 0.000***</td>
<td>0.001 ± 0.000***</td>
<td>0.001 ± 0.000***</td>
<td>1, 103</td>
<td>715</td>
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<tr>
<td>Evenness†</td>
<td>0.001 ± 0.000***</td>
<td>0.001 ± 0.000***</td>
<td>0.001 ± 0.000***</td>
<td>0.001 ± 0.000***</td>
<td>0.001 ± 0.000***</td>
<td>1, 103</td>
<td>715</td>
<td></td>
</tr>
</tbody>
</table>

(b) Change between 2006–2009 and 1990–1993

<table>
<thead>
<tr>
<th>Predictor variable</th>
<th>pH</th>
<th>Soil depth</th>
<th>Mineral N soil</th>
<th>Aspect</th>
<th>N deposition</th>
<th>Max temp summer</th>
<th>Grazing</th>
<th>df&lt;sub&gt;1,2&lt;/sub&gt;</th>
<th>AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Species richness</td>
<td>2.634 ± 1.242*</td>
<td>-0.106 ± 0.056</td>
<td>3.773E-4 ± 2.003E-4*</td>
<td>1.442 ± 0.616*</td>
<td>1.530 ± 0.467***</td>
<td>1, 35</td>
<td>227</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Eutrophic species</td>
<td>0.274 ± 0.093***</td>
<td>3.773E-4 ± 2.003E-4*</td>
<td>1.000 ± 0.000**</td>
<td>1.442 ± 0.616*</td>
<td>1.530 ± 0.467***</td>
<td>1, 35</td>
<td>170</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Oligotrophic species</td>
<td>-0.106 ± 0.056</td>
<td>3.773E-4 ± 2.003E-4*</td>
<td>1.000 ± 0.000**</td>
<td>1.442 ± 0.616*</td>
<td>1.530 ± 0.467***</td>
<td>1, 35</td>
<td>170</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Acidophilic species</td>
<td>-0.032 ± 0.021</td>
<td>8.085E-04 ± 4.253E-04*</td>
<td>1.000 ± 0.000**</td>
<td>1.442 ± 0.616*</td>
<td>1.530 ± 0.467***</td>
<td>1, 35</td>
<td>170</td>
<td></td>
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</tr>
<tr>
<td>Acidophobic species</td>
<td>0.089 ± 0.077</td>
<td>8.085E-04 ± 4.253E-04*</td>
<td>1.000 ± 0.000**</td>
<td>1.442 ± 0.616*</td>
<td>1.530 ± 0.467***</td>
<td>1, 35</td>
<td>170</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Herb species</td>
<td>2.427 ± 0.9348*</td>
<td>-0.056 ± 0.021*</td>
<td>8.085E-04 ± 4.253E-04*</td>
<td>1.000 ± 0.000**</td>
<td>1.442 ± 0.616*</td>
<td>1, 35</td>
<td>203</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Grass : herb ratio†</td>
<td>2.427 ± 0.9348*</td>
<td>-0.056 ± 0.021*</td>
<td>8.085E-04 ± 4.253E-04*</td>
<td>1.000 ± 0.000**</td>
<td>1.442 ± 0.616*</td>
<td>1, 35</td>
<td>203</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Grass : N soil ratio</td>
<td>0.213 ± 0.073***</td>
<td>-0.056 ± 0.021*</td>
<td>8.085E-04 ± 4.253E-04*</td>
<td>1.000 ± 0.000**</td>
<td>1.442 ± 0.616*</td>
<td>1, 35</td>
<td>203</td>
<td></td>
<td></td>
</tr>
<tr>
<td>WA Ellenberg N*</td>
<td>0.028 ± 0.009**</td>
<td>-0.056 ± 0.021*</td>
<td>8.085E-04 ± 4.253E-04*</td>
<td>1.000 ± 0.000**</td>
<td>1.442 ± 0.616*</td>
<td>1, 35</td>
<td>203</td>
<td></td>
<td></td>
</tr>
<tr>
<td>WA Ellenberg R*</td>
<td>0.115 ± 0.018***</td>
<td>-0.056 ± 0.021*</td>
<td>8.085E-04 ± 4.253E-04*</td>
<td>1.000 ± 0.000**</td>
<td>1.442 ± 0.616*</td>
<td>1, 35</td>
<td>203</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Shannon index</td>
<td>0.0133 ± 0.004***</td>
<td>-0.056 ± 0.021*</td>
<td>8.085E-04 ± 4.253E-04*</td>
<td>1.000 ± 0.000**</td>
<td>1.442 ± 0.616*</td>
<td>1, 35</td>
<td>203</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Evenness†</td>
<td>0.001 ± 0.000***</td>
<td>-0.056 ± 0.021*</td>
<td>8.085E-04 ± 4.253E-04*</td>
<td>1.000 ± 0.000**</td>
<td>1.442 ± 0.616*</td>
<td>1, 35</td>
<td>203</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Predictors significant at $P<0.1$, $P<0.05$, $P<0.01$, and $P<0.001$ are indicated by *, **, *** and ****, respectively. Analyses were based on a maximum of 106 calcareous grassland surveys in 1993 and a maximum of 42 comparisons between 2009 and 1993. All analyses were based on vascular plants only. The number of acidophilic species was log($x + 1$) transformed.

df<sub>1,2</sub> is degrees of freedom of the tested effect term and the error term, respectively.

†Models tested with lme rather than lm function.

AIC, Akaike Information Criterion; N, nitrogen; WA, weighted average.
the $P < 0.1$ level, was recorded. A significant negative effect of soil pH was found on the number of acidophilic species, in a model that also included a negative effect of maximum summer temperature. The frequency-weighted Ellenberg $R$-values increased with soil pH, in a model that also included a positive effect of maximum summer temperature.

**Analysis of change between surveys**

A significant, but slight, decrease in species richness, independent of N deposition, was observed between 1990–1993 (average = 41) and 2006–2009 (average = 39) (paired $t$-test $t = -2.274$, $P = 0.028$, df = 41). The change in both $H'$ and $E_H$ over this period was significantly negatively related to N deposition, with a general increase in $H'$ and $E_H$ at low N deposition and a decrease at high N deposition (Fig. 3). N deposition was not significantly related to the change in any of the other dependent variables.

In strong contrast to the spatial analysis of the 1990–1993 data, the factors maximum summer temperature and aspect did not significantly explain variation in changes in any of the dependent variables (Table 1b). In contrast to the spatial models, soil depth entered the models for change in number of oligotrophic, acidophilic, and acidophobic species, but was never significant (Table 1b).

Soil pH showed a significant positive relationship with the change in the species richness, i.e. there tended to be an increase in species richness in plots with a higher soil pH. In addition, soil pH was associated positively with change in the number of herb species and negatively with change in grass to herb ratios. The mineral N content was also weakly positively associated with change in the number of herb species. However, the changes in the number of acidophilic species, the number of acidophobic species, and the cover-weighted Ellenberg $R$ index were not related significantly to soil pH or any other variable.

**Fig. 2** The relationship between total N deposition (kg ha$^{-1}$ yr$^{-1}$) in 1993–1995 and (a) the total number of vascular plant species in 1990–1993; (b) grass to herb ratio in 1990–1993; and (c) the total number of scarce and rare vascular plant species in 1990–1993. The equation of the fitted line in (b) is $y = 0.004x + 0.40$ ($r^2 = 0.03$; $P = 0.053$).

**Fig. 3** The relationship between the total N deposition (kg ha$^{-1}$ yr$^{-1}$) in 2006–2008 and the decrease between 1990–1993 and 2006–2009 of (a) the plot species diversity ($H'$); and (b) the plot species evenness. The equation of the fitted line in (a) is $y = -0.008x + 0.19$ ($r^2 = 0.15$; $P = 0.011$). The equation of the fitted line in (b) is $y = -0.002x + 0.03$ ($r^2 = 0.21$; $P = 0.002$).
The models showed a significant negative effect of N deposition on changes in $H'$ and evenness (Table 1b). However, N deposition was not significantly related to change in the number of eutrophic or oligotrophic species, or change in Ellenberg $N$ index. The change in grazing was significantly negatively related to the change in frequency-weighted Ellenberg $N$ index, whereas the change in the number of oligotrophic species showed a positive relationship to the mineral N content of the soil.

**SEM**

SEM were used to further investigate the direct and indirect effects of N deposition for two variables for which change between 1990–1993 and 2006–2009 was not spatially auto-correlated: $H'$ and number of herb species. Using maximum likelihood estimation, we observed a $\chi^2$ of 4.488 for the model for $H'$ (df = 7, $P = 0.722$) for a goodness-of-fit test of the null hypothesis that the covariance matrix implied by the (expected) model reproduces the observed covariance matrix. Failure to reject the null hypothesis indicates that the model was a good fit. In addition, the CFI of 1.00, the NFI of 0.94, the TLI of 1.14, and the IFI of 1.04 that we calculated all indicated a good model fit.

As expected from the linear models (Table 1b), the SEM model showed that N deposition is a significant explanatory parameter for the changes in $H'$ of the plots (Fig. 4a, Table 2a). A direct negative standardized path coefficient of $-0.302$ (Fig. 4a) on the change in $H'$ was observed. Standardized path coefficients represent the standardized change in a response variable per standardized unit change in a predictor variable, i.e. when N deposition increased by 1 SD, change in $H'$ decreased by 0.302 SD. In terms of ecological significance, this can be regarded as a moderate effect (Chin, 1998). In addition, there was a positive relationship between soil pH and increase in $H'$. Mineral N content exerted an indirect effect on the changes via soil pH, as well as a direct effect. N deposition was positively related to mineral N content. S deposition had no direct effect but a negative indirect effect via soil pH (Fig. 4a, Table 2a). Grazing pressure had an independent direct negative effect on change in $H'$.

The SEM model for herb species showed that N deposition only weakly explains the change between 1990–1993 and 2006–2009 (Fig. 4b, Table 2b) and that this was an indirect effect via soil pH. A much stronger negative effect of S deposition on the change in herb species was observed (Table 2b), also indirectly via soil pH. Grazing pressure did not have an effect and was eliminated from this model. Mineral N content exerted an indirect effect via soil pH (Fig. 4b, Table 2b).

**Change in frequency of individual species**

Some of the 52 individual species that met our selection criteria showed very distinct changes in frequency between the two surveys. Species which declined or increased in frequency over all plots by more than 15% are listed in Table 3, with the change in frequency split between the three N deposition categories. For the three species with an increase in frequency at the highest N deposition, the overall relationship with N deposition was not significant. However, the five species which declined in frequency in the highest class of N deposition (Galium sterneri, Linum catharticum, Briza media, Gentianella amarella, and Campanula rotundifolia) all showed a significant correlation between the decline in frequency and N deposition.

**Discussion**

A key finding from our study is that the effects of N deposition, and of other driving variables, are quite different when considering the spatial variation between sites in 1990–1993, and when considering the temporal change between 1990–1993 and 2006–2009. Here we consider the results from the spatial analysis, the analysis of temporal change, and the SEM modelling in turn (according to our three hypotheses), placing our findings into the wider context of the role of N deposition and other factors in influencing the species richness, composition, and diversity of calcareous grasslands.

**Spatial variation**

We found that N deposition did not explain spatial variation in species richness, diversity, or the frequency of eutrophic species. This is consistent with a study in 1998 of 94 calcareous grassland plots in the United Kingdom, in which there was no significant relationship between N deposition or climatic variables, and species richness (Maskell et al., 2010), even though significant relationships between N deposition and species richness were found in the same survey in acid grasslands and heathlands. These findings contrast with results from experimental studies in which clear N fertilization effects have been shown in calcicolous habitats, with fast-growing eutrophic species outcompeting slow-growing sensitive species (e.g. Bobbink, 1991; van den Berg et al., 2005). Maskell et al. (2010) argued that their relatively small sample size (94 plots), and the correlation between N deposition and climatic parameters, may have masked the significance of N deposition as an explanatory variable. Our sample size is similar (106 plots) to that of Maskell et al. (2010), but a more likely
explanation for the lack of significant relationships with N deposition is that all of our sites are nature reserves, at which the management is aimed specifically at the conservation of high species diversity and in which grazing regimes are implemented to prevent the dominance of eutrophic species. Grazing at moderate

![Diagram](a) Structural equation model depicting direct and indirect effects of N and S deposition, soil pH, mineral N soil and grazing on the change in Shannon diversity between 2006–2009 and 1990–1993. The width of each arrow is proportional to the standardized path coefficient (coefficients shown in italic). Positive path coefficients are indicated by solid lines and negative path coefficients are represented by dashed lines. *Effects that are significant at \( P < 0.1 \). Estimates of the proportion of total variance explained (squared multiple correlations) are shown in bold for each dependent variable. Variables and paths representing unmeasured residual variation are not indicated for simplicity. (b) Structural equation model depicting direct and indirect effects of N and S deposition, soil pH and soil mineral N soil on the change in herb species numbers between 2006–2009 and 1990–1993. The width of each arrow is proportional to the standardized path coefficient (coefficients shown in italic). Positive path coefficients are indicated by solid lines and negative path coefficients are represented by dashed lines. *Effects that are significant at \( P < 0.1 \). Estimates of the proportion of total variance explained (squared multiple correlations) are shown in bold for each dependent variable. Variables and paths representing unmeasured residual variation are not indicated for simplicity.

### Table 2

Summary of results from structural equation modelling, showing the decomposition of correlations into standardized direct, indirect and total effects of seven explanatory variables on (a) the difference in Shannon diversity index and (b) the difference in the number of herb species between 2006–2009 and 1990–1993

<table>
<thead>
<tr>
<th>(a)</th>
<th>Difference in Shannon diversity between 2009 and 1993 (H2009–H1993)</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Direct</td>
<td>Indirect</td>
</tr>
<tr>
<td>N deposition</td>
<td>−0.302</td>
<td>−0.008</td>
</tr>
<tr>
<td>SO(_x) deposition</td>
<td>−</td>
<td>−0.037</td>
</tr>
<tr>
<td>Mineral N soil</td>
<td>−0.022</td>
<td>−0.018</td>
</tr>
<tr>
<td>Soil pH</td>
<td>0.075</td>
<td>−</td>
</tr>
<tr>
<td>Grazing pressure</td>
<td>0.063</td>
<td>−</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>(b)</th>
<th>Difference in herb species between 2009 and 1993 (n2009–n1993)</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Direct</td>
<td>Indirect</td>
</tr>
<tr>
<td>N deposition</td>
<td>−</td>
<td>−0.014</td>
</tr>
<tr>
<td>SO(_x) deposition</td>
<td>−</td>
<td>−0.147</td>
</tr>
<tr>
<td>Mineral N soil</td>
<td>−</td>
<td>−0.072</td>
</tr>
<tr>
<td>Soil pH</td>
<td>0.298</td>
<td>−</td>
</tr>
</tbody>
</table>

Total effects are the sum of direct and indirect effects. Direct effect coefficients are equal to the path coefficients shown in Fig. 4a and b.

Table 3  Effect of nitrogen (N) deposition class on the mean change in frequency of individual species (expressed as the difference of frequency between 2006–2009 and 1990–1993), for those species with a change in frequency of more or less than 0.15 (15%) in the highest N deposition class

<table>
<thead>
<tr>
<th>Species</th>
<th>N deposition class (kg N ha(^{-1}) yr(^{-1}))</th>
<th>Cor.</th>
<th>P</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>0–15</td>
<td>15–25</td>
<td>25–35</td>
</tr>
<tr>
<td>Cynosurus cristatus</td>
<td>0.08</td>
<td>0.12</td>
<td>0.53</td>
</tr>
<tr>
<td>Holcus lanatus</td>
<td>0.14</td>
<td>0.20</td>
<td>0.25</td>
</tr>
<tr>
<td>Polygala vulgaris</td>
<td>0.23</td>
<td>0.06</td>
<td>0.17</td>
</tr>
<tr>
<td>Linum catharticum</td>
<td>0.27</td>
<td>–0.02</td>
<td>–0.18</td>
</tr>
<tr>
<td>Briza media</td>
<td>0.17</td>
<td>0.05</td>
<td>–0.21</td>
</tr>
<tr>
<td>Gentianella amarella</td>
<td>–0.03</td>
<td>0.12</td>
<td>–0.24</td>
</tr>
<tr>
<td>Campanula rotundifolia</td>
<td>0.05</td>
<td>–0.15</td>
<td>–0.25</td>
</tr>
<tr>
<td>Galium sterreni</td>
<td>0.11</td>
<td>0.04</td>
<td>–0.26</td>
</tr>
</tbody>
</table>

Shown are the correlation coefficients (Cor.) between the change in relative frequency of the species and modelled N deposition based on 2006–2008 data and the level of significance of the correlation. Significant correlations and their p-values (P<0.05) are indicated in bold.

stocking rates maintains species diversity by reducing the number of competitive species (Collins et al., 1998; Olff & Ritchie, 1998; Klimek et al., 2007). The absence of a management effect may result from the management regimes at all the sites being successfully focussed on preserving species richness.

Despite the lack of effect of N deposition on species richness and diversity, the frequency of rare and scarce species was significantly reduced, by a factor of 5, at sites with N deposition above 25 kg ha\(^{-1}\) yr\(^{-1}\). These relationships are difficult to test in a rigorous way in multivariate models, as rare species, such as Epipactis atrorubens, Trinia glauca, and Pulsatilla vulgaris, not only have a highly restricted geographic distribution but are also typically highly localized on the sites where they do occur. Fischer & Stöcklin (1997) and Bennie et al. (2006) both found a negative correlation between the initial frequency of a plant species and the extinction rate in calcareous grasslands over recent decades. Over the same period, Bennie et al. (2006) also found an increase in Ellenberg fertility index, although there was no significant relationship between extinction rates and the Ellenberg fertility index. Our results suggest a link between N deposition and the occurrence of scarce and rare species, which may be due to local extinctions over a period of decades.

Maximum summer temperature, along with soil depth, explained most of the variation in species richness and the frequency of different types of species: species richness was higher in plots with a higher maximum temperature. Plots with deeper soils tended to occupy moderate to low slopes and had a higher organic content (\(r^2 = –0.14, P = 0.022\)) and a higher moisture content (\(r^2 = 0.35, P < 0.001\)). The higher organic content and the fact that deeper soils are more resistant to drought stress may be important factors in explaining the higher species richness. Duckworth et al. (2000) found that temperature and soil organic matter are among the strongest parameters explaining spatial variation in vegetation composition of Atlantic European calcareous grasslands. In addition, Bennie et al. (2006) found that calcareous grassland plots on deeper soils were more likely to be invaded by (often mesotrophic) plant species, which may explain the higher species richness.

Comparison between surveys

The species richness was lower in 2006–2009 than in 1990–1993, with an average reduction in species richness by two species over 15 years; this small overall change may represent differences between surveyors in the two periods. Change in species richness was not associated with N deposition, but changes in H\(^’\) and evenness were significantly negatively related to N deposition. The Cohen’s d-value for the change in H\(^’\) was 0.99, representing a medium to strong ecological effect (Cohen, 1988). Progression towards an undesirable reference dataset in high N deposition plots showed the reduction in H\(^’\) between 1990–1993 and 2006–2009 represented a 6.4% movement towards the H\(^’\) value typical of British Neutral Grasslands. This implies that a much longer period would be needed to reach the low values of H\(^’\) typical of neutral grasslands. However, many typical calcareous grassland species may be lost well before this value of H\(^’\) is reached, and we conclude that the change in H\(^’\) at high N deposition sites does represent a significant ecological change.

H\(^’\) is a measure of diversity that accounts for both species richness and evenness of the species present in the sample (Magurran, 2004). N deposition did not significantly affect changes in species richness but did significantly affect changes in evenness, indicating that changes in H\(^’\) were driven primarily by changes in the evenness of the plots. While several previous studies have focussed on the effects of N deposition on species richness, we emphasize that changes to evenness and diversity without the loss of species are also important changes to plant communities, and may indicate the first transition to domination of the community by a few, mostly mesotrophic species. Such changes in evenness and diversity, and a shift towards vegetation dominated by a few species, have been shown in
experimental studies on calcareous grasslands in the Netherlands (Bobbink, 1991).

In addition, there was a significant decline in the frequency of five individual species when N deposition exceeded 25 kg ha\(^{-1}\) yr\(^{-1}\). All five individual species (\(L.\) catharticum, \(B.\) media, \(G.\) amarella, \(C.\) rotundifolia, and \(G.\) sterneri) have a low Ellenberg nutrient score. The decline in frequency in these high N deposition plots was substantial (on average an absolute change of 22%) over a relatively short period of 15 years despite the focused management on these nature reserves. Such changes in frequency are of particular concern for rare species such as \(G.\) amarella and \(G.\) sterneri, which are stochastically more prone to local extinctions. Both \(C.\) rotundifolia and \(B.\) media are regarded as poor competitors (Leishman, 1999; Dixon, 2002; Hutchings \textit{et al.}, 2003) and for \(C.\) rotundifolia, lower frequencies have been found at sites with high N deposition (Stevens \textit{et al.}, 2006; Maskell \textit{et al.}, 2010). \(L.\) catharticum and \(G.\) amarella are, respectively, annual and bi-annual species which may strongly depend on gap formation in the swards for germination (e.g. Turnbull \textit{et al.}, 2005). These results are corroborated by an earlier study on calcareous grasslands by Bennie \textit{et al.} (2006) who found strong decreases in frequency over the period 1950–2000 of \(C.\) rotundifolia, \(L.\) catharticum, and \(B.\) media. Apart from \(C.\) rotundifolia (5) all five species have a high Ellenberg R-score of 7 or 8. Soil pH was not related significantly to the change in frequency, except for \(L.\) catharticum, for which a slight \((r^2 = 0.14)\) relationship was found, suggesting that the pH preference of these species was not a factor in their decline.

The change in species richness was positively related to soil pH, as was herb species number, indicating that plots of higher buffering capacity tended to gain species and plots with slightly lower pH tended to lose species. Despite the fact that calcareous soils are highly buffered, experimental studies have shown that calcareous grasslands also leach base cations when N deposition is increased, as a result of soil exchange processes and plant uptake of N (Horswill \textit{et al.}, 2008). Although soil pH in the second survey was lower (7.14 compared with 7.41, paired \(t\)-test: \(t = 2.52, df = 35, P = 0.016)\), no conclusions can be drawn from this as the may be attributed to the different methods used. National surveys have identified an increase in soil pH in the United Kingdom over the last 15 years, possibly due to a reduction in acid deposition (Emmett \textit{et al.}, 2010). This may have contributed to increased species richness, especially in those plots which were least affected by base cation depletion.

\(N\) and \(P\) together are the major limiting nutrients in calcareous grasslands (Jeffrey & Pigott, 1973; Morecroft \textit{et al.}, 1994) and plant available \(P\) has been shown to influence the productivity, and hence species richness, of calcareous grasslands in several studies (e.g. Bobbink, 1991; Janssens \textit{et al.}, 1998). Plant available \(P\) in the 2006–2009 survey varied by two orders of magnitude (40–3500 \(\mu\)mol kg\(^{-1}\) DW), suggesting that it could be a limiting nutrient at least in some sites. However, no significant effects of plant available \(P\) on the change in total species richness, evenness or diversity were found in our analyses. This is consistent with a long-term experimental study in the United Kingdom which shows that, while \(P\) limitation prevents increases in productivity caused by added N deposition, it does not prevent changes in species composition (Carroll \textit{et al.}, 2003)

\textbf{Importance of direct and indirect effects of N deposition}

The use of structural equation models allowed us to identify and compare the comparative strengths of direct effects of N deposition and indirect effects of both N and S deposition, through soil pH, using hypothesized causal relationships. The responses of vegetation to changing N deposition are complex and involve several mechanisms (Bobbink \textit{et al.}, 2010), and therefore statistical methods such as SEM that incorporate this complexity and aim to disentangle different direct and indirect effects through potential causal structures provide powerful new tools to complement more conventional regression analysis.

The results show very different models for change in \(H'\) and for change in herb species numbers. Changes in \(H'\) were dominated by a direct negative effect of N deposition; although indirect negative effects of S and N deposition via soil pH were also significant, they had a much smaller contribution. The negative direct effect of N deposition on change in \(H'\) was also found for changes in evenness in the mixed effect models, suggesting that eutrophication rather than acidification is the dominant mechanism. In contrast, the direct effect of N deposition on change in herb species numbers was not significant, and was small compared with indirect effects via soil pH, indicating that acidification was the dominant mechanism.

Soil pH, which had a significant positive relationship with both changes in \(H'\) and changes in herb species numbers, was itself negatively influenced by S deposition and mineral N content. The strong acidifying nature of S in deposition compared with N was revealed by the stronger negative effects that were exerted by S deposition on soil pH. Calcareous grasslands have been shown to retain most of the deposited N in the system with little leaching to deeper soil layers or groundwater, thereby increasing mineral N content of the top soils (Phoenix \textit{et al.}, 2008). The increased
mineral N in the soils in turn can have a significant negative effect on soil pH (e.g. Horswill et al., 2008). This is consistent with the positive relationships between N deposition and mineral N content and the significant negative relationships between the mineral N content and soil pH in our study. These findings, using a different method of analysis, are consistent with earlier studies of acid grasslands in which negative effects of soil acidity on herb species numbers were recorded (e.g. Stevens et al., 2006; Duprè et al., 2010; Stevens et al., 2010).

A recent survey of changes in a local UK flora over the last 50 years showed the positive effects of the designation of conservation sites on maintaining biodiversity, especially in calcareous grasslands; in addition the increase of tall species was attributed to the cessation of grassland management (Walker et al., 2009). In our structural equation model, change in grazing also significantly improved the model fit, with a decrease in grazing pressure being directly associated with a decrease in $H'$. A decrease in grazing pressure was also associated with an increase in Ellenberg N score. These results may reflect an effect of poor management on some sites over recent years, since the spatial analysis suggested that grazing management in 1990–1993 was optimal overall and hence did not affect spatial variation in species richness or diversity.

Conclusions

Critical loads for adverse effects of N deposition for calcareous grasslands, based on field experiments, have been set as a range of 15–25 kg N ha$^{-1}$ yr$^{-1}$ (Bobbink et al., 2003, 2010), with the range indicating the variation in sensitivity. Our results provide the first evidence from field data of a decrease in species diversity and evenness, a decline in the frequency of characteristic species, and a lower number of rare and scarce species, when the critical load range is exceeded. These effects are likely to be due to both direct effects of N deposition and indirect effects through changes in soil pH. Since all sites had active management to maintain a high diversity and characteristic species, our results imply that even focussed management on nature conservation objectives cannot prevent adverse effects of high rates of N deposition. Given the high and increasing rates of N deposition in many parts of the world, the threat that N deposition poses to the conservation of species-rich grassland communities, and their characteristic species, under different management regimes needs further assessment.

Acknowledgements

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References


N DEPOSITION EFFECTS ON CALCAREOUS GRASSLANDS


