



# How will the semi-natural vegetation of the UK have changed by 2030 given likely changes in nitrogen deposition?



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## ABSTRACT

Nitrogen deposition is known to have major impacts on contemporary ecosystems but few studies have addressed how these impacts will develop over coming decades. We consider likely changes to British semi-natural vegetation up to the year 2030 both qualitatively, based on knowledge of species responses from experimental and gradient studies, and quantitatively, based on modelling of species relationships in national monitoring data. We used historical N deposition trends and national predictions of changing deposition to calculate cumulative deposition from 1900 to 2030. Data from the Countryside Survey (1978, 1990 and 1998) was used to parameterise models relating cumulative N deposition to Ellenberg N which were then applied to expected future deposition trends. Changes to habitat suitability for key species of grassland, heathland and bog, and broadleaved woodland to 2030 were predicted using the MultiMOVE model. In UK woodlands by 2030 there is likely to be reduced occurrence of lichens, increased grass cover and a shift towards more nitrophilic vascular plant species. In grasslands we expect changing species composition with reduced occurrence of terricolous lichens and, at least in acid grasslands, reduced species richness. In heaths and bogs we project overall reductions in species richness with decreased occurrence of terricolous lichens and some bryophytes, reduced cover of dwarf shrubs and small increases in grasses. Our study clearly suggests that changes in vegetation due to nitrogen deposition are likely to continue through coming decades.

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## 1. Introduction

Human activity increasingly dominates the global nitrogen (N) cycle with anthropogenic production of reduced and oxidised N compounds ('reactive N') more than double natural N fixation (Sutton et al., 2011). Although human impact on the global N cycle has a long history, encompassing legume cultivation for several millennia, fossil fuel burning for several centuries and nitrate mining since the 19th century, the critical change was the invention of the Haber-Bosch process (Galloway and Cowling, 2002; Kopáček and Posch, 2011). Fritz Haber's discovery led to the production of reactive N from inert N<sub>2</sub> on an industrial scale in the post-war era. Global average N deposition increased by more than fourfold from 1860 to 1993 and under an IPCC A2 scenario is predicted to double again by 2050 (Dentener et al., 2006). This increasing deposition

will affect an ever greater proportion of the earth's sensitive ecosystems (Bleeker et al., 2011; Phoenix et al., 2006).

In the United Kingdom oxidised N emissions increased sharply from the 1940s with the greatest increase to 1960 and then a slower increase to a peak around 1980. Reduced N emissions increased more gradually over the same period with total N deposition peaking around 1990 at 430 kt N (Fowler et al., 2004). In recent decades there has been a reduction in emissions of both reduced (−24%; 1990–2007) and oxidised N (−46%; 1990–2007 (RoTAP, 2012)) due to technological change and decline in industrial production. However, changes in atmospheric chemistry have meant that these reduced emissions have not translated to equivalent declines in deposition, with only minor change in total N deposition over the last 20 years (RoTAP, 2012). N deposition is expected to fall during the current decade due to a decline in vehicle emissions and increased uptake of mitigation measures for agricultural emissions, although only limited further change is predicted by 2030 (RoTAP, 2012).

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### 1.1. Approaches to predicting vegetation change

Although numerous studies have investigated N deposition impacts in contemporary ecosystems, and several have addressed future deposition trends, there has been comparatively little attention to predicting future change across semi-natural habitats. Some work of this type has been conducted as part of the CCE critical loads process. Hettelingh et al. (2008) have applied dose–response functions, based on European N-addition experiments of over two-year duration (Bobbink, 2008; Bobbink and Hettelingh, 2011), to predict responses of species richness of selected habitats to projected future N deposition. Results suggest modest increases in plant species richness by 2020 in shrubs, forests and grasslands under scenarios of both maximum feasible reduction and current legislation. There are a number of important limitations to this approach, which mean that the tentative results these authors present may not be realistic. The analysis is based solely on experimental studies with the well-known limitations of small spatial scale, short duration, unrealistic treatment regimes, high N loads and high ambient deposition and for many habitats the number of published studies is very limited. Furthermore, by using relationships with current N deposition, models assume instant recovery from reduced deposition, which is not ecologically realistic as N may accumulate and there may be lags in species responses. In relatively-small treated plots surrounded by unaffected vegetation local sources of propagules may allow re-establishment of sensitive species much quicker than in an entire polluted landscape. It is therefore probable that recovery rates may be systematically over-estimated in experimental studies.

An alternative approach to these simple dose–response calculations has been the use of dynamic soil-vegetation model chains combining models such as ForSAFE-VEG, MAGIC, NTM, MOVE and GBMOVE (de Vries et al., 2010; Smart et al., 2010; Sverdrup et al., 2007; Wamelink et al., 2009). In these approaches a dynamic soil model predicts change in the soil environment in response to changing deposition which feeds into a statistical or process-based model of vegetation response to changing soil conditions. Much of the application of such model chains for prediction has so far been exploratory but results suggest that, for instance, the biodiversity of Dutch forests will remain constant over the coming decades while that of heathland and grassland will increase (Wamelink et al., 2009). The challenges in such modelling remain considerable due to both the intrinsic complexity of the problem and the limited availability of suitable data sources, in particular paired soil chemistry and vegetation datasets. Results necessarily incorporate large uncertainties. Other predictive studies have modelled just a few dominant plant species (Bobbink and Heil, 1993; Terry et al., 2004) or focussed on other response variables such as total biomass or biogeochemical function (Aber et al., 1997).

In many terrestrial ecosystems there is evidence that N inputs are considerably greater than N losses to water courses and the atmosphere, with N accumulating in soils over time (Fowler et al., 2004; Pilkington et al., 2005). Consequently air pollution impact studies have increasingly considered that total cumulative deposition over an extended period of time is a more useful metric than annual deposition (De Schrijver et al., 2011; Duprè et al., 2010; Phoenix et al., 2012; Payne, 2014). As current N deposition will never fall to zero, cumulative nitrogen deposition values can only increase. If cumulative N deposition is used as the sole environmental driver in a predictive study an assumption is thereby made that ecosystem recovery from deposition impacts is not possible. If deposition falls, the rate at which deleterious impacts develop may slow but never stop or reverse.

The available experimental evidence suggests that this is a simplification; some recovery in vegetation does occur but total

recovery is likely to take an extremely long time, if this is indeed possible. Recovery speed is also likely to vary between habitats depending on the ability of the soils to accumulate N and the vegetation composition. For example, in boreal forests Strengbom et al. (2001) showed no detectable recovery after nine years of N treatment cessation and impacts on vegetation composition were still marked after 47 years of recovery. In hay-meadows several studies have shown impacts lasting more than a decade with no complete return to prior conditions (Králóvec et al., 2009; Olf and Bakker, 1991; Stevens et al., 2012a). In UK heathlands experiments have shown impacts on lichen cover, *Calluna vulgaris* growth and flowering and litter nutrients over seven and eight years after N treatment stopped (Edmondson et al., 2013; Power et al., 2006). Stevens et al. (2012a) suggest that some impacts on vegetation may be non-reversible with less-competitive species unable to replace N-tolerant dominants once established. Indeed where eutrophication and acidification have driven coupled above and below-ground regime shifts in pH, phosphorus availability and nutrient cycling then recovery is highly unlikely to occur by simply reducing deposition (Baer and Blair, 2008; Chen et al., 2013; Tateno and Chapin, 1997).

Given the scarcity of recovery studies and their limitations we do not consider the realistic incorporation of ecosystem recovery rates to be feasible. Alternatives are either to use current deposition (assuming instant recovery from reduced deposition) or cumulative deposition (assuming no recovery from reduced deposition) as environmental driver. Although neither gives the full picture we believe that the assumption of ‘no recovery’ is closer to the available experimental results.

Our aim here is to provide provisional predictions of future change in the community composition of UK semi-natural habitats due to N deposition. To do this we assess how possible trends in N deposition may impact on vegetation both qualitatively, on the basis of N addition experiments and gradient studies, and quantitatively by modelling cumulative N deposition relationships with Ellenberg N scores and then using the MultiMOVE model to assess changing species habitat suitability under projected N deposition for the years 2020 and 2030. Overall we hypothesise that over time habitat suitability for many desirable species will decline in all three habitats as a consequence of elevated N deposition resulting in changes in species composition.

## 2. Methods

Three groups of semi-natural vegetation were considered: grasslands, heaths and bogs, and broadleaved woodlands. Data were taken from the Countryside Survey (CS) of Great Britain (<http://www.countrysidesurvey.org.uk/>) for the years 1978, 1990 and 1998. In CS samples are collected from stratified, randomly selected  $1 \times 1$  km squares, 569 squares were surveyed in the 1998 survey.  $2 \times 2$  m vegetation plots were located within each  $1 \times 1$  km square using a restricted randomization procedure to reduce aggregation. In each plot all vascular plants and a selected range of the more easily identifiable bryophytes and macrolichens were identified to a species level and cover estimates made to the nearest 5%. The methods used for vegetation monitoring are described in detail in Smart et al. (2003). Plots were only included in the analysis where the vegetation sampled at time 1 could be classified into one of these three major categories of vegetation responsive to nitrogen addition. The assignment of each plot to each category was done on the basis of its species compositional similarity to the habitat types used for the definition of empirical critical loads for nitrogen in Europe. This step was carried out by determining the match between the Countryside Vegetation System class to which each plot was allocated (see Bunce et al., 1999; Smart et al., 2003, 2004) and

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