

Measures to evaluate benefits to UK semi-natural habitats of reductions in nitrogen deposition



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Rowe EC, Jones L, Stevens CJ, Vieno M, Dore AJ, Hall J, Sutton M, Mills G, Evans CD, Helliwell RC, Britton AJ, Mitchell RJ, Caporn SJ, Dise NB, Field C & Emmett BA (2014) Measures to evaluate benefits to UK semi-natural habitats of reductions in nitrogen deposition. Final report on REBEND project (Defra AQ0823; CEH NEC04307).

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Executive summary

There is strong evidence that nitrogen (N) deposition is having impacts on UK semi-natural habitats. Improved understanding of impacts of cumulative deposition and deposition reductions, and of the extent to which habitat responses to future N deposition can be evaluated through use of existing tools and evidence, is required. There is also a need to assess and recommend alternative metric(s) to evaluate the benefits of reductions in nitrogen deposition.

Reductions in N deposition can have delayed effects, due to persistence of N in soil and vegetation and delays to re-colonisation by target species. Although reductions in current deposition are likely to rapidly decrease plant exposure to N, stored N will result in a sustained release of plant-available N, so cumulative deposition needs to be taken into account. Cumulative deposition above the Critical Load for the habitat during the preceding 30 years is recommended as a pressure metric for soil-based habitats. This is a measure of the influx of damaging N during the period for which stored N is released in significant amounts.

Midpoint metrics relating to ecosystem function and risk of degradation, and endpoint metrics relating to achievement of favourable conservation status and other goals, were not fully operationalised in the study. Promising midpoint metrics are moss tissue N concentration at N deposition rates up to 25 kg N ha⁻¹ yr⁻¹; and N leaching rate at greater N deposition rates. Species-richness may be an appropriate endpoint metric for grasslands, but for other habitats it would be useful to develop specific metrics of habitat quality, for example based on target species. Applying these metrics to the reporting of N impacts would require relationships to be developed between the metric and current, cumulative and decreased deposition.

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Technical summary

Context and objectives of project

There is strong evidence that nitrogen (N) deposition has caused changes to sensitive semi-natural habitats in the UK. For example, N deposition has caused a reduction of plant species richness in a range of habitats (RoTAP, 2012). Analysis of broad-scale vegetation datasets has allowed a temporal and spatial assessment of N deposition impacts. Many effects on species are likely to have taken place before the 1980s, but current N deposition in many parts of the UK is associated with further declines in the occurrence of sensitive plant species (Emmett et al., 2011; RoTAP, 2012; Stevens et al., 2011c).

The concept of 'critical load' is currently the main tool used in risk assessment of N deposition impacts, both for national scale evaluation of policies and for site-specific impact assessment. There is strong evidence that exceedance of critical loads is associated with negative impacts on terrestrial biodiversity at the UK scale (RoTAP, 2012). There is widespread exceedance of empirical critical loads for acidity (mainly due to N deposition) and for nutrient-N across the UK (Hall et al., 2011). Based on current emission forecasts, this is likely to continue over the next 10-20 years. However, current risk assessment tools and evidence are limited with respect to how cumulative doses of N are taken into account. These methods do not tell us how, or over what timescales, ecosystems will respond to reductions in deposition, which nevertheless may continue to be above the critical load. Benefits of reductions in deposition are not well represented by metrics based on the area where critical load is exceeded, since despite decreases in recent years N deposition remains above the critical load over large areas.

The objective of this project was to provide a "think-piece" describing the tools available to evaluate the marginal benefits to semi-natural habitats of reductions in N deposition, and therefore of measures to reduce emissions. The study aimed to improve understanding of what can realistically be achieved in terms of reducing impacts and aiding recovery, and how this varies spatially. Areas which have experienced different inputs of N deposition historically may respond differently to future inputs. There is also a requirement for greater understanding of the timescales over which changes will occur. A particular challenge to answering this is the currently poor knowledge base on the fate of deposited N in soils.

The impacts of N deposition on biodiversity have implications for meeting UK and country 'conservation commitments', such as achieving favourable conservation status of habitats listed under Annex I of the Habitats Directive. In this respect, it is necessary to ensure the ecosystem structures and functions are in place to maintain a habitat over time. Pressures such as N deposition may affect the structure and function of habitats, and hence affect their long-term viability. In many areas habitats are already impacted by N, so there is also a need to more fully understand how future N deposition affects the viability of habitats which may already have been impacted by N, to inform objective setting and action. There is a requirement to demonstrate the benefits of reducing N deposition to protected sites, or sensitive habitats more generally, even where reductions may not lead to non-exceedance.

Evidence for cumulative N impacts and implications for recovery (WP1)

To assess benefits of decreases in N deposition it is important to understand how ecosystems have been affected by past deposition. We reviewed evidence from experiments, surveys and modelling

studies. There is strong evidence that atmospheric N pollution has had profound and long-lasting effects on UK habitats, in particular driving the loss of sensitive species from substantial areas of the UK. Nitrogen can be very persistent in soil and vegetation, and the favourable conservation status of a habitat is likely to be more closely related to cumulative deposition than to current deposition. As well as N persistence and other delays to the recovery of chemical conditions, there can be delays in biological recovery due to constraints on dispersal and re-establishment. Evidence for estimating the length of these biological delays is sparse.

There have been several recent studies of N impacts in relation to cumulative N deposition. However, these studies have used cumulative deposition calculated as total deposition integrated over a long period. Since historical variation in the spatial pattern of deposition is unknown and considered to be relatively small, cumulative N deposition calculated in this way has the same spatial pattern as current deposition and so offers little or no extra explanatory power. Calculating cumulative N deposition as total deposition over a long period also places too much emphasis on historical deposition. The great majority of the N deposited in the early 20th century, for example, will now have been either lost from the ecosystem or incorporated into soil pools that are largely unavailable to plants, so this N is of little biological relevance. The limited evidence that it is cumulative N deposition rather than current deposition that affects habitats is therefore unsurprising, and we recommend re-analysis using a more responsive measure of cumulative deposition such as the 30-year metric described below.

Ecosystems vary in their retention of N, with smaller N stocks and faster turnover rates in epiphytic and epilithic systems than in soil. These ecosystems are usually viewed as a component of a habitat (e.g. epiphytic lichens in Atlantic oakwoods; epilithic lichens in montane habitats). It is necessary to distinguish between soil-based and non-soil-based components, and to note that cumulative deposition is of relatively greater importance in soil-based habitat components where considerable amounts of N can build up. Receptors which primarily respond to current atmospheric deposition will recover faster than those whose responses are mediated by accumulated soil nitrogen pools, and so recent deposition is likely to be more relevant than cumulative deposition for epiphytic and epilithic systems.

Relatively few experiments have studied effects of decreases, in particular small decreases, in N deposition rate and such effects have rarely been reported. One study of reciprocal transplants of epiphytic lichens (Mitchell et al., 2004) showed clear and rapid effects of a decrease in N deposition rate, corresponding to the expected faster response of chemical conditions in such ecosystems. However, effects of decreases have mainly been inferred from relationships between current deposition and observed responses, for example of species-richness. This approach takes too little account of delays to chemical and biological recovery, but it should not be concluded that decreases in N deposition are not beneficial. A key conclusion from studies of large floristic datasets (Emmett et al., 2011; Stevens et al., 2011c) is that species are affected progressively. Sensitive species can be lost even at deposition rates below the critical load. Less-sensitive species can be lost with marginal increases in N deposition from rates that are already well above the critical load.

In less-affected systems, decreases in N deposition will allow N-sensitive species to return, albeit with an uncertain timescale. In highly-affected systems, decreases will allow the return of those species for which the habitat is currently of borderline suitability. The spatial targeting of measures to reduce N deposition will depend on the relative importance attached to recovery of N-sensitive and less-N-sensitive species. This is not an easy question to resolve, not least because the areas in the UK that have been most affected by N are close to major population centres. However, it can be concluded that decreases in N deposition are beneficial across the range of N deposition.

The differential sensitivity to N deposition of different components of ecosystems makes it difficult to define a threshold for damage or recovery. Although the concept of critical load is useful as an indicator of deposition below which damage is likely to be minimal, effects on species have been observed with deposition rates below the critical load. Conversely, if critical load is exceeded the ecosystem should not be seen as completely and irrevocably damaged – as noted above, sustaining or increasing this deposition rate is likely to result in increasing damage, by putting more species at risk.

Decreasing deposition rates will not induce an immediate return to pre-industrial or ‘no-effect’ conditions, because of chemical and biological delays. Decreasing to a rate that is still above the critical load will never allow recovery to below the threshold for damage that was used to establish the critical load, by definition. However, any decrease is likely to favour some species – those for which N pollution has made the habitat only just unsuitable.

Metrics to represent benefits of marginal reductions in N deposition (WP2)

It is useful to distinguish metrics of *pressure*; *midpoint* metrics that indicate progress towards biodiversity targets and other goals in terms of ecosystem functions and services; and *endpoint* metrics that represent the degree to which these goals have been achieved. We reviewed metrics of these three types, including some new proposed metrics, that might better encapsulate and communicate current understanding of N impacts on semi-natural habitats and the benefits of reductions in N deposition.

Both current deposition and long-term cumulative N deposition have limitations as pressure metrics. Using current deposition (and derived metrics such as the area of sensitive habitats where critical load is exceeded) does not represent current understanding of N persistence. Conversely, metrics based on long-term cumulative total N deposition are likely to over-represent the effects of historical N deposition. Metrics based on integrating deposition over a relevant timeframe are likely to better represent persistent effects. For epiphytic and epilithic components of habitats, in which N is likely to be retained for a comparatively short period, cumulative deposition over the preceding three years is an appropriate metric. For soil-based components, a timeframe of 30 years is relevant. The differential sensitivity of different habitats can be taken into account by calculating cumulative deposition above the critical load. We therefore propose a pressure metric that is suitable for all habitats: ‘CE30’ i.e. deposition above the critical load accumulated over the preceding 30 years.

Midpoint metrics could be based on the biogeochemical properties of any part of an ecosystem, where these properties can be related to N deposition. Relationships with several soil properties are not straightforward, however. Bulk soil properties (N concentration and C/N ratio) are affected slowly and inconsistently by N deposition. Measures of soil N availability are susceptible to rapid fluctuation and / or large sampling error, and the variety of methods used has made it hard to derive reliable relationships. The most clear and consistent relationships with current N deposition have been obtained for N leaching rate (for deposition of $> 25 \text{ kg N ha}^{-1} \text{ yr}^{-1}$) and for moss tissue N concentration (for deposition of $< 25 \text{ kg N ha}^{-1} \text{ yr}^{-1}$). Moss tissue N is simple and cheap to measure, and we recommend the development of an ‘MEI’ metric based on this relationship, adjusted for the typical N content of different moss species.

Measurements of some biogeochemical properties can also be seen as endpoint metrics, where these properties correspond directly to ecosystem services – N leaching rate, which affects downstream water quality, is an example. Appropriate endpoint metrics for assessing progress towards conservation goals are mainly based on biotic measurements. Most of the available studies

relate to floristic (plant and lichen) components. The occurrence of many individual species was shown to be related to the current spatial pattern of N deposition in a study of large floristic datasets (Emmett et al., 2011; Stevens et al., 2011c). That study provided strong evidence for a need to control N deposition, but metrics based on individual species would have limited applicability. Some aggregate indicators for the floristic assemblage, such as species-richness or mean 'Ellenberg N' score, have also been shown to be correlated with current N deposition. Species-richness is easily understood, and a large number of species indicates favourable conservation status in many, though not all, habitats. A metric based on relationships between species-richness and N deposition could readily be derived. However, favourable conservation status tends to be considered as related to the occurrence (and sometimes absence) of particular species. We propose development of a metric of habitat quality based on the target species for a given habitat, 'HQI', in conjunction with habitat experts.

Midpoint and endpoint metrics derived using static regression relationships with deposition will respond instantaneously to changes in deposition, i.e. will not take into account delays to chemical and biological recovery. Dynamic modelling approaches to calculating these metrics have the potential to allow for such delays, although currently only chemical delays have been modelled satisfactorily.

Evidence gaps and research recommendations (WP3)

Most biogeochemical studies have assessed changes in relation to current deposition, or to measures of cumulative deposition that are completely correlated with current deposition. It would be useful to reanalyse response data in relation to a more responsive measure of cumulative deposition, e.g. CE30 as described above.

A key uncertainty regarding ecological responses to N deposition is whether plant productivity in a given habitat is limited by N. If productivity is limited by other factors, for example phosphorus availability, then the main effects of N pollution will be limited to direct toxicity, and increased leaching and consequent acidification. If N limits productivity, however, then leaching and acidification responses will be very limited, but extra N is likely to increase biomass, canopy height and litter production. These changes will result in decreases in ground-level light availability and faster closure of gaps in vegetation, which are major mechanisms for species loss. Investigating where, in terms of habitat and N deposition history, N limits productivity would greatly improve understanding of N impacts and should be a research priority for the medium term.

Experimental studies in which N deposition rates have been decreased are rare. Experiments incorporating treatments with small reductions in deposition rate should be established, and measurements that are suitable for calculating key metrics of ecosystem response taken.

Although measurements of plant-available N are not immediately suitable for calculating metrics for reasons explained above, they can provide clear evidence that N deposition is affecting a key ecosystem function. Where N limits plant growth, the amount of plant-available N determines plant productivity, with profound effects on interspecific competition and the viability of the habitat for particular species. There is a requirement for further research on the most appropriate measure of plant-available N, and for method intercomparison studies.

Soil-vegetation models have been developed to simulate biogeochemical and floristic responses to N deposition. These models have the potential to generate various metrics in response to scenarios of change to N deposition and other environmental drivers. Further testing of the models against

empirical data, in particular comparison of alternative models, would strengthen the case for their application in policy evaluation and development.

Next steps

The proposed cumulative deposition metric, CE30, is already defined, and could be applied immediately to assess effects of different N deposition scenarios on the pressure on habitats. The moss tissue N index, MEI, would require some development but could be used in the near-term to indicate likely risk to habitats. The requirement for an endpoint metric that reflects definitions of favourable conservation status is urgent, and steps should be taken in the near-term to operationalise such a metric.

1. Introduction

Atmospheric pollution by reactive nitrogen (N) is a global threat to biodiversity (Bobbink et al., 2010; Phoenix et al., 2006; Sala et al., 2000) and is driving major changes in semi-natural habitats in the UK (e.g. Emmett et al., 2011; Maskell et al., 2010). Nitrogen is often a key constraint on plant growth in natural ecosystems, and the changes effected can be profound, and long-lasting because of N retention and recycling within the ecosystem. Efforts to decrease atmospheric N pollution need to be supported by a good understanding of the effects of present-day emissions on ecosystems, and how these are influenced by the history of N deposition onto a site. This report focuses on the evidence for effects of marginal decreases in N deposition on sites that vary in the cumulative amount of N pollution they have received. Benefits of such decreases are assessed, as are metrics that could be used to represent these benefits.

The aims of the study were to:

- Provide an overview of the current knowledge base in respect of cumulative impacts of N deposition and prospects for recovery in response to decreases in N deposition rate.
- Describe, and critically assess, how habitat response to future nitrogen deposition can be evaluated through the use of current tools (including models) and evidence.
- Recommend alternative metric(s) to evaluate the benefits of decreases in nitrogen deposition.
- Identify evidence gaps and recommend the focus of research and development to address these gaps, and prioritise these for the short term and medium term.

Nitrogen causes damage to habitats and species through toxic effects, the acidifying effect of N leaching, and in particular by increasing the growth rate of competitive plants at the expense of shorter-growing and stress-tolerant species. Some effects are rapid and occur at low levels of atmospheric N pollution, notably declines in occurrence of sensitive species. Others are chronic, and relate to the accumulation and persistence of N in ecosystems. There is an increasing body of evidence that N pollution has caused widespread loss of plant and lichen species in the UK, and continues to impoverish UK habitats even in areas that have received large cumulative doses of N deposition (Stevens et al., 2011c).

Alongside health concerns (mainly related to the role of N oxides in the formation of ground-level ozone), the need to reduce or prevent loss of biodiversity has been a major impetus for decreasing emissions of reactive N. Taking a broader view of the functions and services provided by ecosystems, however, it is clear that N can have positive as well as negative effects. The stimulation of plant productivity by atmospheric N increases provisioning services, by improving crop and forest yields and protein contents. This allows a decrease in the financial and environmental costs of producing and applying N fertiliser. In woodlands, increased N availability increases the fixation and storage of carbon (C), at rates estimated at 15-40 kg C kg⁻¹ N (de Vries and Posch, 2011). Increases in global warming potential due to greenhouse gas (N₂O) emissions from agricultural systems are outweighed by increased rates of C storage in non-agricultural systems, even when fertiliser and manure N applications to agriculture are also considered (de Vries et al., 2011). Whether N deposition has an overall benefit or cost depends on the degree to which ecosystem services are correlated with occurrence of N-sensitive species and the habitats they form, and to the values assigned to these services. In a review of the effects of air quality on ecosystem services (Jones et al., in prep), the economic value assigned to biodiversity was the major determinant of whether a decrease in N emissions had a net cost or benefit, but this value was considered highly uncertain. This report focuses mainly on the effects of N deposition on biodiversity as defined in legal instruments, in particular the EC Habitats Directive and its interpretation in terms of favourable conservation status.

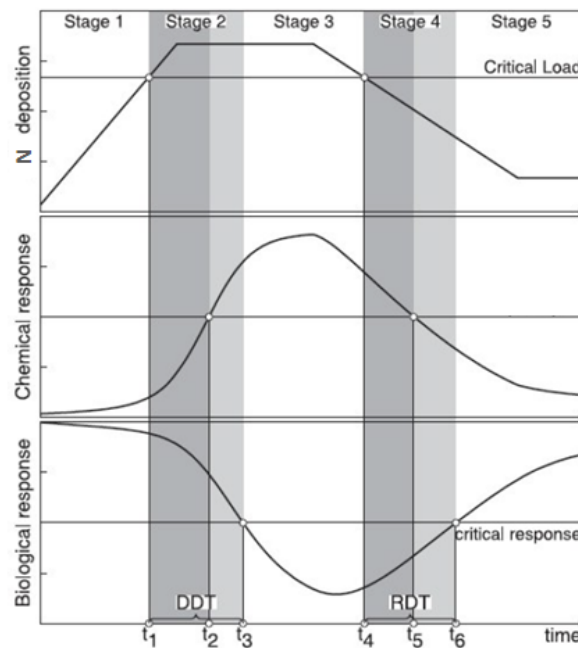
Emissions of oxidised nitrogen (NO_x) come mainly from burning fossil fuels. In the UK the biggest sources of emissions are transport, combustion in industry, and electricity and heat generation (RoTAP, 2012). Emissions of chemically reduced nitrogen (NH_y) come mainly from agriculture and in particular the management of manure. Cattle, poultry and other livestock are the biggest sources of atmospheric NH_y . Between 1900 and 2000 68 Tg of reactive nitrogen ($\text{NO}_x + \text{NH}_y$) was emitted to the atmosphere in the UK, 43 % of this was deposited representing an average of between 1 and 5 t N ha^{-1} deposited to semi-natural habitats (Fowler et al., 2004). Emissions of ammonia have declined from highest levels in the 1990s but decreases have been much smaller than for nitrogen oxides and have been driven mainly by decreases in animal numbers and fertiliser use (RoTAP, 2012). Within these average numbers there is considerable geographic variation in nitrogen deposition. Total nitrogen deposition ranges from less than 10 kg N $\text{ha}^{-1} \text{yr}^{-1}$ in areas like north-west Scotland and west Wales and reaches over 30 kg N $\text{ha}^{-1} \text{yr}^{-1}$ in areas like the Peak District. In much of the country deposition of reduced N is slightly greater than of oxidised N but some areas receive more of one N form than another, for example, deposition of ammonium is high relative to nitrate in the Midlands region (RoTAP, 2012). Overall, the total amounts of reduced and oxidised N deposited on UK habitats are similar.

Initiatives to reduce N emissions have been fairly successful. There was a 21% decrease in ammonia emissions between 1980 and 2010 and a 62% decrease in nitrogen oxide emissions between 1990 and 2010 (DEFRA, 2012). However, there has not been a commensurate decrease in N deposition and its reported impacts; the percentage of sensitive-habitat area where the critical load for nutrient-nitrogen is exceeded declined from 75% in 1996 to 71% in 2007. The current study assesses some of the reasons for the discrepancy between falling emissions and a lack of response of impact indicators. One reason for the mismatch between changes in emissions and changes in critical load exceedance is the amount of N exported in long-range transport. Even where UK emissions have decreased, this may be reflected in reduced export of N rather than declines in deposition of N within the UK. The UK as a whole has seen a decrease of approximately 24 % in wet deposition and 23 % in dry deposition of nitrate between 1988 and 2008, but decreases in wet and dry deposition of reduced nitrogen have been negligible (RoTAP, 2012).

Effects of air nitrogen pollution on ecosystems represent a combination of both short-term peak concentrations and deposition events and long-term exposure. In the case of air concentrations these differences have been reflected by positing both short term (e.g. hourly and daily) critical levels as well as long-term (e.g. annual, multi-year) critical levels. However, the evidence based for these distinctions remains relatively limited, with the result that the main focus of critical levels is currently at the annual/multi-annual scale (e.g. Cape et al., 2009). In practice, it remains an open question what is the relative contribution of peak concentrations to long term averages in the observation of ecosystem responses, from foliar injury to long-term species change. Such uncertainties apply similarly to the effect of temporal distribution in wet deposition, since a significant fraction of annual wet deposited nitrogen can occur in a few precipitation events.

A major issue when considering the effects of N on ecosystems is the amount of buffering or resistance to change in different components of the system. As N deposition rate increases, the rate of increase in a chemical response (such as plant-available N in soil) will increase. If deposition exceeds the critical load, this will cause the chemical indicator to exceed a critical threshold, but only after a delay. This chemical response will drive a biological response, but a biological indicator may only exceed its critical threshold after a further delay (Figure 1).

Figure 1. (Adapted from Posch et al., 2004). Delayed effects of changes in N deposition on a chemical indicator and a biological indicator. Deposition above the critical load (top graph, Stage 2) causes a chemical response, for example in conditions in the soil solution, to exceed a critical level after time ($t_2 - t_1$). The biological response to these conditions is further delayed, and only passes a critical level after time ($t_3 - t_1$), called the Damage Delay Time (DDT). Biological recovery after deposition reduces below the critical load is similarly delayed, by the Recovery Delay Time (RDT).



The persistence of N in ecosystems implies that impacts depend on not only current but past N deposition. Several recent studies (e.g. Dupre et al., 2010; Payne et al., 2011; Phoenix et al., 2012) have attempted to assess whether cumulative N deposition is a better predictor of ecosystem impacts than is current deposition. However, these studies have used total cumulative deposition over long periods (e.g. since 1945 in Phoenix et al., 2012), which usually dwarfs experimental additions over the more limited timescale of experiments. It should also be noted that total cumulative deposition will never decrease and so cannot capture benefits associated with N decrease in chronically polluted systems, other than a slowing of the rate of N accumulation.

Calculations of cumulative deposition are based on adding deposition for each year over a given time period. Using a non-zero deposition threshold allows small amounts of N deposition that are not considered harmful to be taken into account. The best time period for integration of N deposition data may depend on the period of deposition inputs, levels of uncertainty in historical data, and in particular the period for which N is likely to be retained in the ecosystem. Using a shorter time window for integrating cumulative deposition is likely to give better relationships with ecosystem effects, as well as providing a metric that can decrease in response to decreases in current deposition. Methods for calculating cumulative N deposition are discussed in Section 3.2.

Deposition of N represents a pressure on the ecosystem, and metrics derived from deposition rate are pressure metrics. It is useful to consider whether metrics based on ecosystem responses can be also defined. The study examined evidence for such responses from empirical and modelling studies, and assessed whether this was sufficient to define robust and informative response metrics. Ecosystem responses of different kinds can be related to habitat structure and function, including the presence of species and other attributes important to biodiversity value, and also to the services provided by the ecosystem.

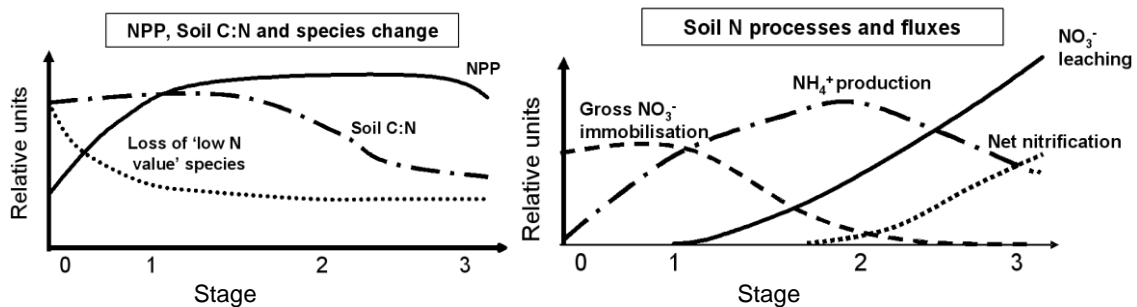
This report addresses the aims of the project firstly by reviewing evidence for cumulative impacts of N deposition, and for recovery following decreases in N deposition (Section 2). Evidence from experiments and surveys is reviewed in relation to chemical change, for example of concentrations of N in plant tissue and soil, and to biological change. Insights provided by modelling studies of biogeochemical and floristic change are also assessed. Section 3 addresses methods for calculating cumulative deposition, and implications of the choice of method for spatial variation in this pressure metric across the UK. Next, the evidence base and current tools and metrics are summarised and critically assessed in relation to the qualities of an ‘ideal’ metric (Section 4). Potentially useful metrics are reviewed and shortlisted in Section 5. In Section 6, evidence that would be useful for supporting, defining and calculating metrics, but is currently lacking, is reviewed. Conclusions and recommendations are made in Section 7.

2. Impacts of cumulative nitrogen deposition and recovery

2.1 Nitrogen accumulation in ecosystems – concepts

Impacts of nitrogen deposition on soils and vegetation are discussed in detail in sections 2.3 to 2.6, but broadly fall into two classes, acute effects and chronic effects. Aber et al. (1998) presented an outline of processes likely to occur with accumulation of deposited N in forests, such as an initial increase in plant productivity, and the onset at later stages of N accumulation of nitrate leaching and consequent acidification. This concept was updated by Emmett (2007) to include effects on N-sensitive species (i.e. ‘low N-value’ species; Figure 2).

Figure 2. From Emmett (2007). Timing, in relation to continued N deposition and progressive N saturation of an ecosystem, of changes in soil C/N ratio, net primary productivity (NPP), occurrence of plant species associated with low N availability, gross microbial nitrate immobilisation, ammonium production and nitrate leaching.



More recently, Lovett & Goodale (2011) distinguished between cumulative N saturation and ‘kinetic’ saturation where the dose exceeds short-term retention capacity. It may be useful to consider N additions in relation to short-term N retention capacity:

1. *Deposition < short-term retention capacity.* Sensitive species may be reduced in abundance or locally lost, and plant productivity stimulated. Possible increase in dissolved organic N flux, otherwise little change in soil solution chemistry.
2. *Deposition ~ short-term retention capacity.* Further decline in of N-sensitive species, increased plant productivity and litter production. Mineral N released into soil solution and largely taken up by plants, but detectable in winter or by using resin sorption or similar methods to preempt plant uptake.

3. *Deposition > short-term retention capacity*. Further loss of N-sensitive species, increased plant productivity and litter production. Increasing amounts of mineral N in soil solution, with a greater proportion in the more damaging nitrate form in mineral soils (Rowe et al., 2012).

The Lovett & Goodale concept is probably more closely related to the dynamic nature of N processing in ecosystems, but the Aber / Emmett concept illustrates well the progression of damage and effects on different indicators at different stages. These schemas are useful for illustration, but it is not easy to arrive at concrete definitions of the 'stage' of N saturation or of the 'short-term retention capacity', so they cannot be used directly for metric definition.

It is apparent from experimental studies that ecosystems can retain large amounts of deposited N, much of it in soil N pools with slow turnover rates (Phoenix et al., 2012). Recent modelling work suggests that since N deposition can stimulate the production of plant litter with relatively high C/N ratio, soil C/N ratio is likely to increase with N accumulation in many systems (Rowe et al., 2011b). This is supported for some habitats by observations from surveys e.g. Countryside Survey (Jones et al., 2004; Reynolds et al., in press). However, increased C/N ratio has not been observed in the heathland experiments at Ruabon and Budworth (Caporn, *pers. com.*) or in a survey of UK acid grasslands (Stevens et al., 2006). It seems likely that the direction of change in C/N ratio induced by increased N deposition depends on the degree to which N limits plant growth in the system, with increases where litter production is stimulated and decreases where immobilisation into soil N is the more significant process. In either case, the rate of change in C/N ratio (and therefore the likelihood of detecting any change) will be smaller where soil stocks of these elements are large.

Nitrogen form is also potentially important and may have an effect on the extent to which N deposition impacts on a community. Plants that take N up directly through their leaves will be exposed to oxidised and reduced nitrogen in similar proportions to atmospheric deposits. Dry ammonia deposition is damaging to many plants and lichens even at low levels, as reflected in the critical level for ammonia of $1 \mu\text{g NH}_3 \text{ m}^{-3}$ (Cape et al., 2009). However, for most plants taking N from the soil, N forms may be strongly modified by chemical and biological transformations in the soil environment resulting in ratios of reduced to oxidised N which differ greatly from deposited inputs (Stevens et al., 2011b).

2.2 Types of evidence

Evidence for the impacts of nitrogen deposition on semi-natural habitats comes from a range of different approaches, encompassing experimental, survey and modelling studies.

There is a large body of experimental evidence ranging from small scale pot experiments to large field experiments. Nitrogen addition methods can be simple: many experiments apply N on a regular basis with a watering can or back-pack sprayer. More sophisticated methods for N addition are also used, e.g. at Whim Bog (see Table 1) wet deposition is applied as fine droplets and is controlled by wind speed and rainfall so that extra N is only applied when it rains, whereas dry deposition is applied as a gas plume creating a gradient of dry deposition (Sheppard et al., 2004). Rates of N application vary considerably between experiments going from very low levels up to several hundred $\text{kg N ha}^{-1} \text{ yr}^{-1}$. Applications have also been made for varying amounts of time (the Park Grass experiment which has received N additions for 150 years is the world's longest running ecosystem experiment), and with varying starting points in terms of background deposition. Evidence of recovery from N deposition in these types of experiments comes from cessation of additions at experimental sites (e.g. Edmondson et al., 2013) and from the use of roofs to exclude deposition when it rains and the re-application of purified rainwater (Emmett et al., 2004).

Surveys provide a complementary approach to understanding the effects of nitrogen deposition (Dise and Gundersen, 2004). These all make use of gradients of N deposition, but can vary considerably in their form, targeting specific vegetation types or regions. Some surveys are targeted at investigating the impacts of N deposition whereas others, such as the Countryside Survey, are not but can be used for this purpose (Smart et al., 2004). They can also take place over varying scales from local surveys targeting gradients created by increasing distance from a point source or larger scale surveys, up to international surveys such as the BEGIN project (Stevens et al., 2010a), which made use of variation in ambient levels of N deposition. A critical consideration in interpreting gradient studies is the ability to statistically account for co-varying factors such as climate or other pollutants. Resurveys (revisiting previous surveys to look at changes) have also been used to assess the impact of N deposition (summarised in RoTAP, 2012). Another approach is the use of volunteer-collected data such as national 10 x 10 km vegetation data collected by the Botanical Society for the British Isles to examine the relationship between species occurrence and N deposition (Henry et al., 2011).

The variables measured in experiments and surveys vary considerably and include impacts on plant and soil biogeochemistry such as soil pH, nitrogen contents, other nutrient, carbon and metal concentrations, and enzyme activity; changes to soil processes such as decomposition and mineralisation; floristic changes such as to species diversity and habitat composition; changes to plant physiology, productivity and phenology; and changes to the soil microbial community such as fungal to bacterial ratios, phospholipid production, mycorrhizal colonisation and microbial biomass. Most biological studies have focused on effects on plants, lichens and/or microbes, and the evidence base for effects of N on animals is small. However, plant diversity underpins the diversity of other taxa (by providing structural diversity and a diversity of substrates), and it is likely that the effects of N-stimulated productivity on ground-level shading is having effects on small invertebrate and vertebrate species. Effects of N on habitat suitability for butterflies have been observed (Wallisdeveries and Van Swaay, 2006). Few studies have been concerned with impacts higher up the food chain, although there is unpublished evidence from the Netherlands of impacts on bird species (red-backed shrike) due to changes in prey type, size and abundance, as a result of nitrogen-mediated habitat change.

Modelling provides a source of understanding that is parallel to empirical studies. The models used to predict changes in soil, vegetation and floristics resulting from N deposition (in combination with other factors) are based on theoretical understanding, and parameterised and tested against empirical data wherever possible.

2.3 Empirical evidence for biogeochemical responses to cumulative deposition

In this section we examine responses of the following biogeochemical measures: tissue chemistry of plants, stocks of total N in soil, measures of available N in soil, N losses through denitrification and leaching, and changes in soil chemistry such as pH and buffering capacity (Ca:Al ratios, cation exchange capacity).

Published evidence on biogeochemical change is available from long-running N experiments, e.g. those of the UKREATE consortium which were extensively reviewed in (Phoenix et al., 2012), summarised in Table 1, and from gradient or national surveys, many reviewed in RoTAP (2012).

Table 1. Description of the nine UKREATE long-running nitrogen manipulation experiments (adapted from Phoenix et al., 2012)

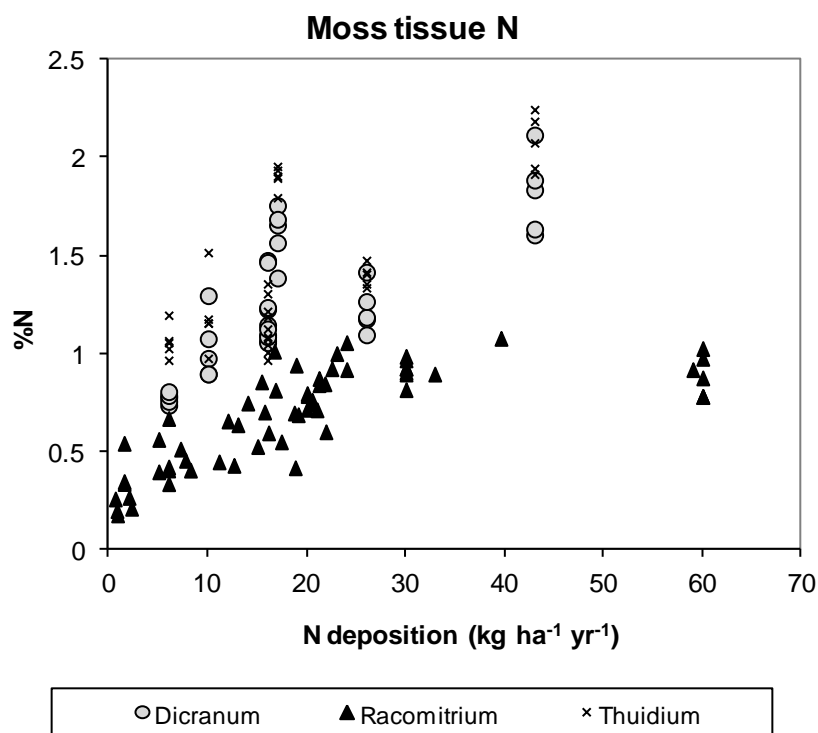
Habitat	Site name (Abbreviated code)	Vegetation type: NVC classification	N treatment rates (kg N ha ⁻¹ yr ⁻¹)	Other treatments	Duration of N treatments – yrs to date or until ceased ⁹	Modelled N dep. at site (kg N ha ⁻¹ yr ⁻¹)
Heath	Ruabon (RUH)	Upland heath: H12 <i>Calluna</i> – <i>Vaccinium</i>	0, 40, 80, 120; or 0, 10, 20, 40, 120	Controlled burn	22 or 13	25
	Budworth (BLH)	Lowland heath: H9 <i>Calluna</i> – <i>Deschampsia</i>	0,20,60,120	None	15	28
	Thursley (TLH)	Lowland heath: H2 <i>Calluna</i> - <i>Ulex minor</i>	0, 7.7, 15.4; or 0, 30	Uncontrolled burn, Controlled burn	7 or 13	20
	Culardoch (CAH)	Low Alpine Heath: H13 <i>Calluna</i> - <i>Cladonia</i>	0, 10, 20, 50	Clipping, Burning	11	11
Bog	Whim (WBO)	Ombrotrophic bog: M19, <i>Calluna</i> - <i>Eriophorum</i>	0, 8, 24, 56 for wet deposition; NH ₃ transect 4-70	None	9	10
Grassland	Pwllperian (PAG)	Upland acid grassland	0, 10, 20	Sheep grazing: Light, Heavy	15	17
	Wardlow acid grassland (WAG)	Acid grassland: U4e <i>Festuca</i> - <i>Agrostis</i> - <i>Galium</i>	0, 35, 70, 140; or 0, 35, 140	None	12 or 16	34
	Wardlow calcareous (WCG)	Calcareous grassland: CG2d <i>Festuca</i> – <i>Avenula</i>	0, 35, 70, 140; or 0, 35, 140	None	12 or 16	34
Sand dune	Newborough (NDG)	Fixed sand dune grassland: SD8 <i>Festuca</i> – <i>Galium</i>	0, 7.5, 15	Ungrazed; Rabbit grazed; Large Stock (ponies, cattle)	8	11

2.3.1 Plant tissue chemistry

Experimental evidence of changes in plant tissue chemistry with N deposition comes from the UKREATE experiments (Phoenix et al., 2012) and other studies. In the UKREATE experiments, there were increases in tissue N of either higher or lower plants in response to N deposition in all of the N manipulation experiments across all habitats. For the higher plants, tissue N response in heathlands (mainly tissue N of *Calluna vulgaris*) occurred at lower N loads (15-60 kg N ha⁻¹ yr⁻¹) than in

grasslands where tissue N changed mainly at higher loads (20-140 kg N ha⁻¹ yr⁻¹). Bryophytes were more sensitive than higher plants and showed changes in tissue N in most grassland and heath experiments and generally at lower N loads (15-140 kg N ha⁻¹ yr⁻¹). In coastal dunes, tissue N of a number of bryophyte species was correlated with N deposition (Jones et al., 2004). In bryophytes, the dose-response relationship becomes saturated at higher N loads e.g. Figure 3 (Jones, 2005; Lamers et al., 2000). There were also changes in tissue N/P ratios reflecting relative P limitation, with increasing N/P ratios with N additions at three of the heathland sites. Another gradient study of *Calluna vulgaris* tissue chemistry showed greater tissue N concentration with more N deposition, but an even greater proportional increase in tissue P concentration, presumably because N stimulated P uptake (Rowe et al., 2008). This suggests that N/P ratio is not a robust indicator of ecosystem responses to N deposition.

Figure 3. Moss tissue N plotted against current N deposition (kg N ha⁻¹ yr⁻¹) for three mosses *Dicranum scoparium*, *Racomitrium lanuginosum* and *Thuidium tamarascinum*. Data from: Jones (2005); Baddeley et al. (1994); Jonsdottir et al. (1995); Pearce & van der Wal (2002); Pearce et al. (2003); Leith et al. (2008); Armitage et al. (2012).



Plant tissue chemistry is important as it is potentially an indicator of luxury accumulation of N, i.e. where N supply is greater than immediate demand for growth, but not beyond the capacity of the plant to take up and store the N. In the short term the N may be stored in plant material rather than the soil N pool, but over timescales of a few years is returned to the soil via litter fall or decomposition and a proportion goes to increasing soil N stocks (although this will not necessarily cause a decrease in C/N ratio, if C stocks increase at a proportionally greater rate). The remainder is recycled into plant or microbial growth or lost from the system via leaching or gaseous emissions.

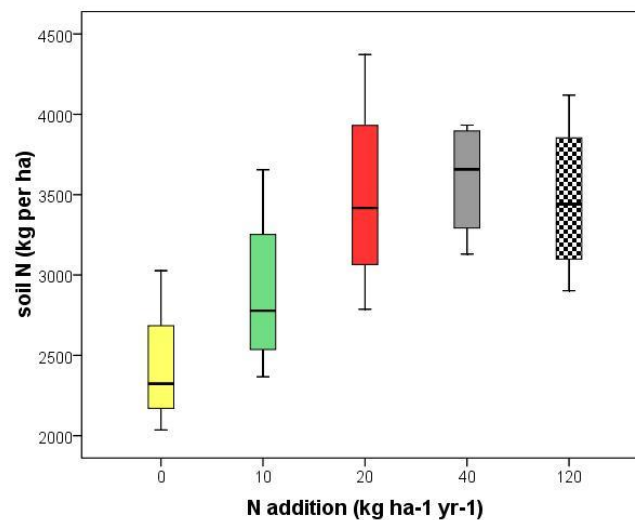
Changes in plant tissue chemistry also play a role in moderating the impact of secondary stresses, and changes in phenology. Secondary stresses can include increased sensitivity to pathogen attack, herbivory, frost or drought stress (e.g. Power et al., 1998; Sheppard et al., 2008; Strengbom et al., 2002). In the UKREATE experiments, secondary stress interactions were observed in all of the

heathland sites (drought, frost or pathogen attack), at moderate to high N deposition loads of 20-80 kg N ha⁻¹ yr⁻¹ (Phoenix et al., 2012).

2.3.2 Nitrogen stocks in soil

Nitrogen deposition increases N stocks in soil through absorption of ammonium ions onto exchange sites, through incorporation of additional organic matter as a result of enhanced above-ground production, and through a positive feedback whereby extra organic matter increases the NH₄⁺ sorption capacity. The N concentration in plant litter is likely to increase with N deposition, although it should be noted that recent plant litter usually has greater C/N ratio than older soil organic matter, and where litter production is stimulated this can cause an overall increase in total soil C/N as discussed in section 2.1. Experimental evidence from the UKREATE sites (Phoenix et al., 2012) showed that N additions increased total soil N pools at some of the heathland sites. For example, in the Ruabon site 'new plots' (on an upland heath with thick organic soils), significant increases in soil N stocks occurred with N application rates of 40 kg and above (Figure 4). There was no change in soil N stocks in the grassland or bog sites, even under very high loads of up to 140 kg N ha⁻¹ yr⁻¹. This is surprising given the proportion of N retained in those soils (see Section 2.3.4), but may relate to methodological difficulties in measuring N stocks in soils where structure and horizon thickness are changing.

Figure 4. Effects on soil N stocks of nitrogen deposition at rates of 10, 20, 40 and 120 kg N ha⁻¹ yr⁻¹ in the Ruabon site 'new plots'. Chris Field, *pers. com.*



2.3.3 Nitrogen availability & cycling

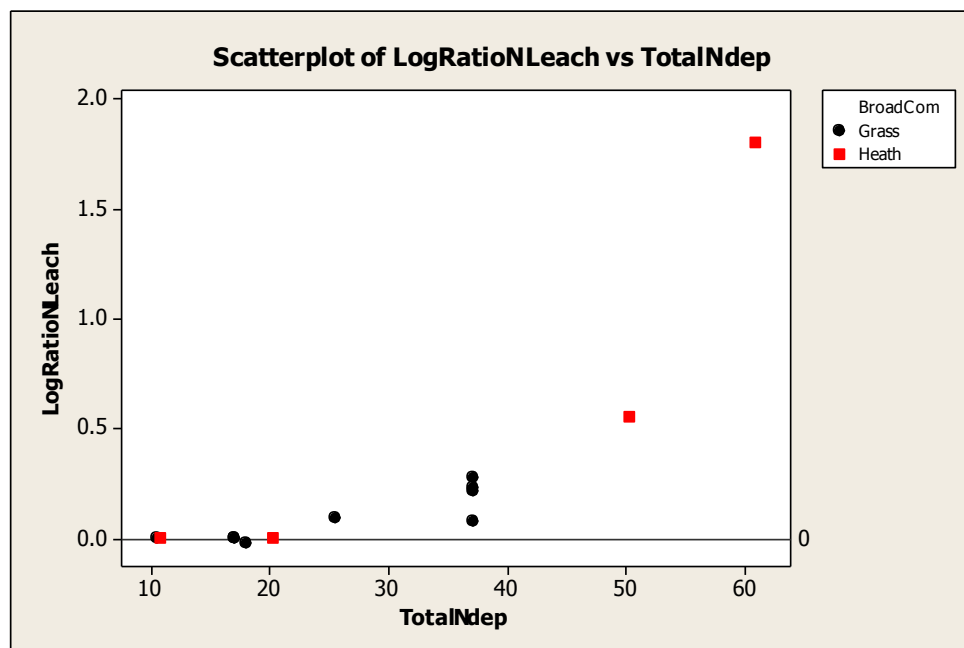
In the UKREATE experiments (Phoenix et al., 2012), mineral (i.e. immediately plant-available) N increased with N additions in heath and grassland sites, but generally only at high N loads (30-140 kg N ha⁻¹ yr⁻¹), with the exception of the montane heath site, where it increased with addition of only 10 kg N ha⁻¹ yr⁻¹. Heathlands appear to be more sensitive than grasslands, with increases in available N occurring at lower N loads. However, mineralisation rates increased at fewer sites, and not at all sites where available N increased. Where mineralisation rates did increase, this was usually at lower N loads. This suggests that observed increases in mineral N were not solely a result of increased mineralisation and may have been a result of temporary retention in the soil of deposited N which is in excess of immediate plant and microbial demand before being lost to leaching. The national Countryside Survey showed clear responses of mineralisable N to N deposition; greater deposition

fluxes were associated with increased stock of mineralisable N in organic soils, and with an increased proportion of nitrate in mineralisable N in mineral soils (Rowe et al., 2012).

2.3.4 Nitrogen losses – denitrification & leaching

Evidence for effects of N deposition on denitrification comes from seven of the nine UKREATE experimental sites. Of these, only two heathland sites showed significant increases in denitrification rates, and generally at high N loads ($> 56 \text{ kg N ha}^{-1} \text{ yr}^{-1}$). Losses from leaching showed more consistent increases in response to N in the UKREATE experiments. The majority of heath and grassland sites showed significant increases in NO_3 leaching, usually at moderate to high N loads ($20 - 140 \text{ kg N ha}^{-1} \text{ yr}^{-1}$). The maximum proportion of leaching was 35% of inputs at the highest N load ($140 \text{ kg N ha}^{-1} \text{ yr}^{-1}$) in acid grassland, so even at this load in a site which has received very high historical loads of N deposition and where denitrification was estimated to be negligible, the system was not fully N saturated. At all other sites, retention rates were higher. Rates of NH_4 leaching were generally very low and rarely showed significant responses to N additions. Unpublished analysis of data from the experiments suggests that the response of leaching to N is dose-related and is similar in heathlands and grasslands (Figure 5).

Figure 5. Leaching of N as a log-ratio of the control treatment, plotted against total N deposition (background + treatment, kg N) for nine UKREATE experiments (unpublished data).



Gradient studies have shown that nitrate leaching in forests is related to the C:N ratio of the litter layer or soil organic layer (MacDonald et al., 2002). The C:N ratio is also related to leaching rates in other habitats, but the critical C:N threshold at which leaching occurs differs by habitat (Pilkington et al., 2005; Rowe et al., 2006). Higher DON concentrations have been shown in dune groundwater along a gradient of N deposition (Jones et al., 2002) and the ICP Forest Inventory Level 2 monitoring plots also show elevated DON leaching correlated with N deposition (Vanguelova et al., 2010). However national surveys and forest monitoring plots did not show consistent elevated responses of inorganic N in soil water with N deposition. The link to N saturation is important since recovery experiments in forests and heathland have shown that in N-saturated sites, when current N loads are reduced, N leaching also reduces (Boxman et al., 1998). Therefore in a saturated system, current deposition usually exceeds biological demand and a large proportion of it is leached straight from

the system. The amount leached is that which is in excess of instantaneous demand (Lovett and Goodale, 2011).

2.3.5 Soil pH

The UKREATE experiments (Phoenix et al., 2012) showed both increases and decreases in pH due to N additions, linked to the form in which N was added. At the bog site, pH increased in the dry ammonia treatment, and also in the wet oxidised N treatment where N was added as NaNO_3 , where addition of the cation Na^+ caused the pH increase. Some pH decreases in treatments using $(\text{NH}_4)_2\text{SO}_4$ were due to addition of the mobile sulphate anion. However, there were also pH decreases when N was added as NH_4NO_3 , in the upland heath and in both acidic and calcareous grassland sites at high N deposition loads ($>70 \text{ kg N ha}^{-1} \text{ yr}^{-1}$). This acidifying effect of N is caused by the replacement of soil base cations, which are leached together with NO_3^- as counter-ions, by H^+ ions. Most national surveys reporting changes in pH (e.g. Countryside Survey) reflect changes due to historical pollutant (mainly sulphur) deposition and show slight increases in soil pH in the last decade. However some gradient studies, have separately attributed acidification effects due to N to show that acidification is potentially the predominant driver of vegetation change driven by N deposition in sensitive acid grasslands (Stevens et al., 2010b). Due to the dramatic decline in sulphur emissions and deposition since its peak in the 1970s, N has taken over as the main acidifying pollutant in the UK (RoTAP 2012).

2.3.6 Soil BC:Al ratio and other measures of buffering capacity

The UKREATE experiments also showed decreases in BC:Al ratios or in Cation Exchange Capacity due to loss of base cations through leaching at heathland and acid and calcareous grassland sites at low-high N deposition loads of $20\text{-}140 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ (Phoenix et al., 2012).

2.4 Empirical evidence for floristic responses to cumulative deposition

There is an increasingly strong body of evidence demonstrating negative impacts on N deposition on plant communities. Nitrogen deposition affects vegetation directly through foliar toxicity and indirectly through eutrophication, acidification and susceptibility to secondary stress. Species most sensitive to N deposition are those that are adapted to low-nutrient conditions or are intolerant of acidification.

Plant species are major components of semi-natural habitats, contributing to the maintenance of habitat structure and function. Experimental studies have shown decreases in plant and lichen species abundance but have rarely resulted in the complete loss of species, except in the case of certain lichens and bryophytes (Phoenix et al., 2012). This is likely due to the previous loss of sensitive species from experimental sites because they have already been exposed to ambient levels of N deposition for many years. Survey evidence, which represents long-term effects of N exposure, clearly shows that declines in prevalence of many species are related to spatial patterns of N deposition (Emmett et al., 2011; Stevens et al., 2011c). These declines continued even at large rates of N deposition, implying that even areas where the sensitive flora has been absent for many years are still losing species because of N deposition. The schema for biodiversity loss presented by (Emmett, 2007) (Figure 2) therefore under-represents the continuing loss of N-sensitive species at advanced stages of N saturation.

National and regional scale survey data show clearly that N-sensitive plant and lichen species may be lost from habitats at N deposition rates less than the critical load (e.g. Dupre et al., 2010; Payne et al., 2013; Stevens et al., 2011c). The evidence base for this is stronger for some habitats than others, for example, there have been a number of experimental and gradient studies in acid grasslands (e.g.

Phoenix et al., 2012; Stevens et al., 2004), calcareous grasslands (e.g. Phoenix et al., 2012; van den Berg et al., 2011) sand dune grasslands (Jones et al., 2004; Plassmann et al., 2009) and heathlands (e.g. Britton and Fisher, 2007; Edmondson et al., 2010) but very little research in other habitats such as coastal vegetated shingle.

Among the clearest evidence for floristic change comes from national and international surveys showing clear declines in species richness as N deposition increases along spatial gradients. The picture that emerges is one of progressive loss of plant and lichen species with increasing N accumulation, with the most rapid loss of species in the earlier stages of N enrichment (Caporn et al., 2011). This evidence has been collated from a range of habitats including acid grasslands (Stevens et al., 2010b), calcareous grasslands (van den Berg et al., 2011), heathlands (Britton and Fisher, 2007; Edmondson et al., 2010), bogs (Caporn et al., 2011) and sand dune grassland (Jones et al., 2004; Plassmann et al., 2009), with many surveys showing similar declines in species richness with increasing N deposition. Species richness is a metric that should be interpreted with caution because it can mask large changes in species composition and, with the invasion of eutrophilic species, species richness could increase. Studies have also shown negative impacts on species composition, diversity and evenness (e.g. Stevens et al., 2006). Examination of individual plant functional groups tends to show declines in the richness or cover of forbs and bryophytes whereas grasses increase in their cover (e.g. Dupre et al., 2010). In surveys of Scottish montane and *Racomitrium* heaths decreases in the species richness of some functional groups were balanced by increases in other species (RoTAP, 2012). Few experimental studies have shown declines in species richness but rather show declines in the cover of individual species (e.g. Phoenix et al., 2012). This is compatible with the findings of surveys where the consequence of reduced cover of species is a reduced probability of finding a given species in a randomly-placed quadrat.

Many experimental and survey studies have shown changes in species composition in response to increased N inputs. Impacts on individual species are variable with some species benefiting from N addition whilst others decline. *Succisa pratensis* (devil's-bit scabious) and *Drosera rotundifolia* (round-leaved sundew) are examples of species that have been shown to be negatively impacted by N addition in experimental studies. Nitrogen addition experiments have shown declines in *S. pratensis* with N addition (e.g. Vergeer et al., 2003) and this species shows a strong negative response to increasing NH_4^+ concentrations in soil solution (van den Berg et al., 2005). *Drosera rotundifolia* has also shown strong declines in occurrence with N addition (Redbo-Torstensson, 1994) and has been shown to shift away from utilising prey-derived N under high N conditions (Millet et al., 2012). In a national study using large-scale species occurrence data, Henrys et al. (2011) were able to identify a number of vascular plant species sensitive to N deposition. Impacts on species may also be mediated by interactions with other stress factors. This has been seen in the Netherlands where the decline in *Calluna vulgaris* (heather) is associated with an increased susceptibility to attack from the heather beetle (*Lochmaea suturalis*) (Heil and Diemont, 1983). However, despite the widespread changes in species composition observed in relations to N deposition there has been very little work on rates of extinction or introgression. At the Park Grass experiment, established in 1856, detailed species survey has revealed that responses to fertilizer addition (N is applied as ammonium sulphate at rates of 48, 96 and 144 kg N ha⁻¹ yr⁻¹ and sodium nitrate at rates of 48 and 96 kg N ha⁻¹ yr⁻¹) may occur rapidly, within the first few years of treatment at such high levels of N addition but the community did not reach equilibrium for 40 years. In subplots where N additions were ceased, the extent of community change depended on soil pH with little change in the community where soil acidity was uncorrected (Silvertown et al., 2006). In order to understand the cumulative impacts of N deposition on semi-natural communities further information is needed on rates of vegetation change.

Lower plants can be particularly sensitive to N deposition. Declines in bryophyte species in response to N deposition have been observed including *Sphagnum* species (e.g. Limpens and Berendse, 2003)

and *Racomitrium lanuginosum* (van der Wal et al., 2003) and some lichen species are particularly sensitive to N deposition (e.g. Mitchell et al., 2005; Stevens et al., 2012b).

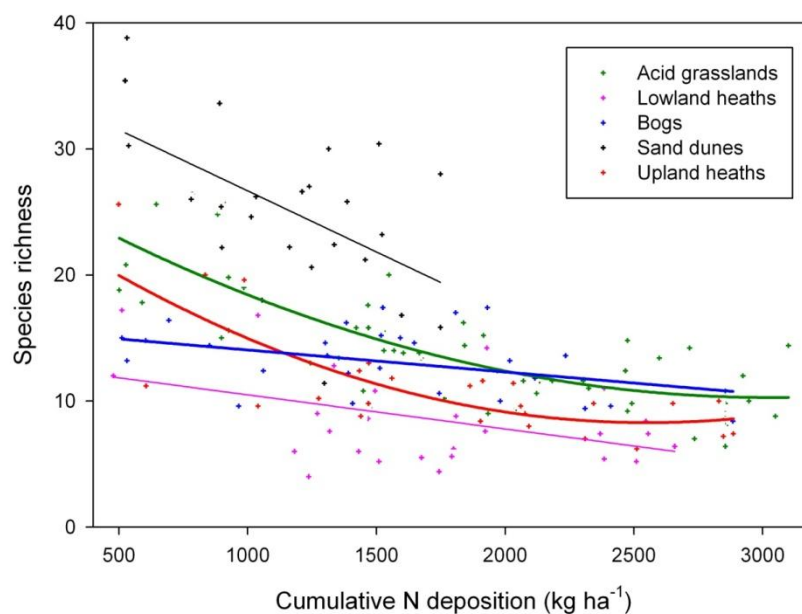
Caporn et al. (2011) investigated the impact of small incremental changes in N deposition above the critical load on floristic change. They compiled data from 226 sites, collected over 8 surveys of 5 UK priority habitats for conservation (sand dune, bog, lowland heath, upland heath, acid grassland). They found that across the habitats and datasets, increasing N deposition was correlated with declines in species richness and changes in species composition. They concluded that there were losses in diversity from well below the habitat-specific critical load range for individual habitats and these losses are sharper at low levels of deposition. At levels of N deposition at and above the habitat-specific critical load, additional increments of long-term N input were associated with further declines in species richness. This was in agreement with Emmett et al. (2011) who found that both changes in both the presence of individual species and ecosystem function indices (canopy height, specific leaf area and Ellenberg N) occur at low thresholds of N deposition (5-10 kg N ha⁻¹ yr⁻¹). Changes in both species occurrence and ecosystem function indices continued above critical load values indicating that ongoing damage occurs above the critical load threshold.

Rates of vegetation change in response to N addition can only be easily established from experimental studies and differ greatly between habitats making it very difficult to identify a specific rate of change. Results from the UKREATE experiments (summarised by Phoenix et al., 2012) have shown relatively minor changes in vascular plant species composition from around 6 years of N addition in some experiments, but with few changes even after 22 years of N addition at high rates in other experiments. However, the latter experiment is located in an area of high ambient N deposition so changes may well have occurred before the experiment and on a managed moorland the cycle of change is periodically reset. It should be noted that several of these experiments are located in sites with high background N deposition where species may already have been lost, and heathland sites where vascular-plant diversity would not be expected to be high. Lichens and bryophytes were generally more sensitive and showed responses in several experiments after between three and 22 years. At Whim bog after nine years of N addition several species increased in cover whilst *Calluna vulgaris* has declined in cover under dry NH₃ deposition. In the grazed acid grasslands at Pwllpeiran only small changes in vascular plant composition were observed after fifteen years and in the dune grassland experiment at Newborough only one species, *Luzula campestris* (wood rush) showed a response after eight years of N addition. At Wardlow Hay Cop acid and calcareous grassland experiments, changes in species composition have been observed after six years for vascular plants and large changes in bryophyte cover were observed, up to 84 % loss of *Rhytidiadelphus squarrosus* with N additions of 25 kg N ha⁻¹ yr⁻¹. A number of recent papers have pointed towards the importance of the accumulated dose response in determining the rate of N deposition impact on vegetation. Considering the accumulated dose response allows both the rate of N input and the duration of N input to be incorporated into a single measure. Phoenix et al. (2012) found clear and significant declines in lower plant cover in the UKREATE experimental sites in response to accumulated N inputs. Dupre et al. (2010) also used deposition when examining historic data from acid grasslands and found clear relationships between accumulated N deposition and species richness and Ellenberg N values, whilst Payne et al. (2013) successfully used vegetation composition to predict cumulative deposition in acid grasslands and heathlands.

Phoenix et al. (2012) investigated relationships of a few soil parameters using the metric of cumulative deposition since 1945. A similar approach was used to develop Figure 6, which shows a decline in species richness with cumulative deposition, in several semi-natural habitats. In this case cumulative N deposition was estimated from current deposition patterns and plotted against a percentage of the maximum species richness found for each habitat type. It shows similar responses across ecosystem types, with the exception of bogs where hydrology appears to buffer N-driven

vegetation change. However, each plant group responds in a different way and each ecosystem is dominated by each group to a different degree. Therefore, habitats that consist of a significant component of sensitive species such as bryophytes and lichens may show a stronger (and more rapid) relationship with current modelled deposition than with cumulative deposition. Any response is likely to be mediated by both management interaction that removes the canopy of a dominant species allowing light to reach lower plants, and the effect that long-term cumulative N-addition has on seed banks and other sources of recovery for sensitive species.

Figure 6. Plant species richness vs cumulative Nitrogen deposition since 1900. Plant data was collected in the multi-habitat survey across varied UK habitats in summer 2009 (lowland heaths, bogs, sand dunes) (Caporn et al., 2010) or, for acid grassland, provided by Carly Stevens as a subset of the data reported in Stevens et al (2004). Nitrogen data is based on the scaling factors from Fowler et al. (2004) and using individual-year CEH models for the recent past. Curvilinear relationships are plotted when there was a significant improvement in fit compared with linear. (R. Payne and C. Field, unpublished).



2.5 Recovery responses to decreases in N deposition.

The consequences of cessation of N additions and decreases in N deposition are much less well understood than the impacts of N addition. The few studies that have investigated recovery from N addition report slow effects in both vegetation species composition and nutrient cycling, even many years after the cessation of N inputs (Clark et al., 2009; Královec et al., 2009; Mountford et al., 1996; Olf and Bakker, 1991; Power et al., 2006; Strengbom et al., 2001). However, bryophytes and lichens which receive the majority of their N directly from deposition may respond to decreases in direct deposition more quickly (Armitage et al., 2011; Mitchell et al., 2004; Power et al., 2006). For example, Armitage et al. (2011) showed partial recovery of moss chemistry, moss mat thickness and community composition after only 2 years of lowered N deposition. In heathlands, Power et al. (2006) found that after seven years of N addition at rates of 7.7 and 15.4 kg N ha⁻¹ yr⁻¹ they still had a large number of statistically significant effects of the N addition on higher plant growth. Nitrogen additions were stopped in 1996 but residual effects of these treatments continued to cause significant differences in canopy height in formerly N treated plots in some years between 1998 and 2003. After eight years, there ceased to be a significant difference between N treated plots and control plots in 2004. The frequency of occurrence of bryophytes was also impacted by former N treatments and despite considerable inter-annual variation significant treatment effects were still

observable in 1998, two years after cessation of N addition. Other effects on lichen frequency, bud burst and flowering of *Calluna vulgaris*, and drought injury, were observed up to eight years after N additions had ceased.

In another heathland experiment on heather moorland at Ruabon, studies were made of the effect of stopping N additions following five years of inputs (Edmondson et al., 2013). After two years, stopping N additions (in the highest treatment of 120 kg N ha⁻¹ yr⁻¹ in addition to background deposition) resulted in significantly shorter shoot extension. Seven years after cessation there were significant declines in litter total N concentration and extractable mineral N and an increase in litter C:N ratio. However, the lichens and bryophytes, which had previously declined in response to additions did not show a recovery.

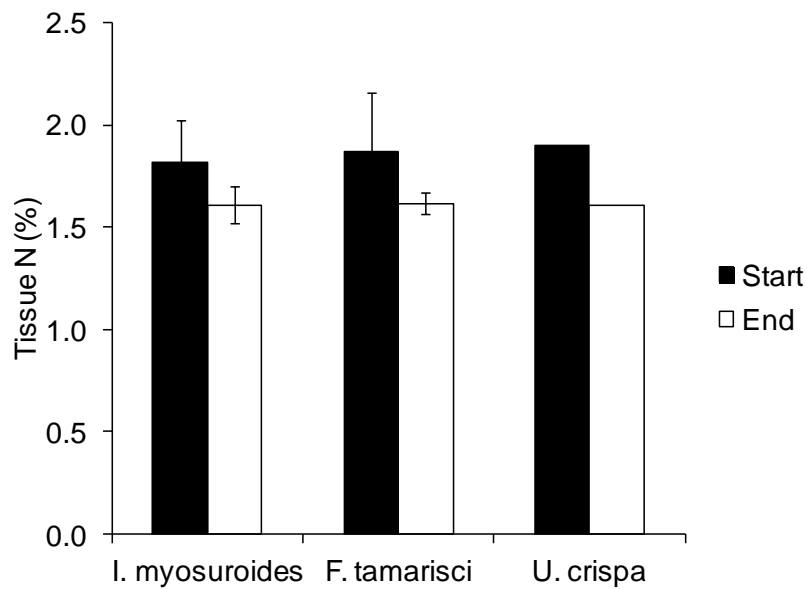
In a hay meadow Stevens et al. (2012a) found that fifteen years after N additions at rates on 25, 50, 100 and 200 kg N ha⁻¹ yr⁻¹ had stopped (following eight years of addition) mean Ellenberg N values were significantly higher than the control in the N50, N100 and N200 treatments. Ellenberg N values in the N25 treatment were not significantly different from the control. Initial findings of Mountford et al. (1996) were that five years after the cessation of N additions negative trends associated with N application were still apparent. Stevens et al. (2012a) question whether the plant community in plots which have received higher levels of N additions will ever revert to a similar species composition to that found in control plots. The reason for this is because species that became dominant in response to N addition such as *Holcus lanatus*, *Rumex acetosa* and *Lolium perenne* previously occurred at lower frequencies in the grass sward. Given the lack of regeneration niches less competitive species are likely to find few opportunities for population growth, unless active management such as turf stripping or top soil removal is carried out to reduce competition. In contrast Kráľovec et al. (2009) found a relatively rapid recovery in species richness in a previously fertilised meadow (80, 160, 240 and 320 kg N ha⁻¹ yr⁻¹ for 21 years) however, differences were still apparent after 18 years and little increase in the species richness of the highest N treatment relative to the control plots was observed after 1996 despite the small plots used in this experiment providing many seed sources. In a boreal forest Strengbom et al. (2001) reported very slow recovery in the species composition of ground flora. In one experiment nine years after the cessation of N additions (34, 68, 108 kg N ha⁻¹ yr⁻¹ for 14 years) species composition showed large differences compared to the controls and in another, 47 years after the cessation of N additions (total N addition of 1447 kg ha⁻¹ over 28 years) bryophytes favoured by high N levels were still more abundant than in control areas.

In a recovery experiment using mesocosms from acid and calcareous grasslands, Jones (2005) showed rapid rates of recovery of some bryophytes and lichens. In the acid grassland mesocosms, *Racomitrium lanuginosum* showed increases in cover in the lowest decrease treatment (pristine = 4 kg N ha⁻¹ yr⁻¹) relative to the ambient control (24 kg N ha⁻¹ yr⁻¹) after only one year of recovery. Growth responses of other moss and lichen species (*Rhytidiadelphus loreus*, *Hypnum jutlandicum*, *Cladonia furcata*) revealed a recovery response to optimum N deposition loads which lay below the ambient deposition at the source site within four years. In general, the lower the optimum N level for a species, the faster the rate of recovery, with the exception of the moss *Polytrichum juniperinum*, which has a low optimum but was the slowest of the acidic grassland species to respond. This species is endohydric i.e. has some internal conducting tissue, and may obtain some N from the soil in addition to atmospheric deposition. In the calcareous grassland, the dominant moss species (*Rhytidiadelphus squarrosus* and *Pseudoscleropodium purum*) had optimum N levels at or above ambient deposition, but did not decline in response to the recovery treatments.

In a reciprocal transplant experiment with epiphytic bryophytes, Mitchell et al. (2004) transplanted three species of bryophytes (*Isothecium myosuroides*, *Frullania tamarisci* and *Ulota crispa*) from oak woodland receiving a modelled atmospheric deposition of 54 kg N ha⁻¹ yr⁻¹ to a pristine oak

woodland receiving $12 \text{ kg N ha}^{-1} \text{ yr}^{-1}$. Within 12 months, tissue N decreased and growth increased in *Frullania tamarisci* following a decrease in atmospheric N deposition, with similar but non-significant patterns in the other species (Figure 7). This suggests that epiphytic bryophytes respond rapidly (within a couple of years) to decreases in atmospheric N deposition.

Figure 7. Tissue nitrogen concentration in three bryophyte species at the start and end (after 12 months) of a transplant experiment where bryophytes were moved from a site receiving $54 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ to a site receiving $12 \text{ kg N ha}^{-1} \text{ yr}^{-1}$. Means ± 1 standard error are shown. For *U. crista*, samples had to be pooled to provided enough material, so no standard error bars are presented. Data taken from Mitchell et al. (2004).



The implication of variable recovery rates of different components of ecosystems (e.g. epiphytes, terrestrial mosses, rooting plants) is that different dose-response relationships may be necessary to represent recovery.

Once species composition is changed it appears that this is hard to reverse, although responses differ for lower plants and for higher plants. Evidence from recovery experiments in a wide range of habitats show very slow return to original species composition of higher plants once N additions are ceased. Working in a heathland, Power et al. (2006) found that after eight years the plant community composition, total soil nitrogen and soil microbial biomass had not recovered from seven years of N addition at a rate of $7.7 \text{ kg N ha}^{-1} \text{ yr}^{-1}$; and Strengbom et al. (2001) reported similar findings for species composition after 19 years of recovery in a boreal forest which had received 34, 68, 108 $\text{kg N ha}^{-1} \text{ yr}^{-1}$ for 28 years. Stevens et al. (2012a) reported only small recoveries in Ellenberg N values and plant biomass after 15 years of recovery from 25, 50, 100 and 200 $\text{kg N ha}^{-1} \text{ yr}^{-1}$ for eight years. They also found that significant differences remained in total soil N concentration between control plots and plots which had received nitrogen additions fifteen years ago, even at the lowest application rate of $25 \text{ kg N ha}^{-1} \text{ yr}^{-1}$.

Responses to reductions in N deposition observed in experimental studies are summarised in Table 2.

Table 2. Summary of observations in experiments where nitrogen deposition rate was decreased.

Reference	Habitat type	Years since decrease	Recovery observed
(Armitage et al., 2011)	alpine moss-sedge heath	2	Partial decrease in moss tissue N
(Clark et al., 2009)	grassland	12	Partial decrease in N mineralisation rate
(Edmondson et al., 2013)	Heathland	2	Decreased <i>Calluna</i> shoot extension
“	“	7	Decreased litter N, extractable N, litter C/N. No recovery of bryophytes and lichens.
(Jones, 2005)	Acid & calcareous grasslands	1	Some recovery of <i>Racomitrium</i> cover
“	“	4	Some recovery of cover of other moss and lichen species.
(Královec et al., 2009)	Meadow	18	Some differences still apparent
(Mitchell et al., 2004)	Oak woodland epiphytes	1	Decreased tissue N in one species
(Mountford et al., 1996)	Grassland	3-9	Recovery of cover of individual species (period depends on previous N application rate)
(Olf and Bakker, 1991)	Mown grassland	14	No decrease in productivity on sandy soil; decrease in productivity and increased species-richness on peat soil
(Power et al., 2006)	Heathland	8	Canopy height recovered. Differences remain in e.g. lichen frequency, total soil N.
(Stevens et al., 2012a)	Hay meadow	15	Recovery of mean Ellenberg N only in previous 25 kg N ha ⁻¹ yr ⁻¹ treatment; dominant species persist.
(Strengbom et al., 2001)	Boreal forest ground flora	19	Differences in species composition still apparent
“	“	47	Bryophytes not recovered

2.6 Effects of marginal (small) changes in N deposition

2.6.1 Effects of small increases in N deposition

Effects of marginal increases in N deposition can be quantified from dose-response relationships developed using a realistic range of deposition for the UK situation. This approach was developed by Caporn et al. (2011) using data for species richness from national survey of five UK habitats.

Published evidence on the botanical composition of peatland following long-term N addition experiments, (e.g. UKREATE) has shown a highly significant pattern of species richness decrease as a function of increasing atmospheric nitrogen deposition with a more rapid loss of species associated with increasing N deposition at lower levels of N pollution (i.e. often below the habitat specific nitrogen critical load (5-10 kg ha⁻¹yr⁻¹ for bogs and 10-20 kg ha⁻¹yr⁻¹ for heaths) (Caporn et al. 2012).

Whilst the evidence for assessing the effects of marginal changes in N deposition is limited to a few selected manipulation studies in the UK, larger scale assessments of the response of plant species and habitats to a wide gradient of N inputs can be assessed using information collated at larger spatial scales (Harmens et al., 2011). Effects of N deposition on the prevalence of a large number of species within several habitats were demonstrated in Stevens et al. (2011c). Relationships established in that study could be used to assess likely effects of marginal increases in N deposition.

2.6.2 Evidence from ecosystems with low N deposition rates

Ecosystems remote from emission sources generally receive low inputs of N deposition and as such are sensitive to environmental change and respond rapidly to perturbations, despite the relatively small changes in deposition over time. Alpine and sub-alpine systems are particularly sensitive to changes in deposition loadings due to the temperature-dependent biological processes which control N cycling (Nadelhoffer et al., 1991). Alpine and sub-alpine ecosystems have therefore evolved under conditions of low N availability, and the deposition of anthropogenically-derived N in precipitation has the potential to have a large impact on nutrient limitation, interspecific competition and, consequently, vegetation composition (Britton et al., 2008; Nadelhoffer et al., 1992).

Our knowledge of the effects of low background N deposition on biogeochemical processes, biodiversity and ecosystem services is limited to a few long-term studies in alpine and sub-alpine ecosystems. Ecosystems that have been exposed to low doses of N deposition in the UK are generally spatially limited to remote upland areas.

A whole system experiment at the Culardoch research platform in the Cairngorms mountains was established to investigate the response of vegetation structure and function, and soil solution quality, to low (10, 20 and 50 kg N ha⁻¹yr⁻¹) doses of N; the highest dose is considered relatively modest compared to other manipulation experiments in the UK. The sensitivity of lichens, which are an important part of the *Calluna-Cladonia* montane heath at the site, to N deposition loads in excess of 20 kg ha⁻¹yr⁻¹ resulted in a clear decline in the cover area (Caporn et al., 2011). The response of the soil solution to the high N treatment (50 kg ha⁻¹yr⁻¹) demonstrated an immediate leakage of NO₃⁻ and NH₄⁺ to soil solution. It was hypothesized that this response resulted from the relatively large, discrete application of N having a toxic impact on soil microbes which limited N immobilization (Helliwell et al., 2010). In contrast, the soil solution NO₃⁻ and NH₄⁺ concentrations increased gradually following the application of the 10 kg ha⁻¹yr⁻¹ treatment and exhibited a clear seasonal trend as a result of biogeochemical processing (Helliwell et al., in prep.). This study demonstrated that increases in the nitrogen deposition flux (>20 kg ha⁻¹yr⁻¹) to sub-alpine ecosystems have a rapid effect on the chemical composition and nutrient status of alpine heathland vegetation and soils.

Bryophytes such as *Racomitrium lanuginosum* were shown to be sensitive to small changes in N deposition (Pearce and van der Wal, 2002). Increasing N deposition to *Racomitrium lanuginosum* has been recognised as being partially responsible for its decline. Pearce et al 2002 & 2003 identified that *Racomitrium* tissue N increased after treatment, with a greater response to low (10 kg ha⁻¹yr⁻¹) rather than high (40 kg ha⁻¹yr⁻¹) N addition. The activity of the enzyme nitrate reductase and *Racomitrium* growth were severely inhibited by increasing N addition. Furthermore, the physiological responses of *Racomitrium* to N indicate that the habitat's critical load (CL) is exceeded

by addition of 10 kg N ha⁻¹ yr⁻¹ (Pearce et al., 2003). In low deposition areas dominated by *Racomitrium lanuginosum*, nitrate reductase activity (NRA) was also correlated with shoot growth and this was identified as a useful biological indicator of moss condition.

The physiological characteristics of nonvascular plants such as liverworts and lichens, makes them highly sensitive to subtle fluctuations in N deposition, and important indicator species (Fenn et al., 2003). Nonvascular plants responded to extremely low deposition loads, for instance, <3 kg ha⁻¹ yr⁻¹ for epiphytic lichens in the Sierra Nevada, California (Fenn et al., 2003). This was also a clear outcome of the synthesis paper by Caporn et al. (2011).

2.6.3 Effects of small decreases in N deposition

Marginal changes need to be calculated separately for increases in N deposition than for responses in relation to recovery from declines in N deposition due to hysteresis effects. There is considerable data from a number of surveys and experiments in the UK to inform the former, although actual dose-response relationships have been developed for only a few parameters so far such as species richness (e.g. Stevens et al., 2004). However, dose-response relationships based on recovery from N deposition have a much smaller pool of experimental data on which to draw and there have been no attempts to synthesise this information yet for any parameter. Of the field recovery experiments described in Section 2.5, only one heathland experiment contained a treatment where an application of less than 10 kg N ha⁻¹ yr⁻¹ was stopped (Power et al., 2006).

As with marginal increases, relationships between current deposition rate and species occurrence established in Emmett et al. (2011) could be used to assess likely effects of marginal decreases in N deposition. However, the evidence base for effects of decrease is lacking, in particular regarding the delays to re-establishment of species. It seems likely that recovery will mainly be observed in receptors which are able to tolerate a higher load of N, and which have shorter lag recovery times.

2.7 Biogeochemical models

Models provide an alternative evidence base for understanding the impacts of N deposition on semi-natural habitats. Models are based on empirical observations and theoretical considerations, and are considered more reliable if they have been verified against independent observations. Since measurements are rarely sufficient to fully set up and test models, and because of uncertainties and debates about which theoretical considerations should be included, model outputs are uncertain. However, this must be balanced against uncertainty in empirical observations (due to sampling error or lack of measured data) and relationships established from these observations (due to choice of statistical model). Dynamic soil-vegetation model chains have the advantage of allowing ideas and scenarios to be explored with a relatively small investment of resources.

Biogeochemical models describe the fluxes and transformations of chemical elements such as N and carbon within ecosystems. Models used to explore air pollution impacts were initially focused on acid-base chemistry, and modelled soil organic matter using a simple saturation model (Aber et al., 1998) in which N leaching was governed by soil total C/N ratio (e.g. MAGIC, (Cosby et al., 1985); VSD (Posch and Reinds, 2009)). Increasingly processes such as differential turnover rates of different components of soil organic matter, and stimulation of litter production by N additions, have been recognised as important. The latest generation of N impacts models include these mechanisms and are likely to be more accurate in predicting the cumulative and long-term effects of N addition (e.g. N14C, (Tipping et al., 2012); VSD+ (Bonten et al., 2010); FORSAFE (Wallman et al., 2005)). The N14C model has been combined with models of acidification and organic matter dissolution to form

MADOC (Rowe et al., submitted), which can therefore predict nutrient-N availability and pH change even on very organic soils.

Soil N compounds have different timescales of availability. Small, soluble ions and molecules such as ammonium, nitrate and oligopeptides are in principle immediately available to plants, lichens and soil microorganisms, although species vary in the N forms they can process, and uptake also depends on organisms having access to these soluble N compounds before they are leached. These soluble compounds are often referred to as plant-available N, and because of their rapid turnover they can be seen as either a pool (measured e.g. in kg N ha^{-1}) or a flux (measured e.g. in $\text{kg N ha}^{-1} \text{ yr}^{-1}$). The mineral forms of soluble N, in particular positively-charged ammonium ions, can be held electrostatically on exchange sites on clays and organic matter. However, these adsorbed ions are in a dynamic equilibrium with dissolved ions and will be released if the solution is depleted by plant uptake or leaching, so can be seen as part of the plant-available pool.

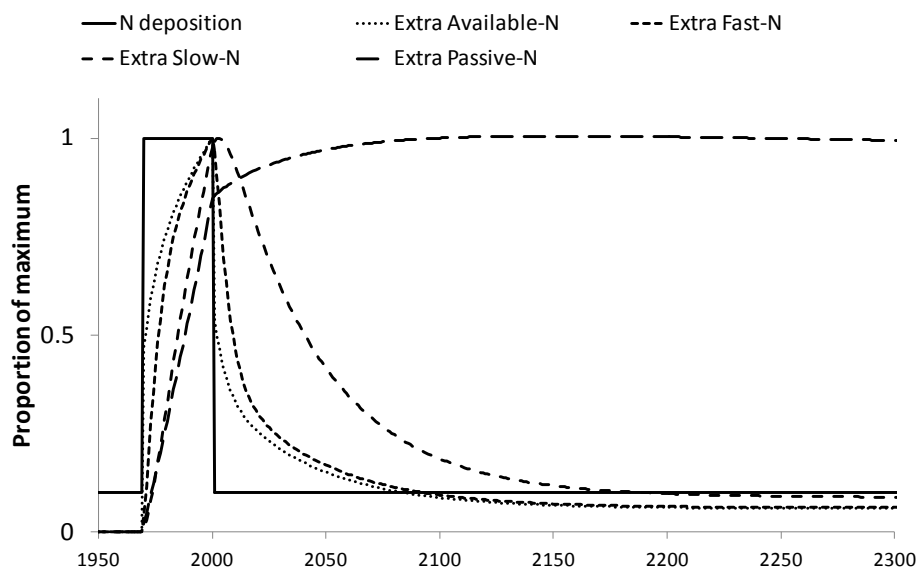
Most of the N in soil cannot readily be taken up by plants and other organisms since it is bound up in larger molecules and/or strongly bound to clay mineral particles. Before it can be used by organisms, this N must be converted to soluble forms in a process of decomposition that is mainly mediated by extracellular enzymes. This process is often referred to as mineralisation, although it is now understood that decomposition does not necessarily proceed as far as mineral N and that many plants and microbes can use small organic molecules as sources of N. Simple proteins that are relatively exposed to enzyme attack can decompose to give soluble N fairly rapidly. The N in complex insoluble molecules, or that is protected within mineral particles, is resistant to enzyme attack and can persist for many years. Between these extremes there is a continuum of availability, but this is commonly represented in soil N models in terms of discrete pools and fluxes. For example, the N14C model uses pools with mean residence times at 10 °C of 2 years ('fast'), 20 years ('slow') and 1000 years ('passive'). The effects of these pools on organisms is summarised in Table 3. Residence times increase at lower temperatures, in dry soils, and in particular under anaerobic conditions, leading to the formation of peats and organomineral soils where waterlogging occurs.

Table 3. Pools and fluxes of N used within an example soil organic matter model (N14C), their timescales of availability, principal factors controlling their magnitude, and likely effects.

Pool or flux	Timescale of availability	Controls	Effects
Plant-available N	immediate	Current deposition plus release from soil organic matter pools, in particular the 'fast' pool.	Can have immediate and acute effects on plant growth, the competition among species, and habitat suitability for key species. Effects more likely where N currently limits growth.
'Fast' N	~ 2 years	Litter formation (amount and C/N ratio) and decomposition.	Likely to affect habitat suitability for key species in the near future, but comparatively short-lived effects.
'Slow N'	~ 20 years	Residues after decomposition of more-labile N pools, and protection of soil organic matter through chemical complexity and mineral-association.	Likely to affect habitat suitability for key species in the medium term.
'Passive' N	~ 1000 years	As above.	Minor effects on key species, but likely to cause a very long-term increase in plant-available N, and/or an increased risk of acute N release if decomposition increases e.g. through drainage or temperature increase.

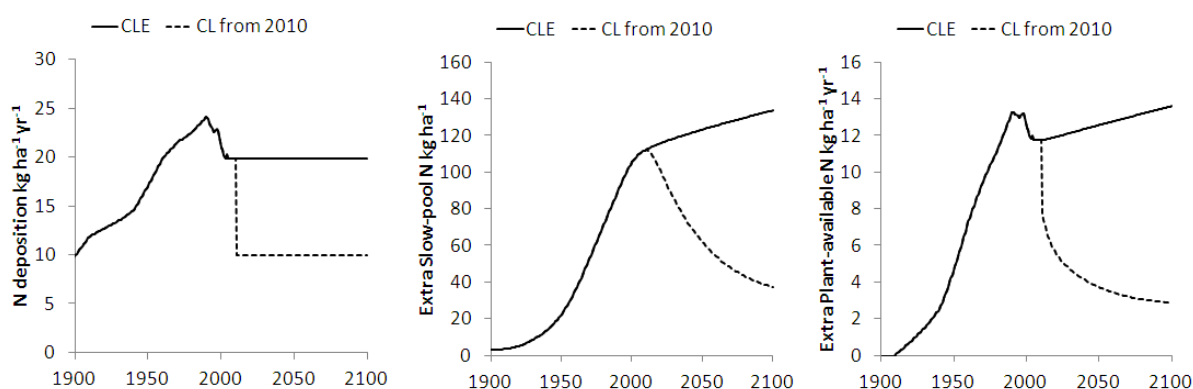
Although models evolve over time and are therefore subject to the criticism that current models will always be proved inaccurate, biogeochemical models may be useful for summarising and communicating the short-term and long-term effects of N deposition. For example, simulated responses to an artificial trajectory of N deposition make clear the different timescales with which different components of the system react (Figure 8). In this scenario, N inputs are abruptly increased, maintained at a high level for 30 years and then abruptly decreased. Extra plant-available N increases and decreases fairly abruptly in response, for example declining to half its maximum value two years after the decrease in N deposition, although there are ongoing changes due to release of plant-available N from other soil pools. The 'slow' soil N pool, by contrast, takes 40 years after the decrease in deposition to reduce to half of its maximum value. The 'passive' soil N pool only stops increasing in size 150 years after the deposition decrease, since it continues to be supplied by inputs from the 'slow' N pool. Note that the pools have been normalised to a maximum of one in Figure 8; in fact the 'passive' N pool is around 700 times larger than the amount of plant-available N in a given year and the 'slow' N pool is around 10 times larger. It is difficult to test such long-term predictions, but the underlying N14C model was set up using ¹⁴C dating to track the slow evolution of soil organic matter pools in the 12000 years since deglaciation (Tipping et al., 2012).

Figure 8. Increases in soil N pools (extra over constant low deposition scenario) with different turnover rates to a hypothetical increase in N deposition from 2 kg ha⁻¹ yr⁻¹ to 20 kg ha⁻¹ yr⁻¹ for the period 1970-2000, as predicted by the MADOC model for a peatland system.



Soil organic matter in fact consists of materials with a wide range of turnover rates, which are aggregated into the pools used in MADOC and N14C to make a tractable model. However, the patterns illustrated are likely to be broadly correct, in that N deposition contributes immediately to plant-available N, but also tends to accumulate in soil pools with rather long turnover times. These slower pools return N into plant-available form gradually, implying that ongoing effects of accumulated N on plant species will not be acute but will continue for a very long time.

Figure 9. Comparison of effects of two N deposition rate scenarios ('pressure') on 'risk' (extra slow-pool soil N in comparison to pre-industrial conditions) and 'damage' (extra plant-available N in comparison to pre-industrial conditions), in a wet heath site (Migneint) as simulated by the MADOC dynamic model. Solid line = Current Legislated Emissions (CLE) scenario for N deposition. Dashed line = decrease to upper end of critical load range for blanket bogs (10 kg N ha⁻¹ yr⁻¹) from 2010.



A more realistic scenario of N deposition is presented in Figure 9, which illustrates MADOC simulations of the effects of N deposition on the buildup of N within the 'slow' soil pool, and on plant-available N, in a wet heath system within the Migneint SAC. Annual N deposition represents pressure on the ecosystem. Deposition in excess of the critical load (10 kg N ha⁻¹ yr⁻¹ in this case) results in an increase in the size of 'slow' N pool. This N is potentially plant-available, being likely to be converted to immediately plant-available N either gradually (as shown in this simulation) or

abruptly if temperature or aeration increase. This ‘slow’ N pool represents a risk that the habitat will be damaged through excessive plant-available N. The plant-available N flux reflects the actual damage occurring to the habitat in any year. Reducing the annual deposition to the critical load immediately reduces damage in terms of plant-available N, and provides longer-term benefits by reducing the further risk of release of plant-available N.

In the epiphytic context, the substrate volume (bark and associated litter) is much smaller, leading to a more rapid turnover of chemical conditions such as N concentrations and substrate pH than in a soil-based system. The chemical conditions on tree bark to support epiphytic communities may therefore turn over on timescales of 3-10 years, compared with 5 to 10 times this for soil-based systems. Ectohydric mosses are perhaps even more decoupled from the substrate, obtaining most of their nutrients from atmospheric deposition, and are likely to have N turnover times under 3 years. While dynamic models have been developed to investigate the timescales of responses to changes in N deposition for soil based systems, such analyses have not yet been made for epiphytic communities.

2.8 Models of floristic change

Models of floristic change are based on empirical data and/or expert judgement about the environmental requirements of individual species or vegetation-types. Several models have been applied for the purpose of reporting to the UNECE Coordination Centre for Effects (CCE), and are summarised in Table 4.

Table 4. Models of floristic change used to simulate air pollution impacts in European countries.

Model	Summary	Reference
MOVE	Original model of species occurrence in relation to environmental gradients.	(Latour and Reiling, 1993)
GBMOVE	Application of MOVE principle to UK species. Models constructed using very large floristic datasets for the UK. ~1100 species.	(Smart et al., 2010)
MultiMOVE	Latest UK version of GBMOVE, uses more sophisticated model-fitting methods. ~1350 species. ‘Fertility’ axis based on soil N availability (see Rowe et al., 2011a).	(Henrys et al., submitted)
PROPS	Species modelled using datasets from several N-European countries. ‘Fertility’ axis based on N deposition rate, i.e. no chemical delays.	(Wamelink and et al., in prep)
VEG	Species cover modelled dynamically	(Belyazid et al., 2011)
BERN	Predicts environmental suitability for phytosociological plant-assemblages (mainly applied in Germany and nearby countries)	
NTM	Directly predicts nature conservation value (based on Dutch criteria).	

These models can be categorised in various ways:

Occurrence vs. cover models. Most of the models are trained on occurrence (presence/absence) data, so predict prevalence (given dispersal constraints) or more accurately habitat suitability. The

VEG model is an exception in that it predicts the abundance or cover of the modelled species. This has advantages in terms of relating outputs to habitat-type descriptions. However, given that cover is not well-correlated with prevalence, it is debatable whether cover can be accurately predicted.

Dynamic vs. equilibrium models. Most of the floristic models are equilibrium models, that is they assume that given certain conditions, the chance of the species occurring can be determined. The VEG model rather assumes that current conditions determine the competitive strength of a species, and the actual cover is calculated as the outcome of a competition process. Applying this model to increasingly large numbers of species shows that it is most accurate for 10-40 species (Harald Sverdrup, *pers. com.*)

Species vs. assemblage models. Models of the occurrence of assemblages (a.k.a. communities, phytosociological types) generally predict with more precision than species models, since the requirements for a particular assemblage are likely to be more constrained than those for a species. This advantage must be set against the uncertainty in relating a set of species to a particular assemblage. Whether an assemblage of species can be seen as a unit has long been debated.

Floristic models have the advantage of producing outputs that can be related directly to conservation priorities and targets. However, the outputs from a biogeochemical-floristic model chain are inevitably more uncertain than those from a biogeochemical model alone. Another consideration is that species models are more likely to be available, and more certain, for more widespread species for which many records exist.

3. Implications for different geographical locations

3.1 Deposition models currently used in the UK

Spatial variation in recent deposition rates is increasingly well-understood. Changes over time in national N emissions can also be estimated (RoTAP, 2012). However, the spatial location of historic sources is known with less certainty, and currently historic deposition rates are modelled by rescaling the modern day spatial emissions distribution to the appropriate year.

The EMEP4UK, FRAME and CBED nitrogen deposition data sets are all available with a grid resolution of 5 km for the UK. A high resolution version of the FRAME model has been developed which can calculate nitrogen deposition and air concentrations with a 1 km grid resolution for the UK. The implications of using high resolution data for assessing effects on the natural environment are discussed for ammonia concentrations in air in Hallsworth *et al.* (2010) and for nitrogen deposition in Dore *et al.* (2012). The use of 1 km resolution data was found to more realistically separate source (agricultural) areas of ammonia from sink areas (semi-natural habitats). This occurs because ammonia emissions are distributed in the AENEID model (Atmospheric Emissions for National Environmental Impacts Determination) based upon suitability of land classification for livestock grazing (Hellsten *et al.*, 2008). Modelling ammonia concentrations with a 1 km resolution gave improved agreement with measurements at semi-natural sites when compared with 5 km resolution data. The percentage area of Special Areas of Conservation with exceedance of the critical level of $1 \mu\text{g m}^{-3} \text{NH}_3$ for sensitive species was 40% using 5 km resolution data and 21 % using 1 km data. This was attributed to the reduced tendency for mixing source and sink areas in the same model grid square using high resolution data. Concentrations of NO_2 were also better represented (close to major roads) using high resolution data. The use of high resolution precipitation data in hill areas can also lead to more accurate spatial representation of wet deposition in upland areas. However, whilst grid resolution can locally make an important difference, the national scale summary statistics for

the exceedance of critical loads for nitrogen deposition for all habitats were found not to be highly sensitive to the model grid resolution. Exceptions occurred for specific vegetation types (i.e. montane due to spatial location predominantly in high precipitation areas). The high resolution simulations were used in Defra funded projects to define strategies to protect natural ecosystems from high ammonia deposition through tree planting in 'Agroforestry Systems for Ammonia Abatement' (Bealey et al., 2012) and using buffer zones in 'Future patterns of ammonia emissions across the UK and the potential impact of local emission reduction measures' (Dragosits et al., 2013).

The distribution of wet deposition is also affected by the grid size. Rainfall patterns over complex terrain are better represented by hi-resolution models. This has been shown in the ROTAP report where the EMEP4UK model (Vieno et al., 2010) was applied at 5 km² and 50 km². The simulation shows that a more highly resolved meteorology was able to capture the elevated rainfall over complex terrain better than the 50 km². As an example, a sample of the EMEP4UK model domain is shown in Figure 10 for wet deposition at different scales (50, 5 and 1km²) and in Figure 11 the same but for dry deposition. Dry deposition of reduced nitrogen is also spatially dependent; however, the effect is less dramatic as it is more dependent on the location and magnitude of emissions. The emissions used in the EMEP4UK model are consistent within the 3 model domains.

Figure 10. EMEP4UK calculated wet depositions of reduced nitrogen for the Edinburgh and highland area at different horizontal resolutions: a) 50km², b) 5 km² and c) 1km². Units are mgN m².

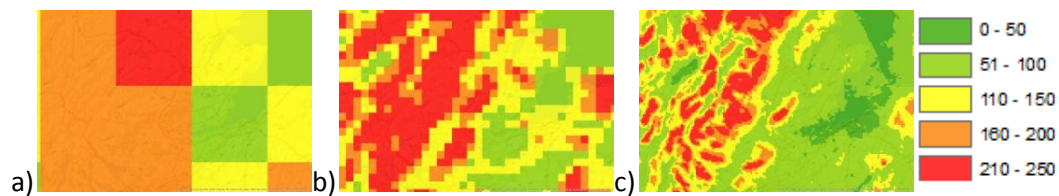
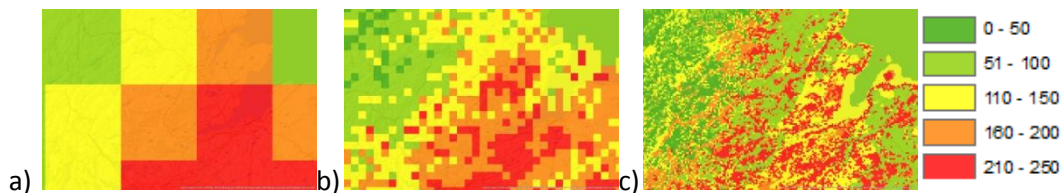


Figure 11. EMEP4UK calculated dry depositions of reduced nitrogen for the Edinburgh and highland area at different horizontal resolutions: a) 50km², b) 5 km² and c) 1km². Units are mgN m².



Both FRAME and EMEP4UK are atmospheric chemical transport models. To calculate deposition of oxidized and reduced nitrogen they require accurate spatial data on the distribution of NH₃ and NO_x emissions in the UK (available from the National Atmospheric Emissions Inventory, <http://naei.defra.gov.uk/>). The availability of data for mapping emissions sources has improved significantly during the last decade. However whilst accurate historic emissions inventories are available back till 1970 for NO_x and 1990 for NH₃, detailed emissions mapping has not been undertaken for these early years. As a result the spatial distribution of emissions is more uncertain for historic years. This results in localised uncertainty in model estimates of dry deposition of nitrogen. However, the location of important sources of NO_x (major roads and motorways) and of NH₃ (agriculture) have not been subject to large scale geographical re-distribution in the UK during recent decades. Furthermore, wet deposition is less sensitive to local spatial emission variations as aerosol particles which make the dominant contribution to wet deposition are subject to long range transport of several hundred km. Matejko *et al.* (2009) compared the trend in wet deposition of oxidized and reduced nitrogen deposition in the UK during the period 1990-2005 and found general agreement between the FRAME and CBED estimates.

CBED (Concentration Based Estimated Deposition) uses measurements of gas and aerosol concentrations in air and of oxidized and reduced nitrogen concentrations in precipitation combined with deposition velocity estimates (Smith et al., 2000) and maps of precipitation to generate spatial estimates of nitrogen deposition in the UK (Smith and Fowler, 2000). The data are available for wet deposition back to 1986 and for dry deposition since 2000 with some estimates for earlier years. CBED data have been used extensively to estimate the recent historical change in nitrogen deposition and the exceedance of critical loads in the UK (Fowler et al., 2005; RoTAP, 2012). The small estimated decrease in total nitrogen deposition to the UK of 13% between 1988 and 2008 is despite large (50%) decreases in emissions of NO_x and 18% decreases in emissions of NH_3 . There are substantial non-linearities in the relationship between emission and deposition of oxidised nitrogen; while they imply a steady increase in rates of oxidation of NO_2 with time, the mechanistic detail remains largely unknown. The main consequence, however, is an estimated decline in exports from the UK of nearly 50%. Fowler et al. (2004) describe estimates of nitrogen deposition over a century, based on the trend in measured rainfall concentrations. Annual nitrogen deposition to the UK was estimated at 312 Gg N for the year 1900, peaking at 787 during the decade 1980-1990 and subsequently falling to 460 Gg by the year 2000.

Uncertainties in the historic spatial patterns of N emission are unlikely to affect conclusions as to the spatial pattern in cumulative deposition if the time period for integrating deposition is of the order of 30 years (see Section 3.2). This is because the main historic changes in spatial emissions pattern (e.g. due to changes in the motorway network and major combustion sources, and in the pattern of livestock farming) had already taken place before 1980.

3.2 Methods for calculating cumulative deposition

The cumulative amount of N deposition considered to have affected an ecosystem depends on: the time period over which deposition is integrated; and the threshold deposition rate over which deposition is integrated (Figure 12). 'Integration' is used here in a mathematical sense, i.e. the method for calculating the area under a curve, within set boundaries, by adding together small (in this case annual) subdivisions. In this section, considerations for deciding on these boundaries are discussed. Some examples are illustrated in Figure 13, which shows the relationship between 2004-6 deposition rate and cumulative deposition as calculated using different periods and thresholds. These are based on a temporal pattern of change in deposition expressed as a ratio to current deposition. At present there is insufficient evidence for changes in the spatial pattern of emissions (see Section 3.1) so all locations use the same temporal pattern of ratios to 2004-6 deposition. Note that when an integration threshold of zero deposition is used, the relationship between cumulative deposition and 2004-6 deposition is a straight line passing through the origin, and the spatial patterns of 2004-6 deposition and cumulative deposition are identical (see Figure 14).

Figure 12. Dependence of cumulative deposition (shaded area) on definitions of the start and end dates of the integration period, and the threshold above which deposition is integrated.

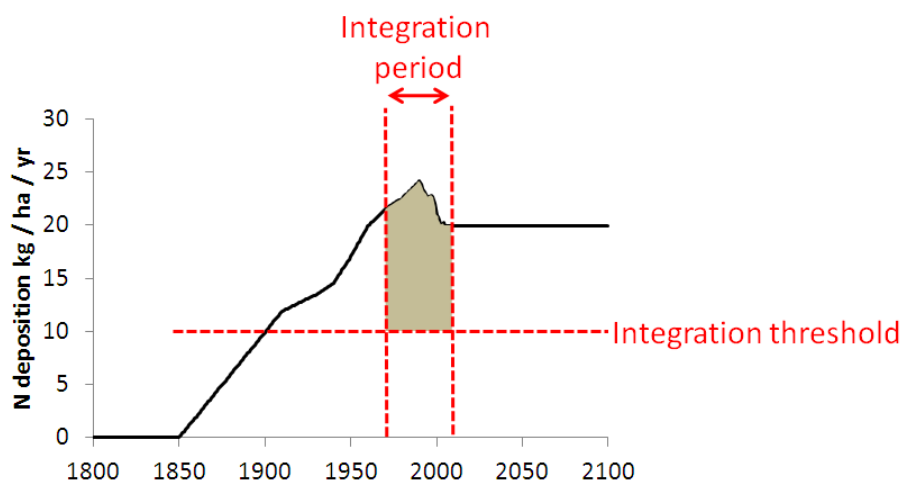
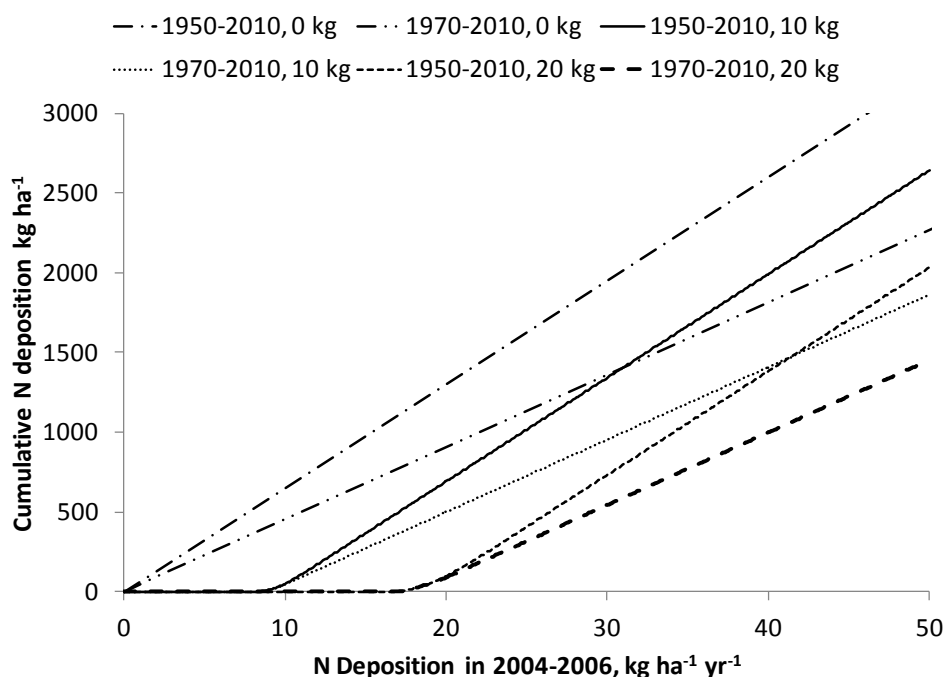


Figure 13. Relationship between 2004-6 deposition and cumulative deposition as calculated using different integration periods (1950-2010 or 1970-2010) and different integration thresholds (0, 10 or 20 kg ha⁻¹ yr⁻¹).



Time period for integrating deposition

Deciding on the end date for the period over which deposition is integrated is comparatively straightforward, although there is a choice regarding which aspects of the relationship to highlight. Using a fixed date, such as 2010, allows a simple comparison of different geographical locations in terms of their cumulative N load during the period of greatest emissions. This approach is used in Section 3.3. By contrast, calculating cumulative deposition up to the current date (i.e. in the year the metric is reported) would highlight temporal changes and deposition decreases, particularly if a moving temporal window were used (e.g. the 30 years preceding the current date).

Deciding on the start date for the period over which deposition is integrated is more difficult. As noted in Section 3.1, deposition estimates are more accurate for recent periods and increasingly uncertain further into the past. This uncertainty may be a consideration when setting the start of the integration period, but probably a more serious issue is that the most suitable start date depends on the turnover rate of N in the ecosystem and thus the mean N retention time (see Section 2.7). Modelling studies such as Rowe et al. (2011b) suggest that extra N is retained in soil for several centuries, and in some ecosystems this extra soil N is likely to carry on becoming plant-available, albeit in gradually diminishing amounts, for very extended periods. Conversely, some ecosystems, or parts of ecosystems, have very little retention capacity for N. For example, epiphytic lichen communities store little N and chemical conditions may recover rapidly following a decline in N input (e.g. Mitchell et al., 2004). Based on expert judgement, typical mean retention times for N within ecosystems are likely to be around:

<u>epiphytes and predominantly ectohydric mosses</u>	1-3 years
<u>predominately endohydric mosses</u>	5 years
<u>soil-based ecosystems</u>	30 years

The suggested periods for epiphytes and mosses are based on the observed rapid recovery of moss and lichen tissue-N and cover proportion with decreases in N deposition rate (Jones, 2005; Mitchell et al., 2004). The mean retention time of N in soil-based ecosystems is based on expert judgement, but could be given a more empirical basis using studies of chemical recovery, or measurements using the stable isotope ¹⁵N of the mean retention time for N within ecosystems (see Section 6 Evidence gaps).

In modelling studies, the N stored in the ecosystem can be quantified in terms of amounts in pools with different rate-constants, and mean turnover time can be calculated. The amount stored in an intermediately-available pool could be used as a metric of risk or damage. We will return to the question of N storage and the use of model outputs as impact metrics in Section 5. However, a simpler approach, if the mean retention time of N within an ecosystem can be estimated, would be to accumulate deposition over an equivalent period. For example, if the mean retention time of N in an ecosystem is 3 years, then the N stored is better-correlated with cumulative deposition over the preceding 3 years than with cumulative deposition over the preceding 30 years; whereas if the half mean retention time is 30 years, the amount stored is better-correlated with cumulative deposition over the preceding 30 years.

Another approach to assessing the best method for calculating cumulative deposition would be to evaluate which method best explained observed variation in relevant responses e.g. of species richness in UK acid grasslands. In view of the scatter in observations, it might be difficult to make a decision on this basis, but such an approach would be worth exploring and is noted in Section 6 Evidence Gaps.

Deposition threshold for integrating deposition

Deciding on a suitable threshold above which cumulative N is integrated is also not easy, since it depends on the amount of N that an ecosystem can process without harmful effects. The most precautionary approach would be to assume that any deposition above zero should be included, but pre-industrial ecosystems would have received some atmospheric N – the amount is uncertain but can be estimated at 1-3 kg N ha⁻¹ yr⁻¹. Also, losses of N from the ecosystem mean that not all of the deposited N can be considered harmful. Values for these N loss terms are discussed in Hall et al. (2003), who suggest values applicable when using the ‘steady state mass balance’ approach to calculating the critical load for nutrient N (CL_{nut}N). This is defined as:

$$CL_{nut}N = N_u + N_i + N_{le(acc)} + N_{de}$$

Where N_u = nitrogen uptake (3-6 kg N ha⁻¹ yr⁻¹ in managed woodlands, zero in other habitats)

N_i = nitrogen immobilisation (1-3 kg N ha⁻¹ yr⁻¹)

$N_{le(acc)}$ = acceptable level of nitrogen leaching (1-5 kg N ha⁻¹ yr⁻¹)

N_{de} = denitrification (1-4 kg N ha⁻¹ yr⁻¹)

This approach is only currently applied in the UK to managed woodlands (empirical critical loads are used for other habitats) and values for the terms for other habitats have not been agreed. However, the approach could be developed and applied to calculate non-harmful N deposition rates for different habitats. The ranges in values given above are taken from Hall et al. (2003), and give a range in total non-harmful deposition of 3-18 kg N ha⁻¹ yr⁻¹.

Whether these fluxes are completely non-harmful is questionable. Net immobilisation implies an increase in N stored in the soil, which is likely to be re-mineralised at some point in the future, leached N may contribute to freshwater pollution, and denitrification may imply an increase in greenhouse gas (N₂O and NO) emissions. Several species have been shown to decline in prevalence at very low rates of N deposition such as 5-10 kg N ha⁻¹ yr⁻¹ (Emmett et al., 2011), and at very low ammonia concentrations such as 1 µg NH₃ m⁻³ (Cape et al., 2009). However, deposition in the lower end of this range is probably acceptable for most terrestrial ecosystems, and 5 kg N ha⁻¹ yr⁻¹ could be used as a value for the amount of N that can be processed without serious harm.

Another approach would be to integrate deposition above the empirical $CL_{nut}N$, as defined using evidence from experiments and surveys (Bobbink and Hettelingh, 2011). Values for most UK habitats (defined at EUNIS level 2 or 3) have been agreed for use in mapping critical load exceedance (Hall et al., 2011). Apart from saltmarshes (which naturally receive large N loads from the sea, and so are rather insensitive to N), these empirical critical loads are in the range 7-15 kg N ha⁻¹ yr⁻¹. It should be noted that these values are greater than the acceptable deposition flux identified in the previous paragraph, perhaps because of the emphasis on relatively short-term experimental studies in defining $CL_{nut}N$. The results of the analyses of large-scale UK survey data (Stevens et al., 2011c), showing effects below the current $CL_{nut}N$, were not available in time to be included in the review of critical loads (Bobbink et al., 2010).

We will return to the choice of deposition threshold for calculating cumulative deposition in Section 5.2.3.

3.3 Spatial patterns in current and cumulative deposition

The implications of using different periods and thresholds for calculating cumulative deposition are illustrated in Figure 14, which shows different potential deposition metrics in relation to the map of dwarf-shrub heath (Figure 14a). Since the same trajectory of ratios to current deposition is applied across the UK, the spatial pattern of cumulative total deposition (Figure 14c) is identical to that of current deposition (Figure 14b). Integrating deposition above a threshold (Figure 14d) increases the distinction between areas with greater deposition where the threshold has been greatly exceeded for many years, and areas with less deposition where the threshold has been exceeded to a lesser extent and for fewer years. The integration periods were chosen to give a similar range of cumulative deposition using both methods. Integrating deposition above a threshold results in a larger proportion of the area being shown as in the lowest category than does integrating total deposition (see also Figure 15). However, integrating deposition above the critical load threshold highlights hotspots of deposition more clearly.

Figure 14. Spatial patterns of total N deposition to 'moorland' (in contrast to 'grassland' and 'woodland') as calculated using the FRAME model: a) distribution of dwarf shrub heath, shown as % area of each 1 km² grid-cell; b) recent deposition flux (annual mean 2004-6); c) cumulative total deposition (1970-2005); d) cumulative deposition over the critical load for nutrient N for dwarf shrub heath, 10 kg N ha⁻¹ yr⁻¹ (1970-2010).

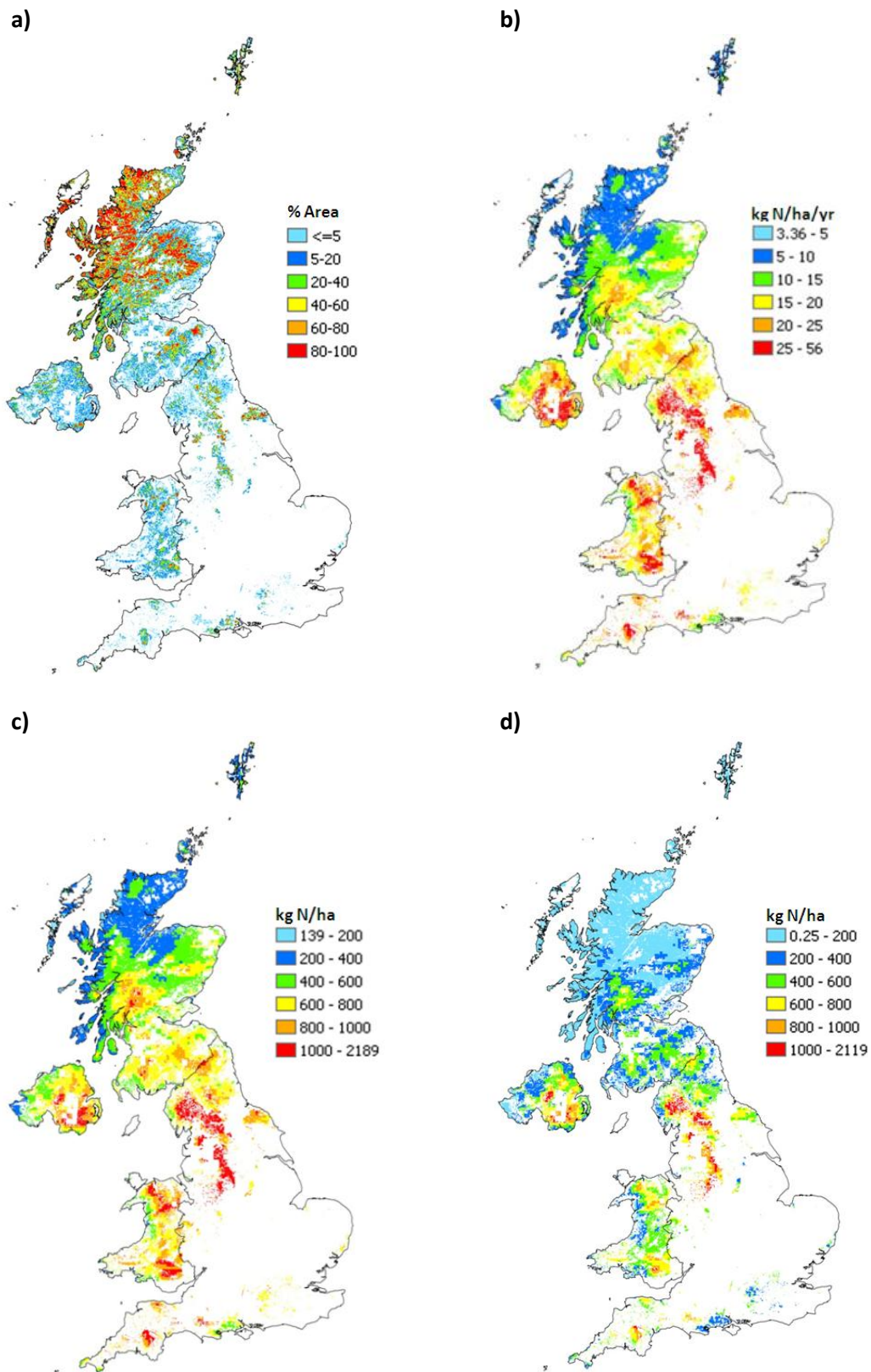
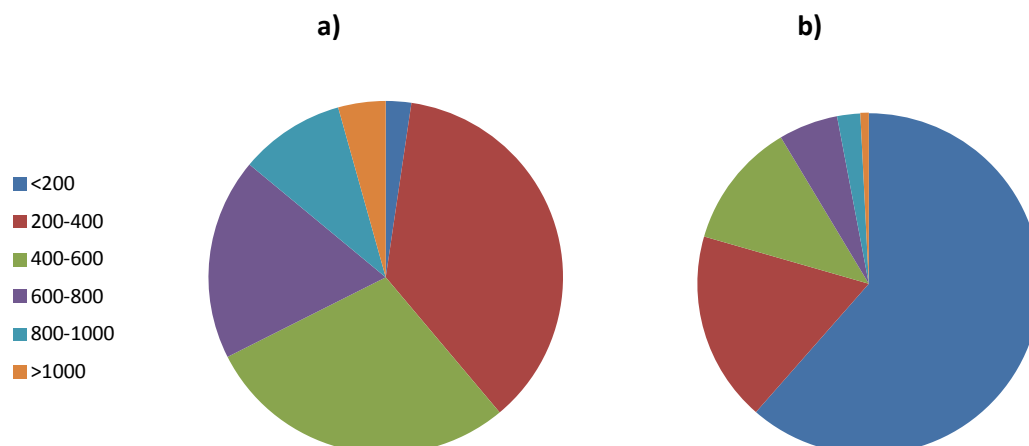


Figure 15. Proportions of total area of UK dwarf-shrub heath that have received different amounts of cumulative N deposition (kg N ha^{-1}), calculated as: a) cumulative total deposition (1970-2005); b) cumulative deposition over the critical load for nutrient N for dwarf shrub heath, $10 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ (1970-2010).



3.4 Implications for UK regions

The choice of integration period for calculating cumulative deposition could affect the spatial targeting of abatement and mitigation measures. By using a non-zero integration threshold such as the critical load, a greater emphasis is placed on areas where current deposition is greatly in excess of the critical load. For example, the effect of a change from integrating total deposition to integrating deposition above the critical load is greater for Scotland, where the area in the lowest deposition class changes from 3% to 74%, than for England, where the area in the lowest class changes from 0% to 1% (Table 5).

Table 5. Effect of different integration thresholds for calculating cumulative nitrogen deposition: a) cumulative total deposition, i.e. all deposition above zero (1970-2005); b) cumulative deposition over the critical load for nutrient N for dwarf shrub heath, $10 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ (1970-2010), on the percentage areas of dwarf-shrub heath falling within different deposition categories in the UK Devolved Administrations and in the UK as a whole.

	Integration threshold	Cumulative deposition, kg N ha^{-1}					
		<200	200-400	400-600	600-800	800-1000	>1000
% of dwarf-shrub heath							
England	a) zero	0%	0%	5%	30%	36%	29%
	b) CL	1%	16%	32%	29%	15%	7%
Wales	a) zero	0%	1%	8%	24%	44%	23%
	b) CL	3%	11%	39%	36%	11%	0%
Scotland	a) zero	3%	44%	33%	16%	4%	0%
	b) CL	74%	18%	7%	1%	0%	0%
N. Ireland	a) zero	0%	7%	30%	42%	16%	4%
	b) CL	14%	38%	27%	14%	6%	1%
UK	a) zero	2%	37%	29%	18%	10%	4%
	b) CL	61%	18%	12%	6%	2%	1%

Another reason that integrating deposition above the critical load tends to increase emphasis on English heaths (7% in the maximum category for cumulative deposition, compared to 0 or 1% for the other DAs) is that the largest areas of dwarf shrub heathland in England coincide with the largest

deposition rates (see Figure 14a). Spatial patterns of cumulative exceedance integrated above the critical load would be different for other habitats, because of different distributions of the habitat, and also since critical loads differ among habitats.

The use of different integration thresholds, as illustrated in Figure 14c and Figure 14d, changes the impression given of where damage has occurred, and therefore has implications for policy responses. The map of cumulative deposition over CL_{nutN} (Figure 14d) focuses attention on the hotspots of deposition, with comparatively large areas of the UK in the lower-risk categories. However, it could be argued that sites that have received large cumulative doses (such as the Peak District) are very damaged and unlikely to respond rapidly to decreases in N deposition; whereas areas such as the Southern Highlands or Dumfries and Galloway are relatively unaffected and species loss could still be avoided. It is clear from the continuing effects of N deposition rates at increasingly large doses (Emmett et al., 2011) that there are benefits to biodiversity from preventing increases in deposition both in high-deposition and low-deposition areas. However, the evidence-base for contrasting effects of decreases in deposition in sites that have received low and high cumulative N doses is sparse (see Section 2.6). Although different ways of mapping cumulative deposition do not avoid the need for potentially difficult decisions about spatial targeting, on balance a map of cumulative deposition above a threshold such as Figure 14d, with its greater distinction between high- and medium- deposition rate areas, is likely to provide better support for such discussions.

The maps of cumulative total exceedance (Figure 14c) and cumulative exceedance of a threshold during a preceding period (Figure 14d) also illustrate patterns within each of the DAs that may be useful in informing policy responses. Using a threshold and a limited preceding period results in the mapping of larger areas as under low-to-medium pressure, and focuses attention on smaller areas that are likely to be more severely affected by N. The map (Figure 14c) shows that areas at relatively high altitude yet close to emissions sources have received N deposition substantially above the critical load for an extended period. These include areas of the Peak District, Lake District and South Pennines in England, the Black Mountains and Berwyn in Wales, and the Mourne and Sperrin Mountains and the Antrim Hills in Northern Ireland.

4. Critical assessment of current evidence and tools

4.1 Critical assessment of empirical evidence

There is a growing evidence base demonstrating the effects of N on the soil and vegetation of semi-natural habitats. The recent JNCC study that related change in species and aggregate indicators to N deposition using statistical analyses of large datasets provides particularly compelling evidence of N impacts on species and habitats (Emmett et al., 2011; Stevens et al., 2011c). It is clear from the restricted distributions and patterns of temporal decline observed in that study for many sensitive species that N deposition has had large detrimental effects on UK biodiversity. There is also a strong evidence base for N deposition effects on ecosystem functions, such as plant tissue N concentration at relatively low rates of deposition and N leaching at relatively high rates.

The persistence of N in systems implies that previous deposition has affected and will go on affecting habitat structure and functions. However, analyses of survey and experimental data in relation to cumulative N deposition have been limited. The UK studies reviewed, in particular Phoenix et al. (2012) and Caporn et al. (2011), used total cumulative deposition, which (given current calculation methods) has the same spatial distribution as current deposition, and so adds little explanatory power. Experimental additions at realistic rates make comparatively little difference to total

cumulative deposition calculated over a long period. Re-analyses of biogeochemical and biological responses using a more informative indicator of cumulative deposition, such as integration above a threshold for a limited preceding period, might reveal useful relationships.

Slow rates of species recovery, particularly when a species has been lost from a site or locality, may be related to establishment or dispersal limitations as well as to the persistence of N. However, no studies were identified that have successfully separated these effects. The timescale and implications of biological recovery delays (see Figure 1) therefore remain highly uncertain.

The evidence base for the impact of decreases in N deposition is also small, being largely restricted to experiments where N applications continued for a time and were then stopped or reduced. Experiments in which rainfall is intercepted and replaced to achieve decreases below the ambient deposition rate are harder and consequently rarer. There is therefore currently little empirical evidence for the benefits of a marginal decrease in N deposition.

4.2 Critical assessment of evidence from modelling studies

Models of biogeochemical changes in soil and vegetation in essence extend the concept of using a regression fitted to empirical data to make predictions, by allowing temporal dynamics and feedbacks to be taken into account. Models have a tendency to grow over time, as more processes are considered and in an attempt to better fit observed data (Rowe et al., in press). However, increasing the structural detail and the number of parameters results in less generalisable models with extensive data requirements, and efficient or parsimonious models (that predict accurately from few inputs) are considered preferable. The optimum amount of detail for simulating N effects on ecosystem biogeochemistry is debatable, and models with a range of complexity are available. Ideally, the models that could be used would be compared against a common dataset, and a choice of model made on the basis of performance. The resources required to set up several models for a range of test sites have made model intercomparison exercises uncommon, although some progress is currently being made in the EC FP7 project 'ECLAIRE'.

The increase in detail in most of the models used within the UNECE-CCE process in recent years is justified, since single-pool models of progressive and irreversible saturation did not adequately represent the dynamics of N retention and loss from ecosystems. Models such as N14C and VSD+ can now simulate effects such as C/N ratio increase with N deposition, and predict changes in quantities that are highly relevant to discussions of risk and damage to habitats. Soil N pools and fluxes that are conceptually useful but hard to measure consistently, such as plant-available N, or N that is likely to be mineralised over timescales of several decades, can be simulated and used for illustration (Figure 9). Although models of soil and vegetation biogeochemistry are mainly useful to illustrate and develop current theoretical understanding, and should not be viewed in the same way as empirical evidence, they provide an important source of information for communicating and visualising effects of N. Considerable effort has been expended on developing these models for UK ecosystems, and their potential for scenario exploration remains underexploited.

Models of plant species niches such as MultiMOVE are essentially collections of empirical relationships, and as such provide important supporting evidence for N impacts, as well as allowing prediction and exploration. Applications to scenario analysis, and to allow managers to assess implications for individual species on a site, have been proposed but as yet these ideas have not been exploited.

4.3 Critical assessment of currently-used impact indicators

Metrics have an important role in communicating the effects of policy decisions. A good metric simplifies but does not misrepresent current scientific understanding, and can be related to effects that are important to people. Ideally a metric should also be measurable or easily related to simple observations. Many types of measurements and observations have been proposed as indicators of N pollution, but some of these are difficult to measure consistently (e.g. N deposition rate, plant-available N, slowly-mineralised N stocks), some are not consistently related to the degree of pollution by N (e.g. soil total C/N ratio, plant tissue N/P ratio) and some are related not only to N pollution but to other drivers, notably management (e.g. sub-shrub/graminoid cover ratio, grass/forb cover ratio, mean 'Ellenberg N' score). Measurements that are potentially useful for surveillance of N effects were reviewed in Emmett et al. (2011), which assessed each measurement for sensitivity to N impacts, ability to discriminate from other environmental drivers, and cost.

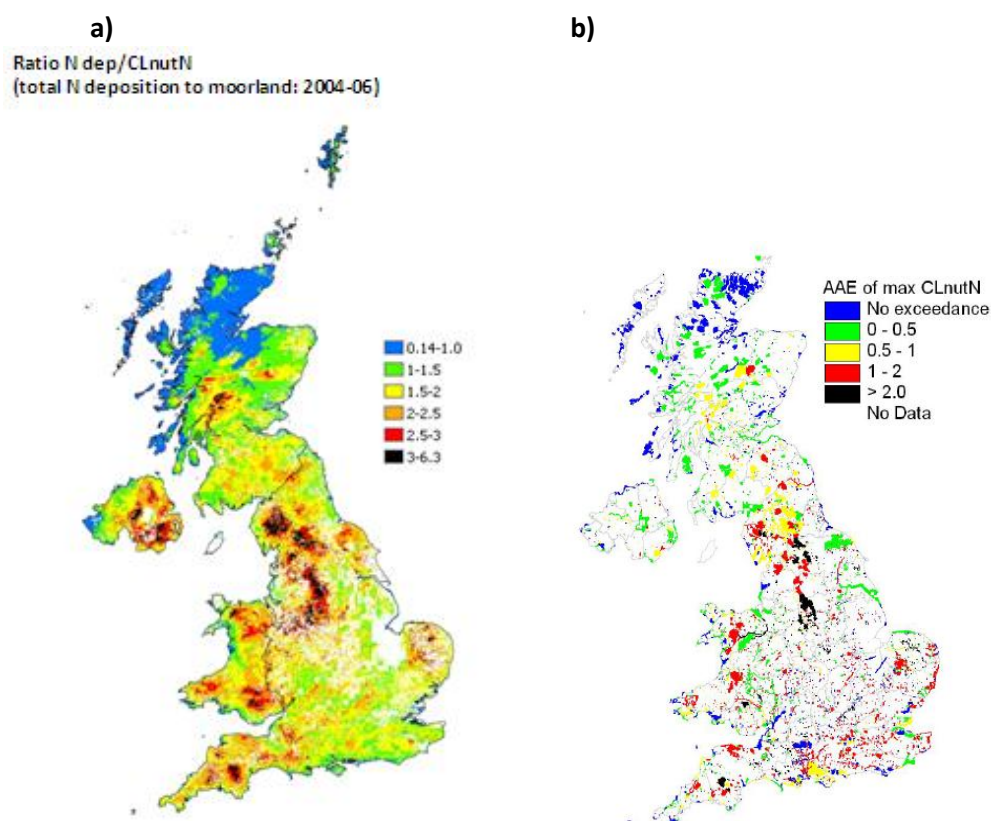
The metrics currently used in reporting risk of impacts of N deposition and for policy evaluation are:

- **% area exceeded:** area of sensitive habitats where CL_{nutN} is exceeded (% of total sensitive-habitat area, or by Broad Habitat type; reported for whole UK and for protected sites);
- **Average Accumulated Exceedance (AAE):** average (weighted by habitats' areas) exceedance of CL_{nutN} for habitats within a grid square ($kEq\ N\ ha^{-1}\ yr^{-1}$).

The CL_{nutN} is currently exceeded over a large proportion of the UK (Figure 16a). This explains why the ' % area exceeded' metric has not responded much to decreases in N emissions and decreases in N deposition rates. Although total NH_4 deposition in the UK has remained approximately constant over the past two decades, there was a decrease in NO_x deposition of approximately 24 % between 1988 and 2008 negligible (RoTAP, 2012). However, over most of the UK total deposition remains considerably greater than CL_{nutN} and so a decrease only brings relatively small areas in the northwest of Scotland, Wales and Northern Ireland below the CL_{nutN} threshold.

The other metric currently reported by the UK National Focal Centre is AAE, which is not based on binary information (i.e. exceedance of CL_{nutN} or not) and so is more responsive to changes in deposition than is % area exceeded. This metric is used in integrated assessment modelling (e.g. Figure 16b) and by the UNECE-CCE for generating aggregate statistics. However, AAE was not chosen for inclusion as a headline indicator (DEFRA, 2012). This may in part be due to the units reported. Originally 'accumulated exceedance' was reported, in $Eq\ N\ km^{-2}\ yr^{-1}$; units were changed to $kEq\ N\ ha^{-1}\ yr^{-1}$ for reporting AAE, but many people are more familiar with units of $kg\ N\ ha^{-1}\ yr^{-1}$. Another reason this indicator may not have been perceived to communicate N impacts well is its name, which does not make the meaning obvious. Exceedance accumulated (over areas) is easily confused with cumulative exceedance (over time). If an alternative name were used, such as 'Average Exceedance', and the metric were expressed in $kg\ N\ ha^{-1}\ yr^{-1}$, it might be more widely understood and used.

Figure 16. Illustrations of currently-reported metrics: a) spatial pattern of N deposition (as estimated by the FRAME model for moorland habitats in 2004-6) expressed as a ratio to the critical load for nutrient N (5th percentile CL_{nutN} for habitats in the grid square); b) Average Accumulated Exceedance (AAE), calculated for protected sites: average (weighted by habitats' areas) exceedance of CL_{nutN} for habitats within each grid square ($kEq\ N\ ha^{-1}\ yr^{-1}$).



Both AAE and % area exceeded are based on current deposition, and are indicators of pressure rather than of impact. They do not represent important temporal effects such as the delayed responses of chemical and biological indicators to changes in deposition (Figure 1) and the persistence of pollutant N within ecosystems.

These temporal aspects of N impacts are important. The large amounts of anthropogenic N stored in many ecosystems mean that soil N availability will be elevated relative to pristine conditions for decades or centuries. Although a decrease in N deposition will have some immediate benefits, particularly for species less exposed to soil N, chemical conditions will not recover instantly. Recovery of species is also likely to be delayed, particularly where the species has become extinct in the locality. The effects of a unit decrease in N deposition are therefore likely to be different to the effects of a unit increase, and this hysteresis should be taken into account when defining a suitable metric.

Timescale is also important when it comes to evaluating change and assessing suitable policy responses. The impoverishment of habitats that results from chronic N pollution occurs over decades and is difficult to notice even with targeted monitoring. Increasing homogeneity of habitats tends to reduce public expectations for how diverse a habitat should be (Miller, 2005). Given the continued increase in global fossil CO_2 emissions and atmospheric CO_2 concentrations, species and habitats will likely be subject to considerable climatic changes in the medium term. These may override the effects of N pollution, although the mechanisms by which N pollution decreases species diversity are likely to continue to operate under a changing climate. Despite the severity of climate

change projections for the second half of the 21st century, it has proved difficult to engage public and policy interest in effective measures to reduce these projected effects. This may reflect a preference for positive messages (Futerra, 2009), but predictions for effects in the future are inevitably of less interest, to the public and to policy decision makers, than more immediate effects. Although N pollution undoubtedly has long-lasting effects, it is therefore important for a metric to reflect effects within a near-term timeframe, such as the timeframe which may be targeted in negotiations over the Thematic Strategy on Air Pollution and National Emissions Ceiling Directive, 2020-30.

Another temporal aspect that is poorly reflected in currently used metrics is the effect of early decreases in N deposition. A metric such as AAE is not affected by deposition in preceding years, so to achieve a target for this metric by 2020, it would be most cost-effective to implement deposition decreases in 2019. This does not reflect current knowledge of the cumulative effects of N deposition, and a change to a metric that does reflect these cumulative effects, and can show the benefits of an earlier decrease, would be desirable.

5. Metrics to represent benefits of decreases in N deposition

5.1 Overview of potential metrics

It is useful to distinguish metrics that:

- a) represent the degree of pressure on the ecosystem;
- b) can be seen as markers or midpoints that represent progress towards a desired endpoint, e.g. chemical conditions that make it likely that this endpoint will be achieved in future;
- c) illustrate achievement of a desired endpoint, e.g. a metric that can be directly related to favourable conservation status.

The former type is clearly a pressure metric, whereas midpoint and endpoint metrics are response metrics. Midpoint metrics can be used to illustrate progress towards a goal such as ‘no net loss of biodiversity’, whereas endpoint metrics illustrate the consequences of inaction. The terms do not necessarily relate to the timescale of change, and ‘midpoint’ should not be taken to indicate progress half-way towards a goal.

To the extent that definitions of habitat quality and integrity are based not simply on flora and fauna but on habitat ‘structure and function’, any midpoint metric could be seen as an endpoint metric. The definition of habitat ‘function’ is problematic, since a change in any chemical variable within any organism or ecosystem pool, or a change in abundance of any organism or group of organisms, could be seen as a change in function. For the sake of clarity, we will refer to changes in chemical conditions mainly as midpoint indicators, and changes in biological response mainly as endpoint indicators. These assignments could change where habitat functions that are considered part of the definition of favourable conservation status can be specified.

It is however useful to consider midpoint metrics as being relevant mainly with regard to ecosystem functions and supporting services, whereas endpoint metrics are relevant mainly with regard to ecosystem goods and final services. Endpoint metrics may also act as midpoint metrics, for example changes in the lichen assemblage may provide early warnings of changes to other endpoint metrics. In the following sections, pressure, midpoint and endpoint metrics that are potentially useful for evaluating the benefits of decreases in N deposition are summarised. Metrics based on empirically-established relationships and modelling studies are discussed.

5.2 Pressure metrics

In assessments of N deposition, the principal pressure metrics are those related to the deposition rate in $\text{kg N ha}^{-1} \text{ yr}^{-1}$. Acute and toxic effects may be more closely related to the peak or background concentration of reduced N, but deposition flux is an appropriate indicator of chronic pressure. For several of these metrics, the deposition rate is expressed in relation to the Critical Load for nutrient N, which varies among habitats as outlined in Hall et al. (2011).

5.2.1 Area of sensitive habitat where empirical critical load is exceeded

As noted in Section 4.3, the % of sensitive-habitat area where CL_{nutN} is exceeded is not very sensitive to decreases in deposition load, principally because deposition is greatly in excess of CL_{nutN} for most sensitive habitats over most of the UK. This is a realistic reflection of the severe and ongoing damage caused by N to UK habitats. However, this metric is an instantaneous measure and takes no account of what is known about the persistence of N in many ecosystems.

5.2.2 Spatially-averaged exceedance of empirical critical load

The Average Accumulated Exceedance, which we propose could be referred to as Average Exceedance, is more responsive to decreases in N deposition than is % area exceeded. This makes it more suitable for encouraging progress towards decreases in N deposition. However, this metric is also an instantaneous measure and takes no account of chemical and biological recovery delays.

5.2.3 Cumulative deposition

Cumulative deposition above a given threshold and over a relevant timeframe (see Figure 12) is likely to reflect the pressure on a habitat better than the preceding two metrics. The timeframe should be of fixed length preceding the date that the metric is calculated for. The length of the timeframe depends on the turnover rate of N within the system. We propose a timeframe of 30 years for soil-rooting plants and habitats in which these are major components, and a shorter timeframe of 3 years for epiphytic and epilithic (sub)habitats.

The threshold rate above which deposition is accumulated is probably best defined as the CL_{nutN} for the habitat in question. There are arguments for instead using the total non-damaging N flux term as defined in Steady-state Mass Balance (SMB) modelling, but although ranges in values for this term have been proposed for UK habitats, values to be used have not yet been agreed. The CL_{nutN} values are in any case similar to the proposed SMB ranges.

Decreases in deposition will decrease this term immediately to an extent, and if maintained at a low level the cumulative deposition within the preceding timeframe will reduce commensurately. This more realistically reflects the likely persistence of chemical effects, without the uncertainties that would be introduced by choosing and applying a dynamic model. However, the term remains an indicator only of pressure, and takes no account of biological response delays.

5.3 Midpoint metrics

5.3.1 Total C/N ratio in soil

The (total C / total N) ratio in the litter layer or in soil has some potential for use as a midpoint metric. A decrease in soil or litter C/N ratio is a fairly reliable indicator of N saturation within the ecosystem. However, soil C/N ratio has been observed to be positively correlated with N deposition

in survey studies such as Jones et al. (2004), and there are theoretical reasons for expecting an increase in soil C/N on sites where plant productivity is stimulated by N (Rowe et al., 2011b). During the period 1998-2007, there was either no change or a small increase in soil C/N ratio in the majority of Broad Habitats (Reynolds et al., in press). A similar lack of response in soil C/N was shown by Dise et al. in a survey of European conifer forests (1998). For these reasons, and because it does not reliably indicate progress towards an endpoint metric, soil C/N ratio is probably not a suitable midpoint metric for chemical changes induced by N deposition.

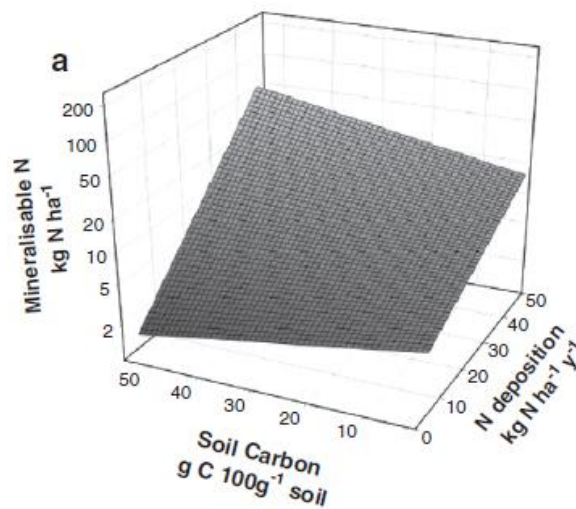
5.3.2 Mineral N in soil

The mineral N content of soil provides an indication of plant-available N, and KCl-extractable (ammonium plus nitrate)-N concentration in soil has been shown to be related to N deposition rate in some experiments, such as that at Wardlow (Phoenix et al., 2012). Although mineral N is probably the major N form taken up by plants, its measurement is not entirely straightforward. The mineral N content of soil is likely to fluctuate in relation to previous rainfall and mineralisation events, which causes temporal variability. Another consideration is that plant uptake and immobilisation into soil organic matter can lead to zero measurements in many N-limited systems, even when the flux into plants is evidently non-zero. Methods for integrating N mineral concentration and preempting plant uptake, such as the use of strong ion-exchange resins to trap mineral N for later analysis, are promising, but as yet the evidence-base for establishing relationships between these measurements and current or cumulative N deposition is insufficient. In view of the difficulty of obtaining consistent measurements of soil mineral N content, this is not recommended as a metric.

5.3.3 Mineralisable N

The amount of N in readily-mineralised form in a soil sample is presumably closely related to the potential flux of plant-available N from the soil sampled. In a large-scale survey of mineralisable N, the total amount of mineralisable N was shown to be related to current N deposition, although mineralisable N was also influenced by soil pH, soil carbon concentration, and mean annual temperature (Rowe et al., 2012). The relationships established in this study (e.g. Figure 17) could be used to translate N deposition into probable values for mineralisable N concentration in soil. This metric has the advantage of being closely related to the exposure of plants to N and therefore likely damage.

Figure 17. Effects on mineralisable N concentration in soil of N deposition rate and soil carbon concentration, at mean values of soil pH and annual temperature within the study dataset. From Rowe et al. (2012).



5.3.4 N leaching

Effects of N on ecosystems, in particular declines in abundance and prevalence of N-sensitive species, are observed well before increases in N leaching are likely to be detectable (see Figure 2). With this proviso, however, it is clear that the onset of N leaching is a good indicator that the ecosystem is becoming saturated with N. The rate of N leaching is not easy to measure directly within terrestrial ecosystems, since *in situ* measurements of soil solution require several months for the suction samplers to stabilise, and direct measurement of water and solute fluxes in drainage requires the installation of lysimeters and even greater stabilisation times. However, samples collected from streams or standing waters draining small catchments provide a potential proxy measure of leachate chemistry, particularly where the catchment is comprised predominantly of the habitat type of interest. It has previously been shown that spatial patterns of NO_3^- in UK surface waters can be explained by N deposition rates (Allott et al., 1995), particularly if the modifying influence of soil carbon content is taken into account (carbon-rich catchments are slower to respond to elevated N deposition, Evans et al., 2006). Surface water nitrate monitoring provides one of the most robust (and low-cost) measures of temporal change in N status at the ecosystem scale; UK data show that nitrate leaching in UK waters has not declined consistently over the last 20 years (Monteith et al., in prep) providing a strong indication that many semi-natural ecosystems remain saturated with N. This contrasts with data from some other deposition-impacted areas, such as Central Europe, where nitrate leaching has declined strongly in recent years (Oulehle et al., 2006). Relationships between current and/or cumulative N deposition rate and N leaching rate could be used to establish a highly relevant indicator, albeit one that is applicable mainly at larger rates of N deposition. The potential to obtain detailed temporal information through repeated surface water sampling represents an important advantage when compared to other methods, such as soil sampling, that can only provide less frequent snapshots. Conversely, the cost and difficulty of obtaining representative samples from a large number of surface water sites limits the utility of N leaching as a metric for examining spatial variations in N status.

Although N leaching is grouped here with midpoint indicators as it provides an indication that N saturation is advanced and there are strong risks to biodiversity endpoints, N leaching is also closely related to water quality (nitrate concentration) and so can also be seen as an endpoint indicator.

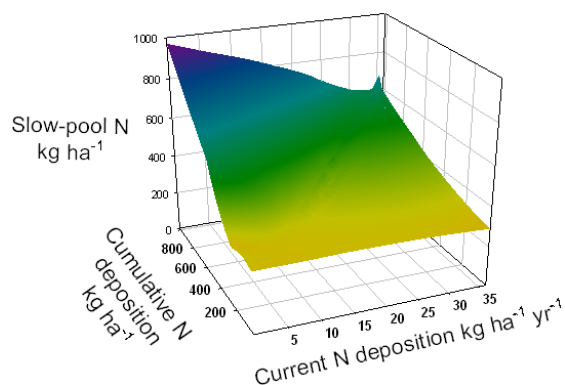
5.3.5 Stored N

The amounts of N stored within pools with different turnover rates are important indicators of likely change to N availability over different timescales. However, with the possible exception of readily-mineralisable N (see section 5.3.3), the measurement of these pools is difficult. Potentially, though, midpoint metrics that are relevant to medium- or long-term change could be derived from model outputs. The 'slow' pool in the MADOC model, as represented in Figure 9, is an example of this type of metric.

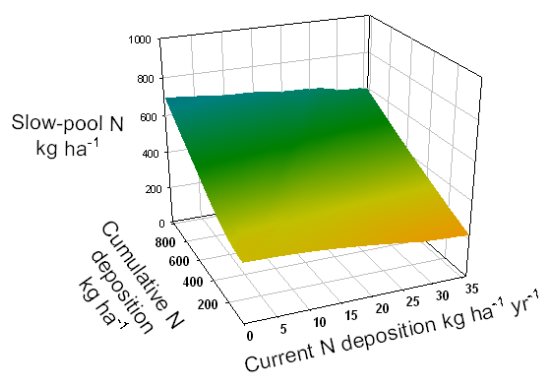
The size of a stored N pool with an intermediate turnover rate provides an indication of the amount of N likely to be released into more damaging, plant-available forms over the following 20-30 years, making it a highly relevant metric. Such a metric would be responsive to changes in current deposition and also to more gradual changes in cumulative deposition integrated over a preceding period of similar length (e.g. Figure 18). However, these advantages must be set against uncertainties as to choice of model, model structure, and parameters. Although empirical relationships are also uncertain, due to sampling error, confounding with other factors, and choice of statistical model for fitting the relationship, it is probably true to say that empirical relationships have wider acceptance among scientists. Confidence in atmospheric deposition models is greater than in dynamic soil models, in part because the underlying processes are better-understood and better supported by monitoring programs, and in part because the behaviour of biotic systems is inevitably harder to predict than that of abiotic systems. Confidence in soil models is likely to increase as more testing is done and models are shown to perform well across different environments and habitats. Currently, however, modelling studies are probably best seen as supporting the development of understanding rather than generating metrics for use in reporting.

Figure 18. Effect on N storage (in slowly turning over organic matter) of current deposition, and cumulative deposition during the previous 30 years, during periods of a) increasing N deposition, and b) decreasing deposition. Based on sensitivity analysis of the MADOC model (Rowe et al., submitted).

a) increasing deposition



b) decreasing deposition



5.3.6 Tissue N concentration

The concentration of N in plant tissue has been shown to reflect spatial patterns of N deposition (e.g. Rowe et al., 2008), and also to be highly responsive to changes in N deposition (e.g. Mitchell et al., 2004). This concentration can be expressed in terms of C/N ratio or % of dry weight, since the C concentration in plant tissue is relatively uniform. Although several studies have shown effects of N deposition rate on N concentration in vascular plants, it may be more promising to base a metric on

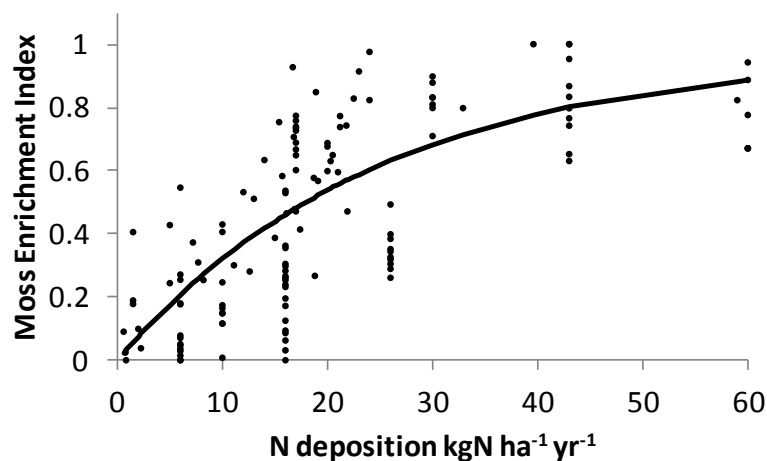
N concentration in mosses. This is because N in mosses, in particular ectohydric mosses that obtain little or no N from soil, is likely to be related to atmospheric deposition over recent years.

Another reason for proposing a metric based on moss tissue %N is that this has been studied over wide deposition gradients (e.g. Harmens et al., 2011). Moss tissue %N data (Figure 1.1 in that study) suggest that individual species respond to elevated N deposition by increasing their N content, but that different species may have a different characteristic N contents at any given N deposition level. There is also evidence that moss tissue %N reaches an upper ‘saturation’ value at high N deposition, above which detrimental consequences (increased N availability and species changes) are considered likely. Again, this saturation value may be species-specific. In light of this, we suggest a simple metric, termed the ‘Moss Enrichment Index’, whereby measured tissue %N is normalised to a value between 0 and 1 based on a species-specific equation:

$$\text{MEI} = (\%N_{\text{observed}} - \%N_{\text{minimum}}) / (\%N_{\text{maximum}} - \%N_{\text{minimum}})$$

Where $\%N_{\text{minimum}}$ and $\%N_{\text{maximum}}$ represent the lowest and highest (i.e. saturation) levels of tissue %N recorded for that species within national or international datasets. The MEI has the advantage of providing a directly measurable, single metric of N enrichment within the ecosystem, which can be expected to respond relatively rapidly to changes in N deposition, and which may provide an indication of ecosystem N status at lower N deposition levels, for which other biogeochemical measurements such as mineral N leaching may be ineffective. In principle, MEI values could be calculated by averaging results for several moss species present at a single site, and could be compared across sites with different moss species present (rather than relying on a single ‘indicator’ species). As an illustration, the tissue N concentration observations for three species of moss that were shown in Figure 3 were normalised to the scale from minimum to maximum concentration for the species, as outlined above, and a relationship to N deposition rate fitted (Figure 19). Some further collation and analysis of moss tissue %N data by species would be required to further develop and evaluate this method.

Figure 19. Effect of N deposition rate on Moss Enrichment Index, calculated by normalising the moss nitrogen concentrations presented in Figure 3 to a range from the minimum to maximum measured tissue N concentration for each species. The curve shown is an exponential increase to maximum, Moss Enrichment Index = $0.974 \times (1 - e^{-0.0403 \times N_{\text{deposition}}})$, fitted by minimising total absolute error.



The Moss Enrichment Index is promising as a midpoint indicator since moss nitrogen concentration is comparatively cheap and simple to measure, and is likely to be strongly indicative of recent N deposition to a site, at least within the range 0-25 kg N ha⁻¹ yr⁻¹. As well as directly indicating an aspect of habitat function, tissue N enrichment in mosses indicates that these species (which may

form a significant part of the biodiversity interest on a site) are at risk. Moss tissue N enrichment also provides a warning that effects on vascular plants are likely to already be occurring (in sensitive vascular plant species) or imminent.

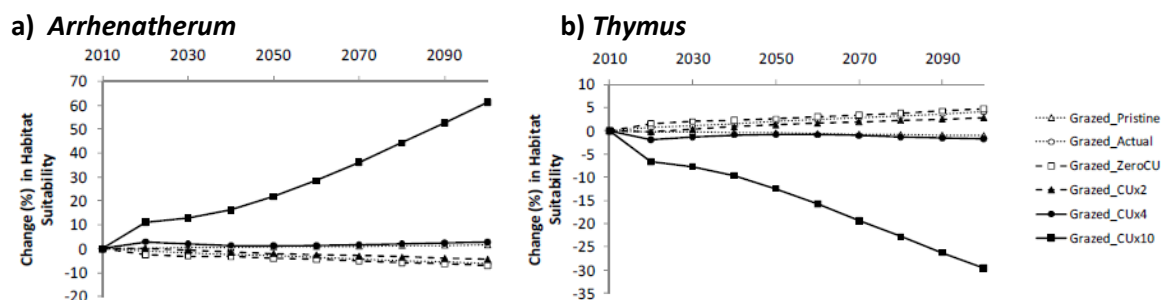
5.4 Endpoint metrics

5.4.1 Model-based and empirical metrics

Since the endpoint metrics proposed here are based on biological responses, similar considerations apply when calculating them based on empirical data (e.g. aggregate measures derived from quadrat data) or model outputs (e.g. aggregate measures derived from habitat suitability predicted for a set of species).

The floristic model that has been applied most in the UK to predict floristic responses to atmospheric pollution is GBMOVE (Smart et al., 2010), now implemented as MultiMOVE (Henry et al., submitted). This has been applied in analyses of N pollution scenarios, mainly to predict responses of habitat suitability for individual species (e.g. Figure 20). Empirically-based floristic models such as these are based on equilibrium or static relationships between environmental conditions and the occurrence of individual species, and take no account of biological delays. These may be considerable, if species are made locally extinct and have to re-colonise over large distances or re-invade habitats that have become dominated by competitive species. However, observed relationships between environmental conditions and occurrence can be a useful indicator of when a habitat has become suitable for a species.

Figure 20. Effects of a range of N deposition scenarios (see source for details) on temporal changes in habitat suitability for: a) *Arrhenatherum elatius*, a negative site condition indicator for dune grassland; and b) habitat suitability for *Thymus praecox*, a positive site condition indicator for dune grassland. From (Rowe et al., 2011b).



The niche models that make up MultiMOVE are strongly based on large empirical datasets, and predictions of habitat suitability for individual species in relation to the environmental axes that are included, such as mean 'Ellenberg N' score, have a high level of certainty. Uncertainties are introduced when using empirical relationships between mean 'Ellenberg N' and soil measurements such as mineralisable N (e.g. Rowe et al., 2011a), principally because the evidence-base regarding nutrient limitations to productivity in UK semi-natural habitats is sparse. Models of the dynamic response of plant-available soil N to N deposition introduce further uncertainty. However, the model chains applied in the UK are continually being improved, and represent the best available means of integrating biogeochemical and ecological knowledge to make predictions of floristic change. It will be useful to continue to explore methods for interpreting their outputs in terms of policy-relevant criteria, and establish metrics based on these outputs.

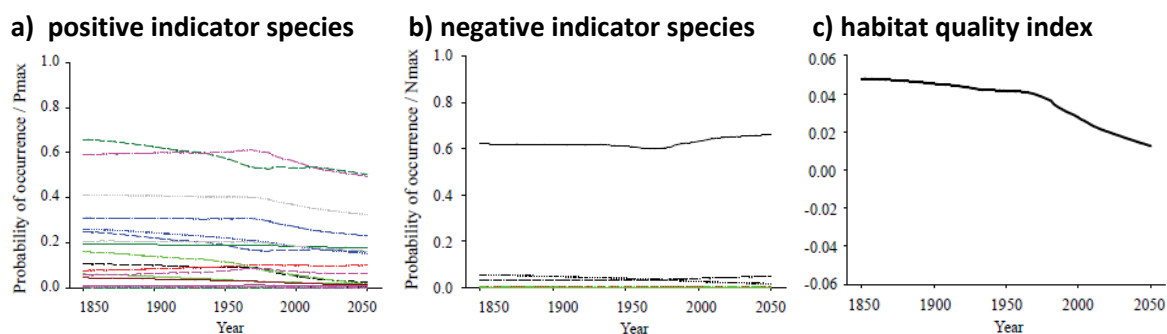
5.4.2 Occurrence and habitat suitability for individual species

Criteria used to define favourable condition for habitats often emphasise the occurrence and/or abundance of individual species. The habitat condition indicators set out for ‘Common Standards Monitoring’ (e.g. JNCC, 2006) are the most specific and concrete definitions currently available for favourable condition. For many habitats, lists are set out of species that indicate favourable or unfavourable condition. If target species can be defined for individual habitats, observations of species occurrence or predicted changes in habitat suitability for individual species could be used as metrics of N impacts. It should be noted that N-sensitivity does not *per se* imply importance to biodiversity endpoints, although in practice the long duration of N effects in the UK, and the combination of a strong relationship between plant height and threat status (Ken Thompson, *pers. com.*) with the increase in light competition caused by N enrichment, mean that N-sensitive species are likely to be of conservation concern.

As well as being relevant to definitions of habitat quality and biodiversity endpoints, individual species are relevant for many ecosystem functions and services, such as provisioning services (e.g. nitrogen-fixing species, and species that maintain pollinator populations) and cultural services (e.g. appreciation of bluebells).

Different methods have been proposed for aggregating data for individual species, from equal weighting of all species to assigning greater scores to species that are of greater importance for biodiversity or other endpoints. For example, Rowe et al. (2009) combined simple positive and negative scores for positive and negative indicator-species with predictions of habitat suitability to produce an aggregate indicator that was responsive to N deposition scenarios (Figure 21).

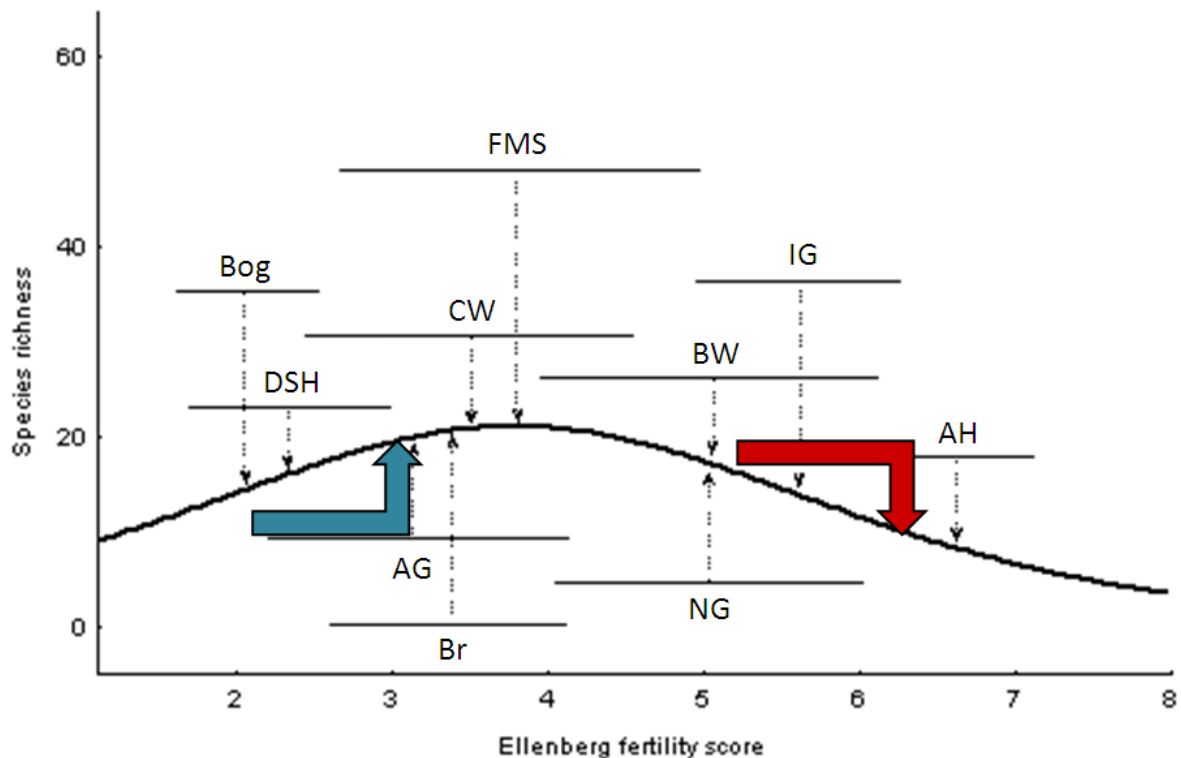
Figure 21. Effects of ‘Current Legislated Emissions’ N deposition scenario on species used as: a) positive and b) negative site condition indicators, and on c) an aggregate habitat quality index. From Rowe et al. (2009).



5.4.3 Species richness

The use of species richness as a metric of habitat quality is discussed in section 5.4.3. This metric is most suitable for habitats where species richness is considered an indicator of favourable condition, such as grasslands. However, extra but untypical species, such as ruderal, invasive or alien plants, increase species richness but are usually considered to indicate a decline in condition. Metrics such as the Shannon-Weaver Index and Simpson’s Index, that include an assessment of evenness (the degree to which the assemblage is dominated by a small number of species), share with species richness the problem that they are not well-related to concepts of favourable condition for some habitats. Models for directly predicting species-richness are under development (Simon Smart, *pers com.*), which take into account the likely increase in species-richness with fertilisation of low-productivity ecosystems and decrease in high-productivity ecosystems (Figure 22).

Figure 22. Adapted from Smart et al. (2003). Species richness in relation to Ellenberg 'N' (fertility) score in all UK 1998 Countryside Survey areal plots (fitted curve). The length of horizontal lines represents the range in Ellenberg N score within each Broad Habitat (BW, Broadleaved woodland; CW, Conifer woodland; AH, Arable & Horticultural; IG, Improved grassland; NG, Neutral grassland; AG, Acid grassland; Br, Bracken; DSH, Dwarf Shrub Heath; FMS, Fen, Marsh & Swamp; Bog). The vertical position of these lines is arbitrary. Solid arrows represent likely direction of change in species richness with N enrichment.



Species richness has been shown to be negatively correlated with current N deposition rate in UK acid grasslands (Stevens et al., 2004), and in UK infertile grasslands and heathlands (Maskell et al., 2010). Such statistical relationships can be used to derive expected species richness from current deposition scenarios. For example, a correlation between species richness and current deposition for calcifuge grasslands in the Atlantic region of Europe (Stevens et al., 2011a) was recently used to generate European maps of the shortfall in potential species richness due to N pollution (Hettelingh et al., 2013). Using species richness as a metric has benefits in terms of translating N deposition scenarios into a term that is widely understood, and easily related to conservation targets.

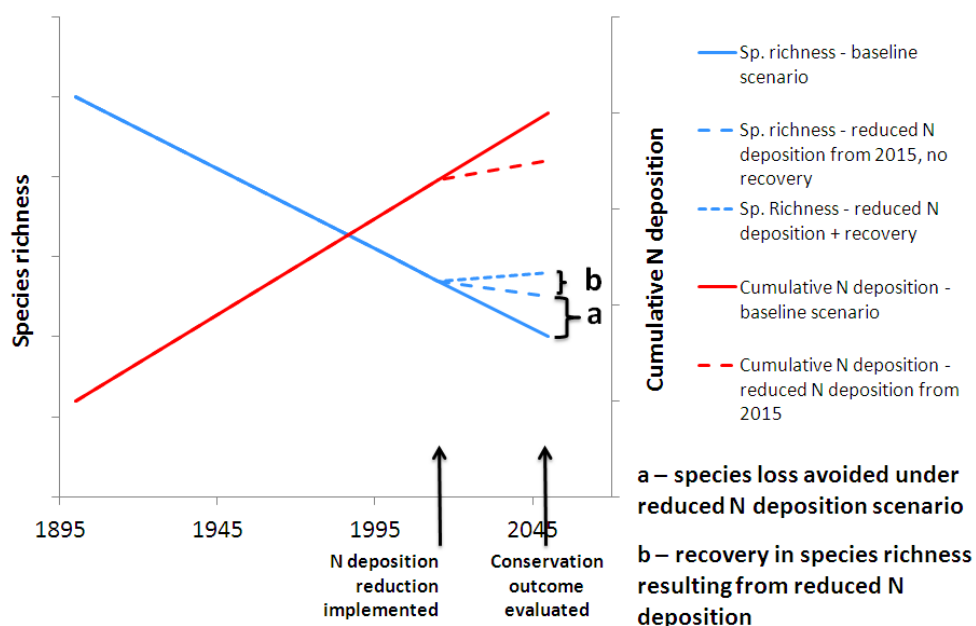
The approach may not be applicable in all habitats, since relationships between N deposition and species richness are less clear, and indeed high species richness may not indicate favourable conservation status, in habitats that are characterised by low vascular plant diversity such as heathlands and bogs. Another criticism that could be levelled at scenario analyses based on species-richness vs. deposition relationships is that these are based only on current deposition, and take no account of delays in chemical and biological responses. This is unrealistic in view of the large amounts of N that accumulate in ecosystems after many years of chronic pollution N, and delays in biological responses such as re-colonisation by previously-lost species.

A similar approach could be applied to assessing effects of cumulative deposition, if relationships can be established between species richness and cumulative deposition. A meta-analysis of experimental studies (Phoenix et al., 2012) failed to clearly establish such a relationship, but re-analysis of survey data in relation to cumulative deposition (however calculated – see Section 3.2) might reveal usable relationships.

For communities dominated by lichens and bryophytes which have little interaction with soil N, especially epiphytic and epilithic communities, the empirical relationship between current N deposition and species richness might be a better basis for predicting the impacts of N deposition scenarios. In this case, average air concentrations and deposition over the past 3-5 years may prove a suitable metric of exposure in relation to epiphytic chemical conditions, although it is expected to take longer to achieve full recolonisation on recovery after pollution decrease.

Benefits of scenarios of decreased N deposition could then be demonstrated by reference to empirically derived relationships between species richness and cumulative N deposition (Figure 23). Existing survey based studies are available for several habitats showing a negative relationship between species richness and cumulative N deposition. The impact of a decrease in N could be calculated as the difference between the two points on the species richness-N accumulation curve which would be reached on a given future date for two contrasting N deposition scenarios. The future date for comparison would need to be sufficiently far into the future for measureable differences to be shown, but could be selected to represent long term conservation targets. Such a metric would be a measure of damage (species loss) avoided by the decreased rate of N accumulation and would represent a minimum benefit of deposition decreases since it takes no account of community recovery under decreased N deposition. This metric would be most applicable to communities dominated by higher plants which are strongly influenced by the accumulation of N in the soil. There may be much variability between habitats, but recovery timescales are probably long, if indeed recovery occurs at all in some cases.

Figure 23. Species loss avoided through decreases in N deposition rate, based on hypothetical relationship between cumulative N deposition and species richness.



The community's ability to recover species under decreased N loading becomes an even more important component of the measurement of benefits from decrease scenarios. If the recovery component is excluded, community composition is simply representative of the maximum N deposition rate which the community has experienced. Assessing the benefits of N decreases for these communities in particular therefore requires knowledge of recovery rates. Existing experimental data on the impacts of N decrease in these communities has tended to focus on physiological or tissue chemistry measures or has used reciprocal transplant techniques, which avoid

the issue of lags in species composition response due to re-colonisation delays. Further experimental evidence on the rates community composition recovery would be needed to give realistic estimates of the benefits of N deposition decreases.

5.4.4 Grass/forb ratio

In grasslands, the ratio of cover of grasses and forbs (i.e. non-grass herbs) was shown to be very responsive to N deposition load (Emmett et al., 2011; Stevens et al., 2011c). This relationship could be used to develop a responsive metric for these habitats.

5.4.5 Mean 'Ellenberg N' score and other plant trait averages

Potentially-useful metrics can be derived from mean values for the traits of present species (and therefore also from predicted species composition), traits such as:

- growth-form (e.g. shrub, sub-shrub, herb)
- physiology (e.g. typical specific-leaf-area, typical height)
- taxonomic group (e.g. graminoid, forb)
- ecological strategy (e.g. competitive, stress-tolerant)
- environmental preference (e.g. Ellenberg N or fertility scores, Ellenberg R or pH score)

Several of these trait-means were shown to be correlated with N deposition in a study based on large-scale survey data (Stevens et al., 2011c). Trait-means likely to be sensitive to N deposition in at least some habitats were summarised by Emmett et al. (2011) as: grass/forb cover ratio; mean Ellenberg R score; mean Ellenberg N score; mean typical canopy height; and mean typical specific leaf area. Three of these (Ellenberg R, Ellenberg N and height) are already used within MultiMOVE to define environmental conditions. The functions used to predict trait-means from soil conditions (e.g. Rowe et al., 2011a) could be used simply to predict changes in mean Ellenberg N without assessing implications for individual species.

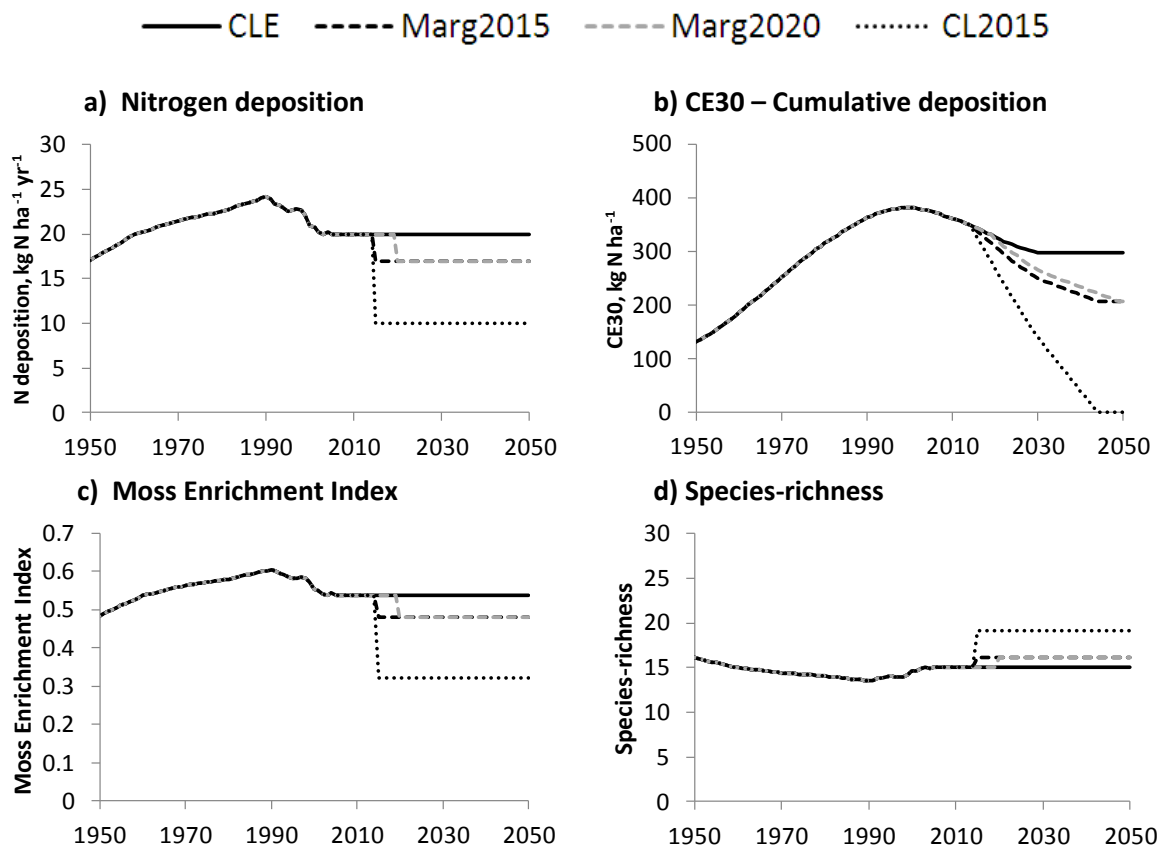
Sutton et al. (2009) proposed an index derived from scores assigned to lichen species on the basis of their preference (or not) for acid and N-rich conditions. This “acidophytes / nitrophiles index” could be applied as an endpoint metric.

5.5 Example applications of metrics to scenarios

Simple scenarios were developed to illustrate example pressure, midpoint and endpoint metrics. These are all based on deposition calculated using the FRAME model under the Current Legislated Emissions scenario for an experimental site on the Migneint SAC in North Wales. Effects of this scenario on example metrics were compared with effects of a scenario in which deposition was reduced to the critical load for wet heath, $10 \text{ kg N ha}^{-1} \text{ yr}^{-1}$, from 2015. To explore effects of early and delayed marginal reductions, two further scenarios included a decrease in deposition of $3 \text{ kg N ha}^{-1} \text{ yr}^{-1}$, in either 2015 or 2020.

Three response metrics were chosen for illustration (Figure 24). The cumulative deposition in excess of the critical load (for wet heath) over the previous 30 years, CE30, was calculated from deposition data. A 'Moss Enrichment Index', indicating the expected N concentration in moss tissue as a proportion of the observed minimum to maximum range, was calculated from current deposition using the transfer function that was illustrated in Figure 19. Similarly, the current deposition was used to calculate an expected species richness, using the equation presented in Hettelingh et al. (2013) i.e. number of species = $24.4 \times e^{-0.0244(N_{\text{dep}})}$ where N_{dep} is the N deposition rate in $\text{kg ha}^{-1} \text{ yr}^{-1}$. This equation was developed for grasslands, which are not the main conservation interest in the Migneint SAC, but serves for illustration.

Figure 24. Illustrations of prototype metrics. Responses to nitrogen deposition (a) calculated under four scenarios: CLE = Current Legislated Emissions; Marg2015 = reduction of $3 \text{ kg ha}^{-1} \text{ yr}^{-1}$ from 2015; Marg2020 = reduction of $3 \text{ kg ha}^{-1} \text{ yr}^{-1}$ from 2020; CL2015 = reduction to the critical load for wet heath, i.e. $10 \text{ kg N ha}^{-1} \text{ yr}^{-1}$, from 2015) of: b) CE30, cumulative deposition in excess of the nitrogen critical load within preceding 30 years; c) Moss Enrichment Index, i.e. predicted moss tissue N concentration as a proportion of the maximum observed for the species; d) Species-richness, i.e. predicted number of species.



The pressure metric illustrated in Figure 24, CE30, does respond to changes in N deposition, but the 30 year lag in the calculation method means the response is not abrupt. As explained in Sections 3.2 and 5.2.3, this is a realistic reflection of likely delays to recovery of chemical conditions due to the persistence of deposited N in soil. The midpoint metric, MEI, and the endpoint metric, species-richness, are instead calculated as functions of current deposition. This results in an instantaneous response to changes in deposition rate, which is not realistic. Delays in chemical recovery could be allowed for by using a dynamic modelling approach (see Section 2.7), although some development would be necessary before the metrics illustrated here could be simulated with confidence. Generic vegetation models could be applied to simulate dynamics of moss tissue N, but would need to be adapted and tested. Species-richness is not an output from current dynamic models, and its simulation would require relationships to be developed between a modelled (and preferably measurable) quantity such as soil N availability and species-richness.

Where a relationship can be established through regression or dynamic modelling between N deposition and a response metric, it will be possible to calculate the value of the metric for a given location at a point in time under a particular scenario, provided sufficient data are available. This will allow scenarios to be illustrated using maps of values for the metric or metrics, once agreement is reached on the calculation method. Habitat-specific aspects of the relationship, for example using a different critical load or a different transfer function to species-richness for a particular habitat,

imply that the habitat at the location needs to be known. Where dynamic models are used, other site-specific information may be required. Data suitable for these applications have been collated, mainly at 1 km² resolution, by the National Focus Centre <http://cldm.defra.gov.uk/>, although some additional method-specific data may be required to implement metrics.

5.6 Multiple and multi-criteria metrics

It is probably not be possible to capture all aspects of pressure and impact within a single metric. It may therefore be appropriate to either report several distinct metrics, or an aggregate metric that combines different criteria.

Calculating and reporting multiple metrics would have the advantage of separating pressure from response, and separating different aspects of response. Metrics could be reported separately and in different ways, as for current reporting of % Area Exceedance and Average Accumulated Exceedance. Alternatively, metrics could be reported together as a list or table, or in combination using a 'cobweb' diagram in which the magnitude of the different metrics is illustrated on different spokes of a wheel.

Combining different indicators into one or a small number of multi-criteria metrics may be a way of circumventing debates around what is the most appropriate indicator, without requiring the users of metrics to assess the relative importance of several reported aspects. However, this relative importance must still be assessed when deciding on the weights assigned to the different components. Deciding these weights with a small group of participants may obscure genuine disagreements over the relative importance of different aspects of ecosystem function and structure, and the reasoning behind them. In the opinion of Suter (1993), "[multi-criteria indices] have no meaning; they cannot be predicted, so they are not applicable to most regulatory problems; they have no diagnostic power; effects on one component are eclipsed by responses of other components; and the reason for a high or low index value is unknown. Their only virtue is that they reduce the complex array of ecosystem responses to various disturbances to one number with a reassuring name."

Reporting of multiple or multi-criteria metrics should be considered in the light of requirements of policy decision-makers. Metrics are more likely to be effective for communication if they are easily understood and few in number.

5.7 Recommended metrics

The metrics most suitable for evaluating benefits of marginal reductions in N deposition are summarised in Table 6.

Midpoint and endpoint metrics derived from current or cumulative deposition using static relationships take no account of delays to chemical and biological response. Delays to chemical response can be incorporated by deriving these metrics from the outputs of dynamic models, where these have been developed. Models of delays to biological response (based on presence in the local species pool and dispersal/colonisation/extinction rates) have been explored but are not yet reliable. The table gives an indication of how long it would take to fully operationalise each metric, based on static or dynamic modelling approaches, assuming that one person is employed full-time on the development. Quantifying the metric at national scale, mapping its distribution, scenario analyses and other applications would take further time.

We consider the most appropriate metrics in each category, taking into account ecological and policy relevance, robustness of underlying principles and evidence, and readiness, to be:

Pressure metric

'CE30': Cumulative exceedance of habitat-specific critical load for preceding 30 years.

Midpoint metric

'MEI': Moss Enrichment Index.

Endpoint metric:

'HQI': Habitat Quality Index.

Of these, only CE30 (cumulative exceedance of habitat-specific critical load for preceding 30 years) is currently fully operationalised. The MEI could be derived from existing data, but additional observations, particularly from habitats that are currently under-sampled, would make the relationship more robust. Operationalising the HQI would require considerable consultation with conservation agencies, but such an index would potentially be a better reflection of biodiversity endpoints such as favourable conservation status than are simple diversity metrics such as species-richness.

Table 6. Recommended metrics, classified by Type: P = Pressure; M = midpoint; E = endpoint. The final two columns give an indication of how near to readiness these metrics are when derived using static relationships and dynamic models, respectively.

Metric	Type	Appropriate for	Recommended calculation method	Evaluation	Timescale for delivery (static)	Timescale for delivery (dynamic)
Average Exceedance ¹	P	All habitats. All deposition rates above CL_{nutN} .	Exceedance of CL_{nutN} , averaged across N-sensitive habitats within a grid-square, weighted by habitat area.	Pros: responsive and simple; ready to use. Cons: takes no account of impact delays.	available	N/A
Cumulative exceedance	P	All habitats. All deposition rates.	Integrated exceedance of (habitat-specific) CL_{nutN} , over the preceding 30 years (for most habitats) or 3 years (for epiphytic/epilithic sub-habitats).	Pros: responsive; well-related to timescale of impacts and to agreed definitions of damaging deposition rate. Ready to use. Cons: timescales based on expert judgement.	available	N/A
Moss Enrichment Index (MEI)	M	Habitats with mosses (i.e. most habitats). Deposition rates up to $25 \text{ kg N ha}^{-1} \text{ yr}^{-1}$.	Convert current ² deposition to predicted moss tissue %N using relationships from Harmens et al. (2011), normalised to the range in %N observed in the moss species (see section 5.3.6).	Pros: well-correlated with (lower) deposition rates, easily measurable, useful 'early warning' metric.	3-6 months	6-12 months
Stored N	M	Habitats with soil ³ . All deposition rates.	Calculate 'slow' N pool in response to time-series of deposition using e.g. the N14C model (Tipping et al., 2012).	Pros: illustrates well a stock of N which places the habitat at risk; modelled values are easily upscaled. Cons: best measurement method remains uncertain.	N/A	3-6 months
N leaching rate	M/E	All habitats. Deposition rates above $25 \text{ kg N ha}^{-1} \text{ yr}^{-1}$.	Develop relationships between current ² deposition and N leaching rate.	Pros: well-correlated with (higher) deposition rates; indicates advanced damage.	1-3 months	3-6 months

Metric	Type	Appropriate for	Recommended calculation method	Evaluation	Timescale for delivery (static)	Timescale for delivery (dynamic)
Mean 'Ellenberg N'	M	Habitats where relationship with deposition has been demonstrated ⁴ . All deposition rates.	Convert current ² deposition to predicted mean Ellenberg N using relationships from Stevens et al. (2011c), and/or predict from relationships with soil properties (Rowe et al., 2011a) to account for chemical delays.	Pros: well-related to theoretical and observed effects of N on species-assemblages; can be modelled and also easily measured. Cons: meaning not immediately apparent.	1-3 months	6-12 months
Species richness	E	Grasslands, potentially other habitats such as mires. All deposition rates.	Convert current ² deposition to predicted species richness using relationships from e.g. Maskell et al. (2010).	Pros: Readily understood. Cons: not applicable to all habitats.	1-3 months	6-12 months
Habitat quality index	E	All habitats. All deposition rates.	Aggregate habitat suitabilities calculated using e.g. MADOC-MultiMOVE in response to time-series of deposition, using appropriate scorings for target and other species.	Pros: potentially better-related to favourable conservation status than is species-richness. Cons: and careful and transparent definition.	3-6 months	6-12 months

¹ Proposed alternative name for Average Accumulated Exceedance.

² If statistical relationships can be established between cumulative exceedance, as calculated over an appropriate threshold and period, and these midpoint and endpoint metrics, these relationships may form a more appropriate basis than current deposition for calculating these metrics.

³ Dynamic models could also be adapted to simulate N dynamics in epiphytic / epilithic habitats.

⁴ According to Stevens et al. (2011c) these are acid and calcareous grassland, heathland, and upland (but not lowland) bog.

6. Evidence gaps

Critical areas of uncertainty remaining in our current knowledge need to be addressed, including fundamental questions over ecosystem responses to marginal decreases in N deposition, capacity to simulate and predict changes, and evidence for deciding between and supporting reported metrics. Evidence gaps are listed below, with an indication of priority on a scale of A (highest priority) to C. Co-funding opportunities have been identified where possible.

For several of the evidence gaps there are opportunities to exploit remote sensing, near sensing (e.g. spectral analysis of photographs taken at ground level or from model aircraft) and wireless sensors for real-time monitoring of productivity, nitrogen concentrations and canopy structural changes. Remote sensing can be oversold, e.g. there is limited ability currently to differentiate between some habitat types, but real-time early warning systems of change could be developed with new sensor and computing technologies. It is particularly important to detect changes that do not match model predictions and may indicate unexplained shifts in ecosystem function. These can be explored using methods explored through NERC's Environmental Virtual Observatory, with new initiatives such as Big Data and NERC/TSB calls for Environmental Data could be opportunities for co-funding. Overall, increased accessibility of data and models to the community to improve our real-time monitoring and predictive capability has the potential to increase efficiency and effectiveness of air quality research and advice.

Table 7. Gaps in the evidence required to develop metrics for evaluating the benefits of reductions in nitrogen deposition, prioritised A (highest priority) to C.

Priority	Evidence gap	Purpose of research	Requires	Potential timescale for delivery
A	Ecosystem responses (e.g. species richness, moss %N, leaching, species prevalence) to cumulative N deposition calculated over a representative threshold and period, e.g. above CL_{nutN} for preceding 30 years. Analyses could be directed at establishing response functions to a pre-defined metric of cumulative N, or towards establishing which metric of cumulative N best explains response data.	To assess the timescale over which deposition changes affect midpoint and endpoint metrics.	Statistical exploration of datasets that have already been analysed in relation to current and total cumulative deposition.	3-6 months
A	Effects of N deposition on plant productivity . Changes in vegetation growth and litterfall are critical effects of N deposition, yet there is little evidence for how these responses to N are affected by other limitations to plant growth in UK semi-natural habitats. Productivity limitations vary with season, habitat, N deposition history, soil type and other factors, and the main limitation can be light, temperature, moisture, waterlogging, N, P, or other nutrient elements.	To determine where (in terms of habitat, and deposition history) N acts as a eutrophying pollutant and where it does not.	Simple, multiple-site experiments to assess productivity-limiting factors at sites selected according to a robust conceptual and modelling framework. Some interactions affecting productivity are being assessed within the NERC Macronutrient programme (e.g. Turf-2-Surf and Long-Term Large-Scale projects).	2-3 years
B	Ecosystem responses to reductions in N deposition (including small decreases) in different habitats and across the range of cumulative N deposition load, to quantify spatial and temporal variability. New / revised experimental studies are required, and should be situated within a conceptual and predictive modelling framework. Metrics proposed in this report should be measured at the baseline and end of each experiment (as a minimum).	To develop understanding of effects of decreases in deposition, and assess whether these are a simple reversal of effects of increases.	Experiments with N decrease treatments. Some existing N-addition experiments could be modified, but rain-replacement experiments are likely to be more informative about effects of decreased pollution.	2-15 years

Priority	Evidence gap	Purpose of research	Requires	Potential timescale for delivery
B	Effects of deposition on plant-available N , including pools that become available over short- to medium-term timescales.	To determine link between current / cumulative deposition and plant exposure to N.	More measurements of N availability in soil and intercomparisons of methods.	1-3 years
B	Model testing and scenario exploration with available biogeochemical models and biogeochemical-hydrological-floristic model chains. The potential of the models that have been developed is underused. These models could be applied to scenarios at different scales, predicting effects of multiple drivers on different indicators, but the models require further testing to quantify the uncertainty that should be associated with their predictions.	To develop capacity for and confidence in scenario analyses.	More testing of models against experimental and survey data. Comparisons of the performance of alternative models (some work on this topic is continuing under the EC ECLAIRE project). Making models more accessible for use by scientists, stakeholders and site managers.	1-3 years
B	Accessibility to data, models and tools	To improve efficiency, avoid duplication and repetition and increase transparency in the translation through to policy.	An accessible platform to host and run models and enable links to existing data portals with user-friendly web interfaces. Small subset of models in the first instance of relevance to some key ecosystem services impacted by air quality. User-friendly interfaces and past libraries of past runs.	1-2 years
C	Chemical recovery delays in epiphytic and epilithic plant and lichen communities.	To determine the appropriate period to calculate cumulative deposition for, for these sub-habitats.	Experimental and modelling studies.	2-3 years

Priority	Evidence gap	Purpose of research	Requires	Potential timescale for delivery
C	Effect of N deposition on sensitivity to acute events and tipping points, particularly in combination with other environmental drivers. Increased sensitivity could allow fundamental shifts in ecosystem structure and function in response to modest events. The theory is well-developed but there is as yet limited evidence in terrestrial systems for this 'conditioning' of the system.	To assess how other pressures might affect the choice of a N effects metric.	New approaches to exploring this area need to be explored. Links to tree-health initiatives are one possibility for co-funding.	2-3 years

7. Conclusions

This report has reviewed the best currently-available evidence in relation to effects on ecosystems of N deposition and of decreases in deposition rate. However, considerable uncertainties remain in relation to both empirical and modelling evidence. The recommendations in this report are made according to best current knowledge, exploiting a number of previous syntheses and reports for JNCC, Defra and other stakeholders (e.g. Emmett et al., 2011; Evans et al., 2011, 2012; Stevens et al., submitted; Stevens et al., 2008; Stevens et al., 2011c; UKREATE, 2010).

The persistence of N in ecosystems, and delays in chemical response to changing N deposition, argue for the use of cumulative deposition as a metric of pressure on habitats. Calculating cumulative deposition over a preceding period similar to the mean residence time of N in the ecosystem would result in an ecologically relevant metric. Such a metric would decrease with reductions in current deposition (in contrast to total cumulative deposition since a fixed date) in a way that reflects but does not over-represent the benefits of current reductions. An integration period of 30 years is probably the most appropriate for soil-based ecosystems. Sub-habitats with less capacity to retain N, such as epiphytic assemblages, can respond more rapidly to change and cumulative exceedance over the previous three years may be more appropriate for these components. Different habitats may vary in their sensitivity to N deposition, as shown by the variation in empirical CL_{nutN} among habitats. This differential sensitivity could be accounted for in the calculation of cumulative deposition by accumulating only the deposition above the agreed UK value for empirical CL_{nutN} (Hall et al., 2011). We therefore recommend 'CE30', the cumulative exceedance of habitat-specific critical load for preceding 30 years, as a suitable pressure metric for all habitats.

It may be more difficult to reach agreement on response metrics, whether for midpoint / function indicators, or endpoint / service indicators, than on a pressure indicator. Different authors emphasise different chemical and biological responses, and may be influenced by their academic specialism, their preferred epistemology and value system, and by the habitats and degree of N saturation that they are most familiar with. However, following a review of potential midpoint indicators, we consider that most promising are tissue N in mosses (for low-deposition systems) and N leaching (for high-deposition systems). These indicators vary in their responsiveness at different stages of ecosystem saturation with N, and are complementary in that moss tissue N concentration increases with reasonable consistency in the range $0-25 \text{ kg N ha}^{-1} \text{ yr}^{-1}$, but not above this rate (Figure 3) whereas N leaching is usually around zero until N deposition reaches around $25 \text{ kg N ha}^{-1} \text{ yr}^{-1}$, after which there are increases in N leaching (Figure 5). We recommend exploration of the relationships between observations of these quantities and current or cumulative deposition, with a view to operationalising them as metrics.

Biological responses also occur at different levels of N saturation, as illustrated by declines in prevalence of individual plant and lichen species observed at both low and high deposition rates (Stevens et al., 2011c). Relationships between species-richness and deposition could also be used to fairly rapidly develop an indicator relevant to biodiversity value, although this would mainly be applicable to grasslands. Aggregate indicators such as species richness or grass/forb cover ratio do not always capture N impacts, since they may not be affected by the replacement of N-sensitive species by N-insensitive ones. The mean 'Ellenberg N' score (see Section 5.4.5) is probably more sensitive to this replacement, but care is needed with interpretation since mean 'Ellenberg N' may be influenced by other environmental drivers. In fact species respond individually to N, and it is not possible to generalise responses for groups such as higher plants or mosses, or even within a genus. It is therefore important to define which species are considered important to the structure of individual habitats. For all habitats, methods for aggregating responses of individual species, taking into account their relevance and importance for the habitat. This work is likely to take place soon, since DEFRA recently issued a Call for Tender (AQ0828) on this topic.

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