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# Long-term impacts of nitrogen deposition on coastal plant communities☆

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ABSTRACT

Nitrogen deposition has been shown to have significant impacts on a range of vegetation types resulting in eutrophication and species compositional change. Data from a re-survey of 89 coastal sites in Scotland, UK, c. 34 years after the initial survey were examined to assess the degree of change in species composition that could be accounted for by nitrogen deposition. There was an overall increase in the Ellenberg Indicator Value for nitrogen (EIV-N) of 0.15 between the surveys, with a clear shift to species characteristic of more eutrophic situations. This was most evident for Acid grassland, Fixed dune, Heath, Slack and Tall grass mire communities and despite falls in EIV-N for Improved grass, Strand and Wet grassland. The increase in EIV-N was highly correlated to the cumulative deposition between the surveys, and for sites in south-east Scotland, eutrophication impacts appear severe. Unlike other studies, there appears to have been no decline in species richness associated with nitrogen deposition, though losses of species were observed on sites with the very highest levels of $SO_x$ deposition. It appears that dune vegetation (specifically Fixed dune) shows evidence of eutrophication above 4.1 kg N $ha^{-1}$ $yr^{-1}$, or 5.92 kg N $ha^{-1}$ $yr^{-1}$ if the lower 95% confidence interval is used. Coastal vegetation appears highly sensitive to nitrogen deposition, and it is suggested that major changes could have occurred prior to the first survey in 1976.



## 1. Introduction

Atmospheric deposition has been identified in many studies as an important driver of vegetation change and species loss (De Schrijver et al., 2011). Impacts can arise from the acidification of systems (e.g. Stevens et al. 2009) or from the eutrophication of systems as a result of nitrogen inputs that exceed critical loads (Bobbink et al. 2010). Eutrophication generally allows more competitive, nutrient demanding species to dominate at the expense of those typical of infertile vegetation, which typify many semi-natural communities of high conservation value (Maskell et al. 2010).

There are clear patterns for species richness along gradients of nitrogen deposition. For example, species richness is lower per unit area where nitrogen deposition is higher for acid grassland (Stevens et al., 2004; Duprè et al., 2010), for coastal vegetation (Plassmann et al., 2009; Remke et al., 2009), for heathland (Southon et al., 2013) and across all semi-natural habitats (Field et al., 2014). Analysing across habitats within the UK also indicated that species richness was lower in areas with higher nitrogen deposition; for acid grassland and heathland it appeared that acidification as opposed to eutrophication was responsible for species loss, but calcareous grassland showed evidence of eutrophication (Maskell et al. 2010). This pattern is replicated for a wide range of ecosystems including alpine, heathland and forests (Bobbink et al. 2010).

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Focussing on species ranges across the UK showed that patterns of species loss were correlated with nitrogen deposition (McClean et al. 2011).

The majority of studies that have demonstrated the impacts of atmospheric deposition on vegetation have been signal attribution studies. These focus on gradients of deposition and correlate one measure of the parameter of interest (e.g. species richness, functional trait or indicator value) with environmental factors including the levels of deposition (e.g. Stevens et al. 2004). In effect they become space-for-time substitutions (Duprè et al., 2010). This type of approach has been successful, especially where gradients of deposition are investigated that are not confounded with other large-scale drivers of patterns such as climate (Maskell et al. 2010). However, they are not appropriate for rarer vegetation types (lower power unless specifically sampled) or spatially restricted ones (short gradients). An alternative approach for these types of situations is to use re-survey studies (e.g. Duprè et al., 2010) which are statistically more powerful as they take advantage of change at specific locations and which can be already focussed on rare or spatially restricted vegetation types.

There are a number of different types of metrics available to understand vegetation change as a result of the action of environmental drivers. Many studies have focussed on species richness or diversity changes as a direct measure of biodiversity change (e.g. Stevens et al. 2004). However, species richness may be affected by a range of other drivers such as succession – early successional changes increasing richness and later ones potentially reducing it – or by changes in management. Plant functional traits are an alternative metric, but it appears those that are sensitive to eutrophication in particular, such as Specific Leaf Area or Leaf Dry Matter Content are also sensitive to climate and to management (Pakeman, 2013; Garnier et al. 2004). One type of measure that is independent of many of the other potential drivers is indicator values. These are exemplified by Ellenberg Indicator Values (EIV, Ellenberg, 1988) which have been derived from the general response of plant species to a series of key environmental gradients. Whilst indicator values for nitrogen (N) and soil pH (Reaction R) are affected by agricultural management, fertiliser and lime use respectively, they can be used to assess vegetation change in a functional manner that is robust to the impacts of succession and that can be attributed to a single driver of change. Ellenberg's N can also be seen as a general indicator of fertility rather than simply a nitrogen indicator (Hill and Carey, 1997).

This paper sets out to assess the cumulative impact of atmospheric deposition on coastal vegetation in Scotland, UK. These vegetation types are necessarily spatially restricted and include habitats restricted to small areas, such as machair. This calcareous, coastal grassland is restricted to the western coasts of Scotland and Ireland and its global cover amounts to only 25 000 ha (Angus and Dargie, 2002). Coastal vegetation is also largely nutrient poor and considered threatened by eutrophication (Bobbink et al., 2003; Jones et al., 2004). The study uses a re-survey approach to assess vegetation change and Ellenberg Indicator Values to try and apportion the functional change in the vegetation to atmospheric deposition. Specifically it tests three hypotheses: Hypothesis 1: the impacts of atmospheric deposition are detectable in coastal vegetation. This covers the continuing impact of eutrophication and the potential for recovery from acid deposition as a result of falling acid deposition (NEGTAP, 2001). Hypothesis 2: the degree of impacts is directly affected by the cumulative deposition between the surveys. Hypothesis 3: that the impacts of deposition may be mediated by the ability of the soil to buffer deposition impacts. For instance, calcareous grasslands appear to be less impacted by the acidifying component of nitrogen deposition than acid grasslands or heathland (Maskell et al. 2010).

## 2. Materials and methods

### 2.1. Survey data

Vegetation compositional data were available from two periods for all dune and machair systems of consequence within Scotland (Fig. 1). The first data were taken from the Scottish Coastal Survey that was carried out from 1975 to 1977 (Shaw et al., 1983); referred to as the 1976 survey as the bulk of the surveying was done in this year. The second survey was carried out between 2009 and 2011, and is referred to as the 2010 survey. One site was surveyed in 2013 due to issues with safe access. The second survey visited 89 out of the original 94 sites and repeated 2532 out of 3783 quadrats with vegetation records from the 1976 survey. Resources were not available to repeat the whole survey, so within a site quadrats were repeated in a random order to avoid bias in resampling different habitats. The analysis here was restricted to 2409 quadrats to exclude rare habitats and quadrats with no vegetation (usually open sand) in either survey.

The second survey repeated the methods used in the original survey (Shaw et al., 1983) and involved the visual estimation of cover of all higher plant species (nomenclature follows Stace, 2010), plus the cover of other classes such as bryophytes, lichens, litter and bare ground within 5 m × 5 m plots. Quadrats were relocated using GPS with British National Grid co-ordinates derived from digitising original sample points marked on 1:10000 maps. Initial testing suggested accuracy was in the order of ±10 m of the original position, based on the location of fixed features in relation to position of some quadrats. Relocation was aided by a summary of the 1976 vegetation and quadrats were not re-surveyed if surveyors were not confident that the vegetation change between the two dates was possible. Thus the degree of change is potentially estimated more conservatively than the true degree of change, but this approach has been shown to be effective in this type of resurvey study of non-permanent quadrats (Kopecký and Macek, 2015; Ross et al., 2010). This type of re-survey has been shown to deliver robust results of vegetation change (Chytrý et al., 2014). Species cover data (percentage cover) were converted into Species richness and Shannon–Wiener diversity scores.

As the two surveys were 34 years apart, climate change has occurred, with the averages of the two 15 year period preceding each survey (averaged over the 5 km grid squares where samples had been taken), showing and increase in rainfall from 1059.8 mm to 1109.9 mm, an increase in summer mean temperatures (June, July August) from 12.60 °C to 13.39° and an increase in winter mean temperatures (December, January, February) from 4.48 °C to 5.06°. However, changes were relatively uniform across the country (Pakeman et al., 2015).

### 2.2. Soil data

Soil parameters were characterized on one 5 cm diameter core of 5 cm depth from the centre of each quadrat. Measures assessed were soil pH (45 ml of water added to 15 g air-dried soil); soil inorganic and organic carbon (by acidification of a 15 mg sample of dried and ground material with phosphoric acid and by heating to 720 °C, respectively, measuring with a Non-Dispersive Infra-Red detector, Shimadzu TOC-VCSH, Milton Keynes, UK); and extractable Calcium (using acetic acid extraction followed by inductively coupled plasma optical emission spectroscopy (ICP-OES; Thomas, 1982)). Due to cost, only a proportion of soils were analysed. The choice was randomised and resulted in 1389 quadrats available for analysis.