



## Using compositional and functional indicators for biodiversity conservation monitoring of semi-natural grasslands in Scotland



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### ABSTRACT

Compositional indicators (i.e. indices that focus on the identity of species, genes or phylogeny) have been widely used to estimate and monitor biodiversity, however, their use in combination with species and/or community functional characteristics remains limited. Using large-scale, spatio-temporal data, we use both compositional and functional indices to investigate land-use change impacts on the vegetation of a semi-natural grassland ecosystem (Machair) for fourteen regions in Scotland, UK. Our study aimed to identify national- and regional-scale temporal vegetation patterns, and through use of simple compositional and functional indices (e.g. Competitor, Stress, Ruderal and Ellenberg scores) link observed changes to agricultural intensification and/or land-use abandonment. Using linear-mixed modelling and non-metric multi-dimensional scaling, we showed significant national and regional-scale changes in species composition over time. Increases in diversity, particularly gains in Machair grassland, identified several regions that may have benefited from past government incentivised schemes to protect the Machair, but which may also be suffering from an extinction lag. Shifts in plant functional signatures (CSR & Ellenberg values) identified varying degrees of internal (competition) and external (land-use) factors, highlighting several regions where biodiversity change could be linked to reduced disturbance (i.e. lower grazing intensity) or greater disturbance (i.e. land-use intensification). Our results demonstrate the utility of simple compositional and functional indices for monitoring biodiversity of semi-natural grasslands and identifying land-use drivers of change across different spatial scales.

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### 1. Introduction

Semi-natural grasslands (SNGs) constitute one of the most species-diverse communities in the World at small spatial-scales (Wilson et al., 2012). In Europe, the high level of biodiversity, characteristic of SNGs, is closely linked to centuries-long associations with agricultural land-use practices (Eriksson et al., 2002; Poschlod and WallisDeVries, 2002). However, for these tightly-coupled, social-ecological systems, cessation of traditional management (e.g. extensive grazing, hay making and rotational arable agriculture) in favour of land-use intensification (i.e. increased use of inorganic fertiliser and pesticides, heavier machinery and higher livestock densities) or complete cessation of any form of management has become a widespread occurrence. These land-use changes are considered the principal cause of habitat deterioration, fragmentation and biodiversity loss among European agricultural landscapes (Benton

et al., 2003; Lindborg et al., 2008) and are often linked to societal changes in response to economic factors. Therefore, given current and forecasted global biodiversity loss (Sala et al., 2000), appropriate design and implementation of conservation management to maintain and enhance these systems has global as well as local and regional importance.

However, there is a growing concern regarding the effectiveness of current monitoring frameworks for attempts to conserve and enhance biological diversity (Lindenmayer and Likens, 2010), particularly of semi-natural landscapes (de Bello et al., 2010). For SNGs, one part of the problem is likely to be the broad spatial grain (i.e. low sample intensity relative to distribution) common to many large-scale monitoring studies (e.g. The Countryside Survey of Great Britain; <http://www.countryside-survey.org.uk/>). European SNGs typically exhibit high spatial heterogeneity, usually as a result of spatially- and temporally-structured variation in agricultural management (Plieninger et al., 2006). Moreover, variation is intensified where land is apportioned to multiple landowners. Although such heterogeneity generally benefits the biodiversity of SNGs (Öster et al., 2007) and surrounding agricultural

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landscapes (Blitzer et al., 2012), patterns of biodiversity change as a result of changing land-use (i.e. intensification or abandonment) become extremely variable across relatively small-spatial scales. As a result, management and conservation policies designed and applied across large spatial extents may not always be appropriate (Fortin and Dale, 2005), often failing to be translated successfully across highly heterogeneous landscapes (Stenseke, 2006).

Another part of the problem is the complexity of biological diversity and the suite of indicators designed to measure its various facets. This issue has been addressed by de Bello et al. (2010) who derived a framework to help standardise and converge the use of biotic indicators (i.e. direct measures of components of biodiversity) and biotic drivers (i.e. factors that govern local biodiversity) for monitoring biodiversity change, specific to semi-natural habitats. Biotic indices (e.g. species richness, Shannon diversity, Simpson diversity, see Magurran, 2004) are commonly applied to measures of  $\alpha$ -diversity (*sensu* Whittaker, 1960, 1972) or a subset of  $\alpha$ -diversity, targeted at indicator species (i.e. suites of species that commonly occur in a habitat under different environmental conditions – e.g. grazed or un-grazed, favourable or non-favourable conservation condition etc.; see Rosenthal, 2003; Wittig et al., 2006). Such compositional measures are relatively easy to calculate and understand (Lamb et al., 2009) but they fail to provide information about potential abiotic or biotic drivers governing observed change. On the other hand, direct measures of biotic drivers (e.g. pH, organic matter and nitrogen content), for which links with biodiversity have been well researched (Hartley et al., 2003; Pärtel et al., 2004), are laborious and cost intensive.

To overcome these limitations, one approach is to use biotic indicators as surrogates for biotic drivers, particularly those that focus on the functional characteristics of species and ecological communities (i.e. functional indicators *sensu* de Bello et al., 2010). It is well understood that plant functional characteristics, pre-adapted to the local environment, provide distinct signatures to drivers of change that are not always easily measurable, and which often relate well to land-use processes (Pakeman, 2004; Quetier et al., 2007; Lewis et al., 2014). For example: the Competitor–Stress–Ruderal (CSR) classification of plant functional types (Grime, 1974) is frequently used to describe variation in vegetation through its functional composition (Cerabolini et al., 2010; Bornhofen et al., 2011; Kelemen et al., 2013). Among SNGs, shifts in CSR functional strategies can be observed. For example, land-use abandonment that leads to size asymmetric competition for light (Mason et al., 2011; Laliberté et al., 2013) will often result in competitive displacement of grazing-tolerant specialists and encroachment of competitors (Tilman and Lehman, 2001). Alternatively, declines in stress-tolerant plant and/or increase in ruderal species should be observed with soil nutrient enrichment resulting from land-use intensification (Critchley et al., 2003). Similarly, Ellenberg indicator values for plants (Ellenberg, 1988), which define species habitat preferences and distributions, also provide clear linkages between plant species occurrences and the environment, proving valuable for detecting changes in species compositions as a result of changes in land-use management. For example: Ellenberg N scores (measure of soil fertility) prove to be good correlates with soil mineralisable nitrogen (Rowe et al., 2011) and grassland productivity (Wagner et al., 2007).

Understanding the nature of vegetation change over time in this way can be paramount to the appropriate design of nature conservation practices, policies and land management across local, regional and national scales for all terrestrial habitats, not least SNGs. Although direct measures of biodiversity drivers are always preferable, inferences made from surrogate measures can be linked to inexpensive, non-laborious field measures, that in-turn act as proxies for land-use disturbances (e.g. vegetation height; presence/absence of grazing animals). Together, monitoring functional

classifications such as plant CSR strategies, and/or Ellenberg values, alongside compositional indices (including indicator species) and proxy measures of land-use should prove extremely useful for highlighting the effects of land-use change impacts across semi-natural landscapes.

In this study, we demonstrate the potential utility of simple functional indicators as proxies of biodiversity drivers in identifying temporal shifts in land use patterns among SNGs. Using archived biological records and revisitation data, valuable for quantifying long-term patterns of vegetation change (e.g. McCollin et al., 2000; Bennie et al., 2006; McGovern et al., 2011), we quantify national- and regional-scale temporal shifts in plant species diversity and composition for an internationally important SNG (Machair). Through the use of simple land-use measures and functional indicators, surrogates for abiotic/biotic drivers, we link changes in vegetation to changes in land-use, and identify regions where Machair SNGs are subject to land-use intensification and those that are subject to management declines over the last three decades. We highlight regions of potential conservation concern and discuss how simple environmental surrogates can be a useful, cost-effective measure for multi-scale environmental monitoring and conservation planning.

## 2. Methods

### 2.1. Study area

We used the grassland vegetation of the Scottish Machair as a case study (Fig. 1). Confined globally to north-western Europe, Scottish Machair contributes over two thirds of the World's total Machair extent (Dargie, 2000). Machair SNGs make an excellent study system as their high floristic diversity and conservation value is tightly coupled to centuries-long associations with human settlement (Gilbertson et al., 1996). Traditionally, land-use practices on the machair involved a form of small-scale rotational arable agriculture alongside livestock production (Supporting Information S1). Today, these habitats present a good example of how SNGs are threatened through agricultural and socio-economic change (Pakeman et al., 2012).

### 2.2. Floristic data

Species compositional data were collated from two sample periods. Baseline data was taken from the Scottish Coastal Survey (1975–1977; Shaw et al., 1983), compiled at the time by the Nature Conservancy Council (NCC) and Institute of Terrestrial Ecology (ITE). The temporal aspect was completed through a partial re-survey between 2009 and 2010 of regions known to include Machair communities (Fig. 1), i.e. all transient communities from seaward embryo dunes to inland sand affected peatlands, including Machair grasslands (Angus, 2006). National data analysis encompassed all temporal survey points, whereas regional-scale data analysis was restricted to each of the fourteen separate geographical regions (Fig. 1). Re-survey methodology followed closely that of the original survey (for details see Shaw et al., 1983), estimating cover of all higher plants and total cover of all bryophytes, including measuring environmental proxies for land-use management within 5 m × 5 m plots (Table S1). Relocation accuracy was estimated at ±10 m of the original geographical British National Grid co-ordinates derived from digitising original sample points marked on 1:10000 maps. This method of revisitation surveying for non-permanent vegetation sampling is proven to be effective in detecting temporal change with confidence (Ross et al., 2010).

Data specific to the focal community (i.e. Machair grassland) was abstracted from the temporal dataset. Only those plots with

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