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Can species-rich grasslands be established on former intensively managed arable soils?

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Abstract

Land use change from intensive arable production to extensively managed grasslands is encouraged through subsidy payments to farmers under the European Union's Common Agricultural Policy. Created grasslands are sown with a species-rich seed mix and receive limited or no fertiliser application with the aim of increasing the provision of non-production ecosystem services. In the UK these agri-environment schemes are funded for periods of 5, 7 or 10 years. This study compared the plant diversity and soil properties of paired intensively managed (IM) arable and recently created (3, 5, 8 and 9 years) extensively managed species-rich grasslands (SRG) at 4 sites in the Scottish Borders. Botanical surveys of the newly created grassland plots showed limited establishment of the species-rich seed mixes and the dominance of grasses that favour more nutrient-rich environments. Soil properties at 0-10 and 30-40 cm depths were measured over 2 consecutive years. Total and available soil nitrogen, phosphorus and soil organic carbon were not significantly different between paired plots. This study indicates that in order to create edaphic conditions for species-rich grassland communities to develop within a 10 year timespan on former intensively managed arable land, radical changes in soil properties are required, which current de-intensification managements are not achieving.

Keywords

Agri-environment; ecosystem services; land use; soil organic carbon; soil nutrients; biodiversity

1. Introduction

A growing awareness of the value of non-production ecosystem services (ES) provision to human health and wellbeing has encouraged the funding of agri-environment schemes in the UK, through which farmers receive funding to alter management practices to increase the provision of certain ES. In general, management to maximise production causes the decline of other ES (MA, 2005) including the regulation of water quality and nutrient cycling and maintenance of biodiversity, with mixed effects reported on climate regulation (Pilgrim et al. 2010).

In the European Union (EU) direct support and subsidies are provided to farmers through the Common Agricultural Policy (CAP). Funding for environmental initiatives is provided under the second pillar of the CAP through the European Agricultural Fund for Rural Development (EAFRD) and includes agri-environment schemes that aim to enhance the environmental value of land, such as the extensification of agricultural management through the creation of semi-natural grassland (EC, 2009). Under these schemes farmers are required to carry out an extensification of management practices by reducing or ceasing fertiliser application, grazing and cultivation, or removing the existing crop or sward and sowing a specified seed mix of desired grassland species. In England by the end of 2012 there were over 80,000 ha of created or restored grassland (Wilson et al., 2013), and £3 million was spent on the creation of species rich grassland and arable reversion to grassland in Scotland from 2008 to 2012 (Scottish Executive, 2012).

Across the UK SRG creation schemes are funded for periods of between 5 and 10 years. Thus within 10 years of adoption the benefits of agri-environment aimed at enhancing the provision of non-production ES should justify both the loss of production and the cost of the financial subsidy awarded to farmers (Horrocks et al., 2014). Despite the commitment of substantial sums of money and land to extensification schemes, there has been little research into (i) the extent to which they enhance provision of multiple ES and (ii) the potential for the legacy of intensive agriculture to continue to limit ES provision during the funding period of the agri-environment scheme. The creation of SRG in Scotland is listed as a land management option under the ‘biodiversity and landscape’ and ‘water quality’ regional priorities (Scottish Executive 2009), so the provision of increased biodiversity and improved water quality are key targets for SRG creation schemes.

The UK is a signatory of the Convention on Biodiversity (CBD) and is obliged to take targeted action to restore biodiversity where intensive agriculture has led to its loss (CBD, 2012). The maintenance of biodiversity enhances the provision of other ES, particularly those mediated by the soil, e.g. the storage, internal cycling and processing of nutrients (Haygarth and Ritz, 2009) and carbon (Goldstein et al., 2012). However, intensive agricultural practices, including the use of fertilisers, pesticides, tillage are incompatible with high biodiversity maintenance (Pilgrim et al., 2010). Changes to soil properties, which include decreased total soil nitrogen (N) increased N availability, decreased and soil organic carbon (SOC), and increased total and (Knops and Tilman, 2000), decreased soil organic carbon (McLauchlan, 2006) and increased total and available phosphorus (P) concentration (Gough and Marrs, 1990; McLauchlan, 2006) decrease botanical diversity.

The most diverse grasslands with plant species of the highest conservation value tend to occur on soils with low nutrient status, as large concentrations of nutrients favours dominance by a small number of species capable of rapid resource utilisation (Critchley et al., 2002; Janssens et al., 1998). Thus, substantial concentrations of legacy soil N and P can limit the biodiversity value of created or restored grasslands (Walker et al., 2004). Legacy soil N and P can also have significant implications for water quality since, increased concentrations in water bodies can result in eutrophication (Søndergaard and Jeppesen, 2007; Dungait et al., 2012). Nutrients leached in forms that are readily available for biotic uptake, such as NO_3^- (nitrate), may have a particularly large, immediate effect on the aquatic system.

Legacy effects of past management on soil properties can still be observed after many decades (Kopecký and Vojta, 2009) and in some cases thousands of years (Dupouey et al., 2002) following the cessation of intensive agriculture. Yet there are very few published reports of the co-dynamics of the major macronutrient (N and P) and C cycles in soils following the cessation of agricultural management (Table S1).

The aim of this study was to establish the extent of the legacy effect of former intensive arable management on ES provision including SOC and macronutrient cycling and biodiversity in recently created (<10 years) species-rich grasslands (SRG) on working farms. We focus in particular on direct measurement of botanical biodiversity provision, and soil chemistry, including N and P, which are key factors regulating both biodiversity and potential nutrient loss to water bodies, key targets of SRG creation.

We tested the hypotheses that:

1. Soil chemical properties (SOC, N, P and pH) will not change within the first 10 years following cessation of intensive management.

2. Legacy macro-nutrients will create soil conditions to which prescribed species-rich seed mixes are not well adapted.

2. Materials and Methods

2.1 Field Sites

Paired IM-SRG field plots (11 m x 11 m) were established in 2010 in fields on 4 farms in SE Scotland. SRG seed mixes (Table 1) had been sown 3, 5, 8 and 9 years previously in a portion of an IM field at each farm. Plot pairs were matched for soil type (silty loam, brown earth, Lauder series; Soil Survey of Scotland, 1981) using soil particle size analysis, slope and aspect. All of the IM plots continued to receive fertiliser throughout the study, in contrast to the SRG plots that had received no fertiliser or biocides since conversion (full details in Horrocks et al., 2015). Hereafter, each site is identified by the letter S followed by a number, which refers to the age in years since establishment of the SRG. Before conversion to SRG, sites S3, S8 and S9 had been under arable rotation for at least 20 years, and site S5 had been under intensive arable management until 2 years prior to establishment of the SRG.

2.2 Soil properties

2.2.1 Sampling and preparation

Sites S3, S8 and S9 were sampled in spring (late March) and summer (early July) in 2010 and 2011. The site at S5 was not sampled in 2011 having withdrawn from the agri-environment scheme at the end of 2010. Soil cores (5 cm diameter x 10 cm length, n=5) were sampled in a cross diagonal pattern from each plot (Been and Schomaker, 2013). Surface soil cores (0-10 cm depth) were taken in spring and summer. In spring, samples from 30-40 cm depth were also taken using a soil corer from a pit dug beneath the surface sample to 30 cm.

Fresh soil samples were sieved (2 mm) prior to analysis. The bulk density (BD) of the surface soil was determined for spring soil samples in 2010 using steel cores (56 mm diameter and 40 mm depth; Eijelkamp, Giesbeek, the Netherlands) according to Hopkins et al. (2009) and was used to calculate nutrient concentration per ha.

2.2.2 Total N and SOC

Total N and SOC (% mass) were determined for spring soil samples using elemental analysis (Carlo Erba NA 1500 analyser; CE Instruments Ltd, Wigan UK). Approximately 15 mg ground (pestle and mortar) oven dried soil was weighed into a foil capsule.

2.2.3 Total available N

Total available soil N and P concentrations were determined in both spring and summer soil samples to determine the level of intra-annual variation in more labile nutrient forms (Hatch et al., 2002 , Blake et al., 2003).

Total available soil N, defined as the sum of ammoniacal N (NH_4^+-N), nitrate N (NO_3^--N) and nitrite N (NO_2^--N) concentrations, was measured in 5 g of fresh soil extracted with 100 ml of 6% potassium chloride (KCl) on an orbital shaker for 1 hour at 150 revolutions min^{-1}. The suspension was allowed to settle for 10 mins before 20 ml was filtered through Whatman No. 42 filter paper (Whatman plc., Maidstone, UK) and analysed for NH_4^+ and NO_3^-/NO_2^- using a Bran & Luebbe Auto Analyser III (SPX Flow Technology, Brixworth, UK). Two blanks were prepared for each run and processed in an identical manner (Pansu and Gautheyrou, 2006). It was assumed that all oxidised N was present as NO_3^- since NO_2^- concentrations are usually negligible relative to NO_3^- (Shen et al., 2003).

2.2.4 Available P

Available soil P (defined as acetic acid extractable soil P) concentration was determined using the same method of extraction as for available N with 100 ml of 2.5% acetic acid in place of the KCl solution (Edwards and Hollis, 1982). The concentration of phosphate (PO_4^{3-}-P) in the extracts was measured using a Bran & Luebbe Auto Analyser III.

2.2.5 Total P

Total soil P concentrations were determined for spring soil samples using the Kjeldahl method (Taylor, 2000). Twenty ml of 95% sulphuric acid (H_2SO_4) and 6 Kjeldahl copper sulphate ($CuSO_4$) catalyst tablets (Fisher Scientific, Loughborough, UK) were added to 0.5 g of oven dried and ground soil and heated in a Buchi K-437 digestion system (Buchi UK Ltd., Oldham, UK) for 30 mins at 250°C, followed by 90 mins at 350°C. Once cool, digests were filtered through Whatman No. 42 filter paper, made up to 250 ml with deionised water, shaken by hand and then left for 10 hours to reach equilibrium. A 60 ml aliquot was analysed using a Bran & Luebbe Autoanalyser III using the same method as for available P.

2.2.6 Calculating soil nutrient concentrations

Gravimetric soil moisture content was determined for each homogenised batch of fresh soil prior to analysis for N and P, by drying a 20 g subsample at 105°C until constant weight was attained. The value was used to calculate soil N and P concentration per mass of dry soil (mg kg^{-1} dry soil) and converted to nutrient content ($kg\ ha^{-1}$) using bulk density values measured for each field plot.

2.3 Botanical survey

The percentage cover of plant species identified using Rose (2006) and Hubbard (1992) was recorded in July 2010 and 2011 using a 1 m x 1 m quadrat subdivided into 0.1 m x 0.1 m sections at 5 randomly located points within each SRG plot. The value for percentage cover was converted to a Domin score using the Joint Nature Conservation Committee Standard conversion table (Rodwell, 2006). Values for key traits, indicating their ecological niche, were collated from references for all species identified at the sites and included in the seed mixes. Traits used were i) Ellenberg indicator (EI) values (Ellenberg, 1979) for light and N (after Hill et al., 1999); ii) categorisation within the Competitive (C)-Stress tolerant (S)-Ruderal (R) system of plant functional types (Grime, 1974; Grime et al. 1996), with scores ranging from -2 to 2 on each axis (C,S and R) allocated according to Hodgson et al. (1999); and iii) canopy height taken as the maximum height according to the LEDA European plant trait database (Kleyer et al., 2008).

2.4 Data analysis

The Shannon diversity index (*H'*) was calculated for plant diversity in each plot (Equation 1), using the mean % cover to determine the abundance of the ith species as a proportion of total total cover (*Pi*) for each species:

$$H' = \sum_{i=1}^{s} -(pi \times \ln pi) \qquad (1)$$

Where, P*i* = abundance of the i^{th} species as a proportion of total cover

All soil analyses were conducted in duplicate and the mean of the replicate values was used for the data analysis using GENSTAT14. Where a normality plot indicated non-normal data distribution for a given variable, data were normalised by taking the natural logarithm (constant e). Data from paired plots at each site were compared for every sampling occasion using a two sample t-test, following a check for equality of variance the mean and standard deviation of the measurements and indication of significance of the t-test are reported for all plots and sampling occasions in tables. Subsequently a randomised block design ANOVA was applied to combined data from 2010 / 2011 to identify any significant consistent effects of management over the two year sampling period, management (IM / SRG) was modelled as a fixed effect, across the 4 sites, with the data blocked according to the site pair, considered as a random block (S3, S5, S8 and S9).Where both spring and summer analyses were performed (available soil N and P), separate models were written for the spring and summer data to enable the spring data to be analysed using a 2-way split plot design, with soil depth (0-10 cm / 30-40 cm) and management both taken as fixed effects $p<0.05$ was considered significant. The results of the ANOVAs are reported in the text.

3. Results

3.1 Soil organic carbon

The total SOC content did not vary significantly, as a function of management ($p=0.28$) or depth ($p=0.46$). The smallest SOC contents tended to occur at site S3, ranging from 11.1 (±2) to 24.2 (±9.3) t ha^{-1}, and the greatest at site S9, ranging from 24.1 (±11.7) to 38.7 (±3.5) t ha^{-1} in both sampling years. This pattern was observed in both the IM and SRG plots (Table 1).

3.2 Total and available nitrogen

The mean total soil N content did not vary significantly between IM and SRG plots, as a function of sample depth ($p=0.55$) or year ($p=0.11$). There was a trend for the smallest total soil N content to occur at site S3, ranging from 1.08 (±0.19) to 1.85 (±1.44) t ha^{-1}, and the greatest at site S9, ranging from 0.81 (±0.07) to 3.69 (±0.32) t ha^{-1} for both depths (Table 1). The greatest soil available N contents were at site S9, where there were peaks in total available soil N (>70 kg ha^{-1}) measured in both the IM and SRG plots in spring 2011 and in the IM plots in summer 2010 (Table 2); on both occasions the content in the IM plots were significantly greater. The total soil available N in S3, S5 and S8 tended to be less than those observed at site S9 and showed no consistent relationship with management. There were significant differences between paired plots for individual sampling occasions but these showed no consistent effect of management (ANOVA, spring $p=0.30$, summer $p=0.06$).

3.3 Total and available phosphorus

The mean total soil P (Table 1) did not vary significantly between IM and SRG plots, as a function of sample depth (p=0.33) or year (p=0.36). Likewise soil available P content (Table 2) did not vary significantly with management in spring (p=0.24) or summer (p=0.97). There was a trend for the smallest total soil P content to be recorded at site S3, ranging from 0.11 (±0.04) to 0.40 (±0.1) t ha^{-1} across both depth ranges, and the greatest at site S9, ranging from 0.43 (±0.09) to 0.81 (±0.07) t ha^{-1}. There were significant differences between paired plots for individual sampling occasions but these showed no consistent effect of management. The SRG plot at site S5 had a significantly higher total soil P content compared to the paired IM IM plot in the 0-10 cm depth. In Spring 2010 the IM plot at site S8 had significantly ($p<0.05$) greater mean soil total P content in the 30-40 cm depth range compared to the paired SRG plot.

3.4 Soil nutrient ratios

The soil C:N ratio was ~10 across all sites and did not vary significantly with management (p=0.12) or depth (p=34). The N:P ratios were much more variable (3.3-12.1; Figure 1), but as with the C:N ratio did not vary significantly with management (p=0.29) or depth (p=0.50).

3.5 Botanical survey

The SRG at plot S9 had the greatest diversity (as determined by the Shannon diversity index) and species richness in 2010 and 2011 (Table 4). In 2011 the species richness (total number of species recorded) in the S9 SRG plot was about double that for the S3 and S8 SRG plots. In 2010 the lowest diversity was recorded in the SRG plot at site S3, whilst in 2011 the lowest diversity occurred at site S8. The only plot at which an increase in diversity was observed between 2010 and 2011 was S3 SRG, where *H'* increased by 0.62. In the SRG plots at S8 and S9 *H'* decreased by 0.13 and 0.26 respectively between 2010 and 2011. In 2011 all three of the SRG plots sampled showed an increase in species richness from the previous year.

At all sites grass species provided over 50% of the total cover with forbs much less dominant (Figure 2). The most dominant grass species (mean % cover >10; Domin score ≥ 5) tended to be those not present in the seed mix, including *Agrostis stolonifera* and *Holcus lanatus* at site S3, *Phleum pratense* at site S5, *Arrenhatherum elatius, H. lanatus*, and *A. stolonifera* at site S8 and *A. stolonifera* at site S9 (Table 4). Grass species present in the seed mixes that achieved >10% cover included *Dactylis glomerata* at site S3, *Cynosurus cristatus* at site S5, and *Poa pratensis* at sites S8 and S9, whilst species present in the seed mix which failed to establish included *Festuca pratensis* at site S3, *Alopecurus pratensis, F. rubra, P. pratenis* and *Agrostis capillaris* at site S5, *A. capillaris, C. cristatus* and *F. ovina* at site S8 and *A. capillaris* and *F. ovina* at site S9. The only forb species not present in the seed mix that provided a mean cover of >10% (Domin ≥5) was *Trifolium repens* at sites S3, S5 and S9. Other forb species that established despite not being present in the seed mix included *Rumex obtusifolius* at site S5, *Cirsium vulgare, Ranunculus bulbosus, T. repens* and *R. obtusifolius* at site S8 and *Bellis perennis, Cerastium fontanum, C. vulgare, Plantago lancelota, R. bulbosus, Silene alba,* and *Taraxacum spp.* at site S9. Forb species present in the seed mix and providing >10% cover (Domin ≥5) included *Rhinanthus minor* at site S5 and *Lotus corniculatus* at sites S8 and S9. At site S8, 5 out of 8 sown forb species were not recorded in any quadrat in either year, whilst from the same seed mix sown at site S9, only 1 of the 8 species failed to establish. The percentage cover from legumes at the four sites ranged from 10.2% at site S8 in 2011 and 23.2% at site S9 in 2010.

3.6 Plant traits

The most dominant grass species (Domin value ≥ 5) had either a generalised strategy according to C-S-R theory (Grime, 1974), scoring 0 across the three axes according to Hodgson et al. (1999) or a more competitive / disturbance tolerance strategy, scoring higher on the C and R axes compared to the stress tolerance (S) axis (Table 5). The EI-N scores of the most dominant grasses (range 5-7; mode 6) were indicative of species found in soils of intermediate to high fertility, with the exception of *C. cristatus* at site S5, which had an EI-N of 4. The modal EI-light value of the dominant grasses was 7 with all species being typical of well-lit environments (Hill et al., 1999). The requirement for high light environments was also a characteristic of the forb species which established, as well as of those which failed to establish from the seed mixes. The established forb species typically have a generalist or ruderal / competitive strategy according to the CSR theory, with the exception of *Centaurea nigra*, a stress tolerator not present in the seed mix, which established at site S9 (Domin value = 4) and *Lotus corniculatus* var. *sativus* a cultivated variety of a stress tolerator present in the seed mix at sites S8 and S9. Typically the forb species identified and present in the seed mix had a lower EI-N compared to the grass species (range 2-9; mode 4). The non-sown species that had the greatest dominance included *T. repens*, *R. obtusifolius* and *Cersatium fontanum* which have EI-N values of 6, 9 and 4, respectively.

4. Discussion

The effectiveness of agri-environment schemes has been a subject of recent debate. The schemes have been criticised for providing limited benefit and can also have unforeseen costs, for example, by increasing production pressure and environmental damage elsewhere to compensate for production losses in agri-environment schemes (Ekroos et al., 2014). The current study provides valuable insight into the value of extensive grassland creation schemes. Whilst the findings are primarily applicable to the specific soil type studied (brown earths), the results highlight the potential for legacy effects of intensive management on soil chemical properties to limit the value of agri-environment schemes for enhancing ecosystem service (ES) provision.

4.1 Enduring effects of intensive management on soil nutrients

The cycling and changes in C, N and P content in soils are regulated by physical, chemical and biological processes. In intensively managed systems N, P and C cycles become decoupled as plants can obtain their required nutrients directly from the soil solution following fertiliser application (Dungait et al., 2012; Soussana and Lemaire, 2014). A transition towards more 'natural' soil processes would tend to reduce total P in soils and increase SOC and total N (as components of organic matter), thus altering the stoichiometry of the soil nutrients. In this study we focus in particular on direct measurement of botanical biodiversity provision, and soil chemistry, including N and P, which are key factors regulating both biodiversity and potential nutrient loss to water bodies. We hypothesised, however, that in the newly created SRG sites (<10 years) in this study, legacy effects of former intensive management would limit succession towards a more 'natural' system with soil macronutrient content showing no detectable change compared to the IM sites, thus limiting improvements in key ES provision. The data from four working farms in Scotland largely support our hypothesis. The percentage total N in our study plots (0.1-0.3%) was closer to those measured by other authors in IM soils, as opposed to semi-natural grassland habitats. For example a study of UK grasslands reported a mean soil total N content of 0.5% at long established semi-natural grasslands, compared to a mean value of 0.3% at adjacent intensive agricultural sites (Gough and Marrs, 1990). Another study of permanent, species-rich grassland in Western Europe found soil total N ranging from 0.3 to 0.9% (Janssens et al., 1998). These comparisons with other IM sites and established SRGs highlight the extent of the legacy effect of former intensive management on the soils in this study, as there is no significant increase in total soil N, which would be expected when comparing IM sites with long established SRG.

Highly managed systems can become 'leaky' and maintain relatively high concentrations of available soil N (Wardle et al., 2004). In more 'natural' systems rates of N release from organic matter mineralisation may be regulated through plant-soil feedbacks hence these systems tend to be characterised by improved N use efficiency and retention (Chapman et al., 2005). The IM and SRG plots in this study maintained similar, high contents of available soil N, with no significant management effect on total available N content, supporting the theory that mineralisation rates were rapid.

The availability of soil P, which remained high in the SRG sites in this study may also encourage N mineralisation, by supporting elevated rates of microbial activity and encouraging plant growth and the production of high quality, readily mineralised plant matter (Janssens et al., 1998; section 4.2). Rates of P cycling are an order of magnitude less than N (Dungait et al., 2012), thus, fertiliser applications during intensive management tend to lead to soil P accumulation, which may take many decades to decline following cessation of fertiliser application (Dodd et al., 2012; Falkengren-Grerup et al., 2006). Desorption or dissolution of the total P pool can maintain soil available P (Koopmans et al., 2004; Vu et al., 2010). The persistence of accumulated soil P following cessation of intensive management was observed in this study, as was the maintenance of a consistent pool of soil available P; neither total nor available soil P content differed significantly between the IM and SRG sites.

In agro-ecosystems at steady state, net loss or gain of SOC is not observed, i.e. the amount of C lost through decomposition processes and harvesting is the same as the net ecosystem production (Jones and Donnelly, 2004; Smith et al., 2010). There are well-recognised benefits associated with increasing SOC in agricultural soils, i.e. to mitigate climate change and improve soil quality. Management changes, including conversion from arable cropping to permanent grassland have been found to increase SOC (Conant et al., 2001; Guo and Gifford, 2002), however there was no measureable difference in SOC between the paired SRG and IM plots in this study. We assume that high rates of organic matter mineralisation at our sites balanced SOC and N inputs from the SRG plants, thus preventing the hypothesised increases in SOC and total N.

4.2 Legacy soil nutrients limit biodiversity provision

The relatively abundant soil available N and total P contents recorded at the SRG sites in this study are likely to impact on the nature of the plant community established, favouring dominance by a few plant species typical of more nutrient rich environments and thus limiting the biodiversity, species richness and conservation value of the created SRG. The seed mixes sown in the SRG plots in this study met the requirements of the Scotland Rural Development programme for low productivity mixes and contained plants typical of species rich grasslands that develop in relatively nutrient poor soils (Scottish Executive, 2011).

The dominance of non-sown species, particularly grasses and the limited establishment of sown species, demonstrated that success in establishing the desired sward at the SRG sites was limited. Analysis of the traits of the most dominant grass species found them to be characteristic of generalist species able to compete effectively in environments with low nutrient stress (scoring lower on the S axes relative to C and R and high EI-N) or species able to take advantage of disturbance due to high fecundity and rapid growth (scoring relatively high on the R axes relative to C and S score). Other authors have reported similar observations, and found that high soil P content in particular can limit biodiversity and prevent establishment of species typical of low nutrient environments (Pywell et al., 2003). The conservation of biodiversity is a central goal of agri-environment schemes. Maintaining biodiversity has been shown to support the provision of other ecosystem services, such as efficient nutrient cycling and to increase ecosystem stability through functional diversity (Cardinale et al., 2012), thus biodiversity is a key measure of the ability of a landscape to provide multiple ES. The plant communities in the newly created SRGs in this study were less diverse and differed substantially from those found in well-established species rich hay meadows, which are a threatened European habitat (Garcia, 1992). Traditional hay meadows in Sweden have been found to have *H'*s of 2.56-3.71 and mean EI-Ns ranging from 2.3 to 4.5 (Linusson et al., 1998). Similarly Shannon diversity indexes ranging from 0.5 to 5 were measured in old, permanent grasslands, with the diversity at the majority of low fertility sites being greater than 2.5 (Janssens et al., 1998). Most of the plots in this study fail to achieve such high diversity.

Grassland diversity has been shown to negatively correlate with the grass:forb ratio (Willems and Nieuwstadt 2009), indicating that the dominance by a few grass species is driving the relatively low diversity in the SRG plots in this study. Many of the forbs which did establish generally had low abundance (Domin value of 1). All the forb species had a requirement for high light environments (EI-light 7 or 8), hence reduction in light availability in the sward caused by dominant tall growing grasses is likely to have been a significant factor in limiting forb establishment and overall biodiversity at the sites (Hautier et al., 2009). Amongst the most dominant forb species were the legumes *T. repens* and *L. corniculatus*. The former is similar to other dominant species at the sites as it has low stress tolerance (low S score) and typically grows in relatively nutrient rich soils (EI-N =6), however the latter is stress tolerant and typically grows in relatively infertile soils (EI-N =2).

The large soil available P content in the SRG soils could explain the relative dominance by legumes as P availability has been found to correlate positively with legume abundance (Bobbink, 1991). Rates of N fixation from legumes in UK grasslands have been estimated to be between 74-280 kg N ha^{-1} yr^{-1} (Cowling, 1982). The abundance of legumes in the SRG plots should have a positive feedback on soil fertility through N fixation, providing a supply of easily decomposable (low C:N ratio) litter, which is readily mineralisable. Another potential source of N input to the SRG plots is atmospheric N deposition which is estimated at 15.12 kg N ha^{-1} yr^{-1} in the area of the field sites (APIS; CEH 2014). The combination of N fixation and deposition could explain the relatively high available and low total soil N observed in the SRG plots. The soil in the SRG plots showed no significant difference in N content to the IM soils, which received fertiliser applications in line with recommendations in the RB209 fertiliser manual for wheat and winter Barley (DEFRA, 2010), consisting of an initial fertiliser application of approximately 40 kg N ha^{-1} in February each year followed by additional applications in May / April of up to 150 kg N ha^{-1}. These fertiliser applications were similar to the potential N fixation by legumes in the SRG sites.

Fertiliser applications to IM plots were made to coincide with crop establishment and the period of maximum stem extension. During this time, N uptake rates by cereal crops are likely to have been greater than those of the grassland species in the SRG, which further explains the similarities in measured soil N between IM and SRG plots, despite cessation of fertiliser application to the SRG (Horrocks et al., 2014).

The observation that some stress tolerant species typical of more nutrient poor soils, such as *L. corniculatus* and *C. fontanum,* did establish at the sites could be due to the spatial variability in soil nutrient availability observed at the sites, allowing species with lower competitive ability and greater stress tolerance to establish in patches of lower nutrient availability. In the case of *L. corniculatus* var. *sativus* it is possible that the particular non wild type cultivar identified at the site had a superior competitive advantage, which could explain why it was able to develop such dominance at sites S8 and S9 (Schröder and Rudiger, 2012).

As well as taking advantage of spatial variability in soil nutrient availability, some species will be able to benefit from temporal environmental changes. For example, at site S3 in 2011, the increased cover from forbs and increased diversity could have been in response to the 6% cover from bare ground at the site in 2010, providing niches for light-loving disturbance tolerant forb species such as *Taraxacum* agg. and *T. repens* that would otherwise have been shaded out by dominant grass species. Such shifts could be short lived as more competitive species dominate again in future years. Another factor that can affect establishment of sown species is the size, composition and longevity of the weed seedbank present at a site. There are not data for the weed seedbank at the study sites, but assessment of the effect of the weed seedbank could be a valuable addition to future studies.

The botanical survey results support the hypothesis that high legacy soil nutrient content, in particular soil P, limits biodiversity provision at the recently created SRG sites by allowing the dominance of a limited number of low conservation value grasses. The success of disturbance tolerant species (high R score) could also be expected as the SRG sites were all ploughed prior to sowing so species able to rapidly colonise disturbed soils would have been advantaged (Pywell et al., 2003).

The dominance of non-sown species and relatively poor performance of forbs suggest the composition of seed mixtures selected for the sites were not appropriate to the soil conditions as a limited number of competitive grass species were able to dominate. The results highlight the need for management actions that decrease soil fertility prior to attempting to establish species-rich semi natural swards (Pywell et al., 2003; Smith et al., 2003). Whilst these have been recommended previously in the literature it is apparent from this study that wider implementation is required in the field. Soil testing to identify sites suitable for SRG establishment should be encouraged (Hautier et al., 2009).

5. Conclusions

Through comparisons of repeated measurements of multiple soil properties at paired IM and SRG sites this study has provided a much greater insight into soil properties before and after entry into agri-environment schemes. The data provide strong evidence for a substantial legacy effect on soil properties which could limit the benefit of newly created SRGs in supporting enhanced ES provision, including plant biodiversity provision. Despite successful establishment of some target seed mix species in the newly created grassland sward, overall the diversity, richness and composition of the plant communities were low when compared to long established species-rich grasslands, managed extensively for many decades. Overall the study draws into question the value of funding agri-environment schemes that encourage the short term creation of ‘semi-natural’ grasslands as the benefits they provide in terms of ES provision are limited. Instead resources (money and land) may be better prioritised to maintaining existing and long established semi-natural grasslands, or sowing moderately diverse mixtures containing more competitive forbs (Woodcock et al., 2014), which have been demonstrated to provide significant ES benefits.

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Table 1. Mean (n=5) total soil organic carbon (SOC), nitrogen (N) and phosphorus (P) content in soil samples collected from paired intensively managed (IM) and 'species-rich' grassland (SRG) plots at 4 sites in the Scottish Borders. Values in brackets show 1 standard deviation, where a t-test indicated that values were significantly greater ($p<0.05$) than at the paired plot that figure is in bold.

Site	Season	Depth	Total SOC (tonne ha^{-1})				Total N (tonne ha^{-1})				Total P (tonne ha^{-1})			
		(cm)	2010		2011		2010		2011		2010		2011	
			IM	SRG	IM	SRG	IM	SRG	IM	SRG	IM	SRG	IM	SRG
S3	Spring	0-10	14.4 (2.0)	16.2 (0.9)	20.1 (14.9)	**24.2 (9.3)**	1.40 (0.20)	1.63 (0.07)	1.85 (1.44)	1.70 (1.20)	0.12 (0.11)	0.19 (0.12)	0.29 (0.05)	0.40 (0.10)
	Spring	30-40	11.7 (1.7)	14.0 (1.1)	17.4 (12.3)	11.1 (2.0)	1.21 (0.19)	1.38 (0.14)	1.26 (1.27)	1.08 (0.19)	0.11 (0.04)	0.12 (0.04)	0.25 (0.02)	0.31 (0.04)
S5	Spring	0-10	24.8 (4.3)	28.3 (4.0)	-	-	2.46 (0.47)	2.58 (1.24)	-	-	0.29 (0.05)	**0.46 (0.14)**	-	-
	Spring	30-40	24.0 (6.1)	24.2 (7.5)	-	-	2.38 (0.56)	2.32 (0.63)	-	-	0.24 (0.06)	0.43 (0.19)	-	-
S8	Spring	0-10	28.2 (5.7)	23.5 (1.9	24.5 (5.1)	19.7 (4.1)	2.67 (0.49)	2.22 (0.20)	2.31 (0.51)	1.22 (1.02)	0.46 (0.09)	0.35 (0.05)	0.48 (0.12)	0.32 (0.03)
	Spring	30-40	22.4 (2.2)	21.7 (2.4)	33.1 (12.0)	32.1 (11.6)	2.20 (0.18)	2.13 (0.24)	3.06 (1.04)	2.58 (0.65)	**0.40 (0.07)**	0.31 (0.04)	0.34 (0.13)	0.41 (0.06)
S9	Spring	0-10	**38.7 (3.5)**	31.2 (1.1)	29.6 (12.4)	24.1 (11.7)	3.69 (0.32)	2.92 (0.16)	2.72 (1.33)	1.73 (1.28)	0.81 (0.07)	0.60 (0.03)	0.64 (0.16)	0.52 (0.03)
	Spring	30-40	37.9 (2.7)	28.1 (1.6)	25.5 (16.7)	25.2 (9.3)	3.64 (0.26)	2.67 (0.22)	2.33 (1.49)	2.28 (0.83)	0.77 (0.04)	0.54 (0.05)	0.70 (0.05)	0.43 (0.09)

Table 2. Mean (n=5) available nitrogen (N) and phosphorus (P) content in soil samples collected from paired intensively managed (IM) and 'species-rich' grassland (SRG) plots at 4 sites in the Scottish Borders. Values in brackets show 1 standard deviation, where a t-test indicated that values were significantly greater ($p<0.05$) than at the paired plot that figure is in bold.

Site	Season	Depth (cm)	Available N (kg ha $^{-1}$)				Available P (kg ha $^{-1}$)			
			2010		2011		2010		2011	
			IM	SRG	IM	SRG	IM	SRG	IM	SRG
S3	Spring	0-10	8.14 (2.9)	8.3 (4.7)	18.8 (0.27)	21.6 (16.4)	34.5 (19.9)	32.4 (7.5)	31.6 (5.8)	30.4 (11.4)
	Spring	30-40	**8.6 (3.1)**	4.3 (1.3)	12.8 (3.6)	12.8 (2.8)	33.5 (21.1)	33.0 (8.8)	**32.9 (8.5)**	16.4 (11.5)
	Summer	0-10	**15.7 (0.6)**	7.7 (0.9)	17.1 (13.6)	7.9 (4.2)	2.5 (0.24)	**17.1 (3.4)**	44.5 (6.1)	42.4 (5.4)
S5	Spring	0-10	4.1 (0.8)	**7.2 (2.2)**	NA	NA	11.0 (10.4)	36.5 (53.5)	NA	NA
	Spring	30-40	6.2 (2.9)	6.8 (3.2)	NA	NA	14.0 (13.9)	42.2 (61.9)	NA	NA
	Summer	0-10	8.9 (1.4)	**14.4 (2.8)**	NA	NA	**5.6 (0.3)**	3.3 (0.1)	NA	NA
S8	Spring	0-10	31.2 (4.1)	25.7 (3.7)	**48.8 (8.1)**	31.0 (9.9)	23.3 (11.9)	14.0 (6.5)	37.8 (3.8)	27.0 (7.5)
	Spring	30-40	12.8 (2.5)	13.3 (2.9)	**40.0 (8.2)**	15.1 (1.3)	14.6 (3.0)	11.9 (3.2)	35.5 (7.1)	27.2 (7.7)
	Summer	0-10	10.4 (0.1)	11.7 (3.2)	10.6 (5.7)	8.4 (2.4)	20.9 (1.1)	7.3 (2.5)	29.7 (5.2)	**34.0 (5.2)**
S9	Spring	0-10	13.5 (5.4)	**29.5 (3.8)**	**112 (20.2)**	70.7 (21.9)	22.7 (11.6)	13.1 (6.1)	36.8 (3.7)	25.3 (7.0)
	Spring	30-40	8.4 (2.6)	**13.8 (2.1)**	**72.1 (9.4)**	10.7 (2.0)	38.0 (5.4)	21.3 (11.4)	**65.6 (4.2)**	21.6 (4.3)
	Summer	0-10	**66.8 (35.1)**	10.0 (1.6)	14.4 (7.0)	14.2 (3.2)	**20.3 (1.1)**	6.9 (2.3)	28.9 (5.1)	31.9 (6.5)

Table 3. Summarising the diversity, percentage of sown species established and species richness in each of 4 species rich grassland (SRG) plots in July 2010 and 2011.

	Shannon diversity index		% of seed mix species which have established		Total species richness	
Plot	2010	2011	2010	2011	2010	2011
S3 SRG	1.33	1.95	50.0	66.7	8	11
S5 SRG	1.66	NA	38.9	NA	11	NA
S8 SRG	1.82	1.69	23.1	30.8	9	11
S9 SRG	2.33	2.07	61.5	61.5	15	21

Table 4. Plant species identified at each of 4 species rich grassland plots (S3, S5 S8 and S9). Domin scores allocated according to the Joint Nature Conservation Committee Standard (Rodwell, 2006) based on mean (n=5) percentage recorded in 1 m x 1 m quadrats in July of 2010 and 2011. Ellenberg indicator values for light and nitrogen (N) were obtained from Hill et al. (1999), the C-S-R category to which each species is assigned was obtained from Grime et al. (1996) and scores from -2 to 2 for each axis allocated according to Hodgson et al. (1999). Canopy height is the maximum canopy height (m) taken from the LEDA plant trait database (Kleyer et al., 2008). Domin values are entered for all species present in the seed mix for each site; values in bold and underlined indicate species established at sites that were not present in the seed mix.

		Domin score				C-S-R scores			Ellenberg indicators		
Species	Type	S3	S5	S8	S9	C	S	R	Light	N	Canopy height
Alopecurus pratensis	grass		0	**1**		1	-1	-1	7	7	0.7
Anthoxanthum odoratum	grass				**3**	-1	0	0	7	3	0.25
Festuca pratensis	grass	0			**3**	0	0	0	7	6	0.8
Dactylis glomerata	grass	5			**4**	1	-1	-1	7	6	1.1
Arrenhatherum elatius	grass			**6**		1	-1	-1	7	7	1.8
Holcus lanatus	grass	**5**		**5**	**4**	0	0	0	7	5	0.5
Phleum pratense	grass	4	**8**		**3**	0	0	0	8	6	0.9
Agrostis castellana	grass	4			**3**	0	0	0	6	4	0.2
Festuca rubra	grass	4	0		**4**	0	0	0	8	5	0.9
Agrostis stolonifera	grass	**5**		**7**	**5**	0	-2	0	7	6	1.3
Agrostis capillaris	grass		0	0	0	0	0	0	6	4	0.4
Cynosurus cristatus	grass		7	0	1	0	0	0	7	4	0.75
Poa pratensis	grass	3	0	6	5	0	0	0	7	5	0.5
Festuca ovina	grass			0	0	-2	2	-2	7	2	0.35
Lolium perenne	grass		**4**			0	-1	0	8	6	0.2
Achillea millefolium	forb		0	1	1	0	-1	0	7	4	0.8
Bellis perennis	forb				**1**	-1	-1	1	8	4	1
Centaurea nigra	forb		0	0	4	-1	1	-1	7	6	0.65
Cerastium fontanum	forb	**3**			**3**	-1	-1	1	7	4	0.25
Cirsium vulgare	forb		0	**1**	**1**	0	-2	0	7	6	1.2
Galium verum	forb		0			0	0	-1	7	2	1
Hypochaeris radicata	forb		1			0	0	0	8	3	0.1
Papaver rhoeas	forb					-2	-2	2	7	6	0.9
Plantago lancelota	forb		3		**1**	0	0	0	7	4	0.4
Prunella vulgaris	forb		1	0	1	0	0	0	7	4	0.3
Ranunculus acris	forb		3	0	1	0	0	0	7	4	0.1
Ranunculus bulbosus	forb			**1**	**1**	-1	0	0	7	4	0.25
Rhinanthus minor	forb		7	0	0	-2	-1	1	7	4	0.6
Rumex acetosa	forb		0	3	1	0	0	0	7	4	1
Rumex obtusifolius	forb		**3**	**3**		0	-2	0	7	9	1.2
Silene alba	forb				**1**	-1	-2	1	7	6	1
Taraxacum agg.	forb	**1**			**1**	-1	-1	1	7	6	0
Trifolium repens	forb	**5**	**7**	**1**	**5**	0	-1	0	7	6	0.5
Vicia cracca	forb		0			1	-1	-1	7	5	2
Lotus corniculatus var sativus	forb			5	5	-1	1	-1	7	2	1.7
Lathyrus pratensis	forb		0			0	0	0	7	5	1.2
Leontondon autumnalis	forb		0			-1	-1	1	8	4	0.15
Leucanthemum vulgare	forb		1	0	1	0	-1	0	8	4	0.6

Figure 1. Mean (n=5) soil organic carbon (SOC): total nitrogen (N) plotted against total N:total phosphorus (P) ratio at 4 sets of paired intensively managed (IM) and species rich grassland (SRG) field plots, measured in July of 2010 and 2011.

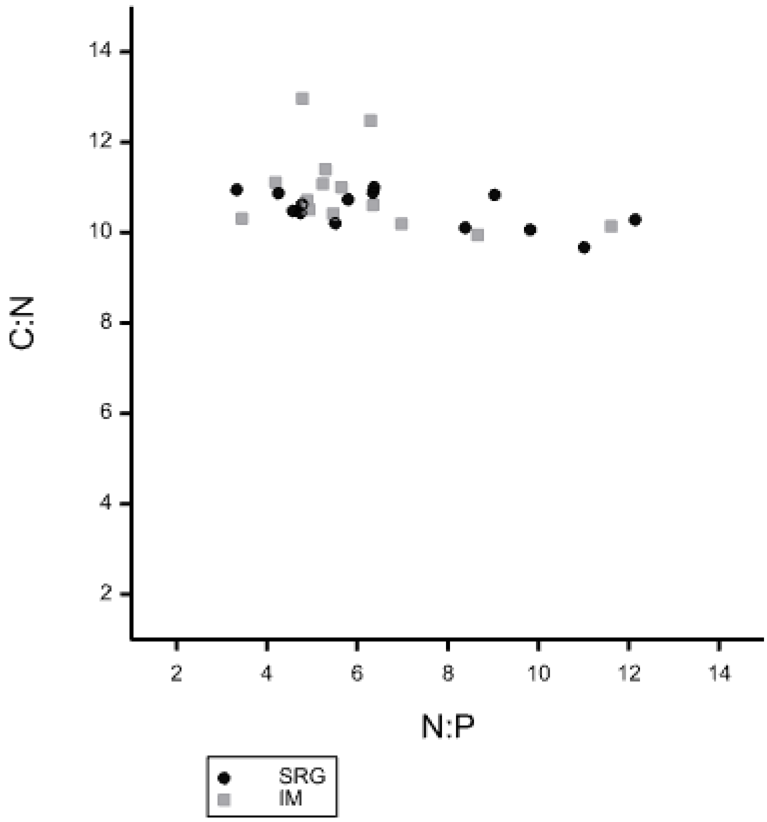

Figure 2. Mean (n=5) % cover within a 1 m x 1 m quadrat provided by grasses that were either present in the seed mix (sown) or not (non-sown) and leguminous (L) and non-leguminous (NL) forbs that were either sown or non-sown at 4 species rich grassland plots (S3, S5, S8 and S9) in July of a) 2010 and b) 2011. S5 was not surveyed in 2011 as the field had been withdrawn from the scheme 2011.