Journal for Nature Conservation 20 (2012) 162–169

Contents lists available at SciVerse ScienceDirect

Journal for Nature Conservation

journal homepage: www.elsevier.de/jnc

Changes in the vegetation composition of hay meadows between 1993 and 2009 in the Picos de Europa and implications for nature conservation

Heather E. Prince [a,*], Robert G.H. Bunce [b,c], Rob H.G. Jongman [b]

[a] *University of Cumbria, Rydal Road, Ambleside, Cumbria LA22 9BB UK*
[b] *Alterra, PO Box 47, 6700 AA, Waganingen, The Netherlands*
[c] *Estonian University of Life Sciences, Kreutzwaldi 64, 51014 Tartu, Estonia*

ARTICLE INFO

Article history:
Received 5 July 2011
Received in revised form
20 December 2011
Accepted 10 January 2012

Keywords:
Biodiversity loss
Herb-rich grassland
Indicator species
Northern Spain
Resource loss
Stratified random sample

ABSTRACT

The Picos de Europa are a range of predominantly Carboniferous Limestone and Sandstone mountains mainly in the Cantabrian region of northern Spain. The highest peaks are precipitous and reach 2600 m. There are complex gradients between Lusitanian, Alpine and Mediterranean environmental zones, as well as variable soil types. In combination with the long history of traditional agricultural management, a wide range of diverse habitats and species is present. The herb-rich hay meadows have long been recognised as having a high nature conservation value but, as elsewhere in European mountains, such grasslands are threatened by changing agricultural practices. Accordingly, in 1993, 92 quadrats were recorded using a restricted list of indicator species from stratified random samples. The authors repeated the sample in 2009. Changed land use had only occurred in approximately 3% of meadows, however, farmyard manure was no longer used, probably because of shortage of labour. Statistical analysis of the vegetation data showed a range of significant changes consistent with the increased use of slurry, as well as re-seeding of some fields. The grass swards had not only become denser, with fewer species present, but there was also a loss of sensitive indicators especially of calcareous conditions and open vegetation. By contrast, competitors had increased and the vegetation had become simpler, with the balance of vegetation types shifting to more nutrient rich conditions. These changes have mainly occurred in the more fertile meadows used for silage. The core of about 35% of herb-rich meadows, mainly cut for hay, has remained relatively stable but the results show that they are at risk if the current trend continues. If management practices that form the core of traditional agriculture are not maintained, one of the most important resources of herb-rich meadows in Europe will be lost.

Introduction

A wide range of diverse habitats and species is present in the Picos de Europa mountains of northern Spain. The hay meadows have long been recognised as having particular botanical significance and a high nature conservation value (Farino 1987, 1988; Goldsmith & García 1983; Rivas-Martinez et al. 1984). The exceptional plant diversity results from the combination of pedogenic and climate factors interacting with centuries of management by man. Traditionally, the practice of altitudinal transhumance enabled the species-rich meadow areas to produce hay in the spring and summer, which provided fodder for overwintering livestock (Bunce et al. 1998; Ruiz & Ruiz 1986). The mountain hay meadows of the Picos de Europa are still mainly managed using traditional agricultural practices, although these are changing because of the introduction of silage, particularly on less steep fields. In Spain, there is no available literature on the changes over time in the species structure of mountain grasslands, hence the present paper. This study assesses changes in the vegetation composition of hay meadows and interprets these in the light of changing agricultural practices. The conservation importance of the species-rich mountain hay meadows in this region is also emphasised.

Over the last 50 years, the status of many Spanish cultural landscapes has changed because of differences in management systems and social factors. The pace of these changes has accelerated since Spain joined the European Union (EU) in 1986, but especially in the last ten years. Spanish agriculture had already moved away from self-sufficiency before the accession to the EU, but the Common Agricultural Policy (CAP) with its subsidy incentives has accelerated this process (Hindmarch & Pienkowski 2000). Agricultural development in the Picos de Europa has been slower than in other parts of Europe. However, some noticeable changes have occurred. There has been a decline in the local equivalent of pollarding ash trees (*Fraxinus excelsior* L.), which produces the distinctive "candelabra"

* Corresponding author.
E-mail address: Heather.Prince@Cumbria.ac.uk (H.E. Prince).

1617-1381/$ – see front matter
doi:10.1016/j.jnc.2012.01.002

shape indicative of the traditional agriculture. There has also been a shift from the traditional cattle breed (*Tudanca*) to high output Charolais and Limousin for beef, and Friesian and Swiss Grey for milk. These foreign breeds are ill suited to the swards on steep mountain pastures and therefore they are rarely taken to higher altitudes in the valley (Waterhouse 1996). In combination with a decline in the farming population and reduced land management, less cutting or burning has led to scrub encroachment on the steeper slopes. The need for a higher productivity in meadows and pastures has been met through silage production and an increasing use of slurry. The former also allows earlier and more frequent cutting, which prevents plants from setting seed (Farino 2009; Frame 2000).

The reduced area of hay meadows raises concerns over the loss of floristic biodiversity and its associated fauna (Baura et al. 2006; Hopkins et al. 2000). The remaining meadows on steeper slopes tend not to be fertilised and have lower land management pressure due to a variety of social factors, especially the aging population.

Cultural landscapes, such as those in the Picos de Europa, in which component ecosystems have developed over many centuries in response to management practices, are now recognised as being of major conservation importance (Bergland 1991; Pedroli et al. 2007; Selman, 1994). Mountain hay meadows in Europe are under threat mainly because of changes in traditional management systems (Bunce et al. 2004). Species richness is higher in less fertilised meadows and there is evidence that there are also more species in meadows that are cut less frequently in the Italian Alps (Marini et al. 2008) and Norway (Losvik 2001). Their conservation importance is recognised by their presence in the list of Annex 1 habitats in the EU Habitats Directive. Both lowland and upland hay meadows are listed but the Picos de Europa meadows fit better into the latter category. Hay meadows are rare habitats in Britain and a priority habitat in the UK Biodiversity Action Plan (Jefferson 2005; O'Reilly 2010). Agricultural intensification, involving ploughing and re-seeding and a shift from hay-making to silage production, has taken place over the last 50 years and has been linked to losses in species-rich upland hay meadows. Many meadows continue to exhibit a decline in floristic richness, despite recent conservation initiatives, including voluntary agri-environment schemes (Jefferson 2005). A more recent initiative is seeking to restore and enhance hay meadows in the North Pennines Area of Outstanding Natural Beauty (AONB), U.K., through 'Hay Time' (North Pennines AONB Partnership, 2011) by harvesting seed from species-rich meadows and spreading it on sites that have lost species and by providing detailed habitat management advice. Bunce et al. (1998) suggested that an option for nature conservation in the Picos de Europa is to introduce agri-environmental measures similar to those in other parts of Europe (cf. 'Ecological Compensation Areas' in Switzerland (Hofer et al. 2011)).

Changes in the floristic diversity and composition of hay meadows are caused by several management factors. The present study reports changes in mountain meadows and pastures over almost 20 years and interprets these in terms of known drivers.

Location

The Picos de Europa mountains in northern Spain, are a range covering approximately 600 $km^{2.}$ The highest peaks reach 2600 m and consist of precipitous Carboniferous Limestone cliffs, screes and grasslands. At lower altitudes the mountains and valleys also include Carboniferous Sandstones and Shales which have gentler slopes, though some cliffs are also present. There are also some smaller areas of conglomerates giving local variations in soil conditions. The western side of the massif is hyper-oceanic, because of the Atlantic influence, with a gradient towards the east caused by a progressively stronger Mediterranean influence; especially in the valleys and on south facing slopes. This gradient is reflected in the transition between the European Environmental Zones (Metzger et al. 2005) from Lusitanian in the west, to Alpine South in the highest mountains and Mediterranean mountains on the lower slopes, especially in the east. The range forms the highest part of the Cantabrian chain which extends almost 400 km south west from the Pyrenees. It is geographically isolated from other mountains in the Iberian Peninsula, as reflected by the number of endemics, some of which are in common with the Pyrenees. Otherwise the region contains a complex mixture of phytogeographical elements including Alpine, Atlantic, Lusitanian and Mediterranean species as well as many plants with wide ranges throughout Europe. A correspondingly wide range of habitats is also present.

Methods

The initial botanical survey, using the nomenclature of the Flora Europaea (Tutin et al. 1993), took place in April 1993 and a replicate survey was completed in May 2009 (a seasonal difference of three weeks) and covered the Deva valley, Cantabria, Picos de Europa.

A detailed field survey of meadows was undertaken by Charlotte Mason College in 1992, to ground truth the maps and to identify all the area of meadows in the valley. Base maps showing meadow delineation were used (*Mapa Topografico de Espana Camaleňo* (81–1) and *Potes* (81–2)). 23 groups of meadows were identified by geographical proximity. The sample of quadrats thus comprised 23 stratified random square kilometres, with four random points in each square (a total of 92 quadrats). The meadow altitudes vary from 290 m to 1085 m, a range of 795 m. In 2009, map detail was further enhanced by employing 'Sigpac' (http://sigpac.mapa.es/fega/visor); as used by the Cantabrian Regional Government; which enabled a combination of aerial photography and maps to be used to locate the sites, which were predominantly on limestone; although some were located on calcium-rich sandstones and shales.

Sampling

Each point on the map was the central point for a 4 m^2 quadrat (Bunce & Shaw 1973) with a 5 m minimum distance from any track or field boundary. Notes of location and field sketches were made in 1993 to facilitate replication. 86 indicator species with minimal taxonomic challenges were selected; following research by Farino (1992) and through discussions between the authors; with an estimation of percentage grasses, percentage moss and percentage bare ground. The indicator species were selected as representative of the full species complement in order to cover the valley in the limited time available. Some key species for conservation were also included.

Species abundance was categorised on the Braun–Blanquet scale of 1–5: 1: 1 individual present; 2: <5% cover; 3: 5–25% cover; 4: 26–50% cover; 5: 51–100% cover. For calculations of comparative abundance, a median percentage was used (1: 1%; 2: 2.5%; 3: 15%; 4: 37.5%; 5: 75%) to pick up the species of low frequency.

In 1993, a pilot study of eight quadrats in which all species were identified was used to assess the reliability of indicator species as a measure of meadow composition. The results confirmed that indicators were reliable and could be used to determine the status of meadows, as the additional species recorded were common species and widespread in a number of different open habitats.

The co-ordinates of the point and the altitude were recorded on maps in 1993. In 2009 the points were located using a Global Positioning System (GPS). The mean height of the vegetation was estimated, and grazing and application of farmyard manure were also noted. In 1993 the following environmental variables were added: slope angle and aspect; water content; pH; and, depth of soil. The distance from villages and tracks was also recorded.

Surveyors in 1993, working in pairs, comprised trained botanists from Wageningen Agricultural University, the authors, and professional botanists who quality assured the samples (Buit & Moonen 1993; Prince 1993). By 2009, two of the professional botanists were still present for quality assurance, together with the first author. There were, therefore, three replicate surveyors.

The surveys comprised identical indicator species, although in 2009 each of the grasses was identified, whereas in some of the 1993 quadrats, only selected grasses were recorded and entered into the data analysis; however, all grass cover in both surveys was given an abundance category.

In 2009, all 92 quadrats were surveyed and categorised as *Match*, *Near match* or *No match*. *No match* included meadows with a change of land use (e.g., new building), or where insufficient or inaccurate information in the previous survey prevented location the meadow. *Near match* was in the same meadow, but a different location, and *Match* was as close as possible to the original quadrat position.

Analysis

A number of direct descriptive statistics on the basic data were analysed using Excel (Microsoft). The overall composition and relationships between the vegetation samples were carried out using multivariate analyses through TWINSPAN (Two Way INdicator SPecies ANalysis) and DECORANA (DEtrended CORrespondence ANAlysis) (Hill 1973, 1979a, 1979b). TWINSPAN classifies species and samples, producing an ordered two-way table of their occurrence. The process of classification is hierarchical; samples are successively divided into categories, and species are then divided into classes on the basis of the sample classification. DECORANA summarises the relationship between all species and samples through ordination, but removes the artefacts of a reciprocal averaging computation by detrending. It has been employed here mainly to analyse sample sites. These programs were used via the Community Analysis Package (Pisces) in 2009 and via mainframe computers in 1993 for species –sites analysis.

The data were also processed using MAVIS (Modular Analysis of Vegetation Information System, available from the internet) for Ellenberg scores and C–S–R comparisons (Grime 1979). MAVIS enables links to be made between botanical field data and a number of widely used classifications of British plant species. Data were inputted in the form of single species lists for each plot in each year. Each taxon is coded in terms of its frequency of occurrence within a group of individual samples The ouput is a standard description of the entered data in terms of each classification and allows comparisons to be made between plots and between years.

The mean Ellenberg values per plot indicate implicit shifts along the gradients of incident light, moisture, pH and substrate fertility, using unweighted indicator species (Ellenberg et al. 1991). These values have been used in a number of vegetation change studies across Europe (McCollin et al. 1999; Smart 2000), with validated correlations between Ellenberg values and environmental parameters (Ersten et al. 1998; Hill & Carey 1997). Changes in ecological conditions over time can therefore be hypothesised from mean score changes between the two sample dates.

Results

Basic data

Match

In 2009, 93% of meadows were a *Match* (47%) or a *Near match* (46%) relative to the 1993 data, i.e. these were reliable, replicate samples. In three of the meadows, i.e. 3%, there was a change of use; for example, to a garden or to a new building; involving a complete loss of meadow vegetation. These samples were therefore excluded.

Table 1
Changes in seven species between 1993 and 2009.

Species	Years	Mean abundance (%)	% Change
Rhinanthus minor L.	1993	1.17	+0.81
	2009	1.98	
Linum bienne Mill.	1993	0.45	−0.27
	2009	0.18	
Lotus corniculatus L.	1993	4.47	−1.99
	2009	2.48	
Daucus carota L.	1993	1.49	−1.48
	2009	0.01	
Ranunculus bulbosus L.	1993	4.34	−3.03
	2009	1.31	
Sanguisorba minor Scop.	1993	4.63	−3.05
	2009	1.58	
Sherardia arvensis L.	1993	1.14	−1.11
	2009	0.3	

Application of farmyard manure

Records from the survey in 1993 showed that 12 of the meadows (13%) were manured whereas only one (1%) was recorded in 2009.This decline confirms local observations that farmyard manure is no longer used in the fields and is now dumped in heaps on the edges of villages.

Abundance

The abundance of the majority of species had changed between the two samples. Using attribute percentages for the categorical data (mid-points as described above) a paired *t*-test was performed to determine if there had been a significant reduction in abundance between 1993 and 2009. The mean difference in species (mean difference = 6.25; SD = 7.64, $N = 88$) $t = 7.68$, one tail, ($p = 0.0005$), provides evidence ($t > t$crit) that there is a significant reduction in abundance and that the vegetation had become less complex.

Using matched species records, a positive or negative value was calculated overall and 95% of quadrats showed decreasing values in categories of abundance, confirming a major change in the structure of the vegetation. The percentage cover of grass in the quadrats was relatively stable, but the Braun–Blanquet scale is only sufficiently sensitive to detect major changes.

The change between 1993 and 2009 in the mean cover abundance of seven species indicative of diverse meadows is given in Table 1.These species are 'stress tolerators' (Grime 1979) in the Picos de Europa and are not able to compete against more vigorous species, and the increase in *Rhinanthus minor* L. is probably due to seasonal differences.

The complete species list together with occurrences in both years is shown in Appendix One, but the list of species with over five records is given in Table 2. There is no evidence of inaccurate identification of species.

Only seven species showed increases in species abundance from 1993 to 2009 and one of these, *Trifolium repens* L., is indicative of more fertile conditions. Otherwise the changes reflect decreases in species abundance, probably differentially through the population, thus *Bellis perennis* L. and *R. bulbosus* were probably lost due to the increased density of the swards of more fertile grasslands. Although *D. carota* declined to a great extent, subsequent observations have suggested that this could be due to the difference in the dates of

Table 2
Occurrences of species in 92 quadrats surveyed in both 1993 and 2009.

	N2009	N1993	Change
Ajuga reptans L.	7	15	−8
Anthoxanthum odoratum L.	43	48	−5
Anthyllis vulneraria L.	7	14	−7
Asphodelus albus Boiss	8	3	5
Bellis perennis	42	64	−22
Centaurea sp.	22	34	−12
Cerastium fontanum Baumg.	15	21	−6
Dactylis glomerata L.	26	43	−17
Daucus carota	16	39	−23
Echium vulgare L.	8	5	3
Erica vagans L.	4	6	−2
Geranium dissectum L.	20	13	7
Geranium sanguineum L.	6	3	3
Helianthemum sp.	11	17	−6
Hieracium pilosella L.	16	31	−15
Hippocrepis comosa L.	6	5	1
Holcus lanatus L.	36	33	3
Linum bienne	15	20	−5
Lithodora diffusa (Lag.)I.M. Johst.	5	3	2
Lotus corniculatus	55	60	5
Luzula campestris (L.)DC.	11	27	−16
Medicago sp.	27	36	−9
Medicago sativa L.	11	8	3
***Origanum* sp.**	4	11	−7
Plantago media L.	12	18	−6
Plantago lanceolata L.	80	82	−2
Potentilla erecta (L.) Raeusch	1	11	−10
Potentilla reptans L.	5	11	−6
Primula veris L.	7	8	−1
Prunella vulgaris L.	4	10	−6
Ranunculus bulbosus	39	61	−22
Ranunculus ficaria L.	5	8	−3
Rhinanthus minor	24	24	0
Rubus fruticosus L.	4	5	−1
Salvia verbenaca L.	6	4	2
Sanguisorba minor	44	55	−11
Saxifraga granulata L.	7	10	−3
Sedum sp.	8	11	−3
Sherardia arvensis	22	26	−4
Thymus sp.	1	5	−4
Trifolium pratense L.	62	74	−12
Trifolium ochroleucon Huds.	10	0	10
Trifolium repens	62	44	18

N2009 and N1993: number of quadrats (out of 92) where a species was present in 2009 and 1993 respectively for records >5. Species with 10 or more records in 1993 and a decline of more than 50% in occurrence are in bold. There are no records from 2009 with an increase of 50% occurrence.

recording between the two years. All the other declines are of more sensitive species indicative of herb-rich grasslands.

The table therefore emphasises that the frequency of 'stress tolerator' species has declined between the sample dates. However, species of higher fertility such as *T. repens* have increased.

Number of species

In the viable quadrats in 2009, species number varied in the range 2–17, with the mean being 10.39; in the matching quadrats in 1993, species number ranged between 3 and 22 (mean 12.36). This suggests that the mean number of indicator species has declined by 1.97 over 16 years, a decrease of 16%.

A paired *t*-test was performed to determine if there had been a significant reduction in species between 1993 and 2009. The mean difference in species (mean difference = −2.03; SD = 4.36, $N = 89$) $t = -4.40$, one tail, ($p = 0.0005$), provides evidence ($t > t$crit) that there is a significant reduction in number of species.

Therefore the overall species richness has declined in parallel with the other measures discussed above.

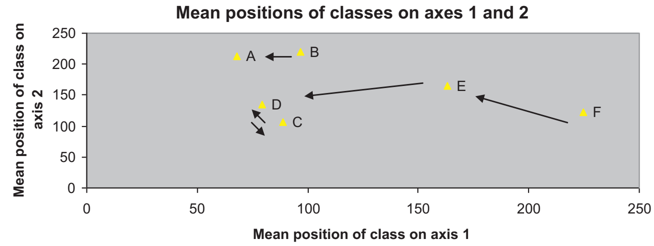

Fig. 1. TWINSPAN plots on DECORANA axes, classes A–F (TwoWayINdicatorSPeciesANalysis and DEtrendedCORrespondenceANAlysis).

Ellenberg and C–S–R scores

Quadrat data were inputted into MAVIS separately from 1993 and 2009 and thus, the mean scores from both years could be compared and shifts along the gradients determined (see Table 3).

Overall, the quadrat data showed an increase in competitor (C) species (53% of quadrats), a decrease in stress tolerator species (S) (60%) and an increase in ruderals (R) (68%). However, the last named is not considered a reliable indicator because of the limited number of species in the indicator list. The Ellenberg scores showed an increase in wetness (by +5.7 mean) and in fertility indicators (+10.2) between 1993 and 2009, and a decrease in pH (0.8) and light (−2.2) indicators. The decline in calcareous indicators is probably because sensitive species have been outcompeted by more vigorous plants. These results confirm the direction of change indicated by the previous tables.

Stands of stress tolerators may have been weakened by the greater abundance of competitor species. Mean Ellenberg scores showing a change to wetter, more fertile, but shadier conditions imply an increase in substrate fertility by the use of slurry and, in some cases, encroachment in the meadows by scrub and woody species.

It should be noted that MAVIS excludes non-British species but fewer than 10% of species are involved.

Multivariate analyses

The interpretation of the TWINSPAN output for 1993 and 2009 meadows, which is a combined analysis of both years, is partly on the indicators but mainly by the balance of frequencies between the groups. The usual arbitrary stopping rule for TWINSPAN was applied, i.e. to maintain a balance between too many groups and interpretability. Some diverse indicator species did not appear in 2009 as discriminants (e.g. *R. minor*, *L. bienne)* and *D. carota* does not appear at all in 2009 as a major indicator species.

Note that meadows are cut for hay or silage, but pastures are only grazed. The percentage figures given are from 2009 and therefore represent the extent of the current resource. The first four groups are generally on level ground with deep soils. Some of these groups contained manure heaps.

A Very fertile meadows (17%) dominated by grasses – virtually all cut for silage.

B Fertile meadows (1%) mainly dominated by grasses, but with some species more resistant to high nitrogen levels such as *Trifolium* spp. – usually cut for silage.

C Quite fertile meadows (7%) low sward – cut for silage or hay.

D Neutral meadows (25%) containing some diverse indicator species, such as *R. minor* and *L. corniculatus* and *G. dissectum* – usually cut for hay.

The following groups are separate from A to D on the ordination diagram (Fig. 1), suggesting that the vegetation of groups E and F is distinct from the more homogenous fertile groups.

These two groups are usually on steeper slopes away from villages and tracks. The vegetation is smaller in height.

Table 3
Distribution of quadrats by TWINSPAN (TwoWayINdicatorSPeciesANalysis) classes, 1993 and 2009, A–F, by absolute numbers and % change and associated competitor (C) and stress tolerator (S) mean scores (Grime 1979).

		2009	1993	Change	2009	1993	Change	Species	C mean score 2009	C mean score 1993	S mean score 2009	S mean score 1993
A	16	15	1	+14	94%	6%	+88%	*Medicago* sp –grass	2.75	3.00	2.39	2.33
B	13	1	12	−11	8%	92%	−84%	*Ranunculus* spp. – *T. pratense* – *C. fontanum*	2.40	2.43	2.60	2.53
C	16	6	10	−4	38%	62%	−24%	*H. pilosella* – *Sedum sp.*	2.61	2.55	2.75	2.62
D	39	22	17	+5	56%	44%	+8%	*G. dissectum* – *R. bulbosus* – *A. odoratum*	2.34	2.29	2.78	2.82
E	66	34	32	+2	51%	49%	+2%	*R. bulbosus* – *L. corniculatus* – *T. pratense*	2.11	2.09	2.99	3.18
F	28	11	17	−6	39%	61%	−42%	*E. vagans* – *S. minor* – *R.minor*	2.00	2.00	3.33	3.52
$\sum$	178	89	89									

E Infertile meadows (38%) containing the highest number of species in managed meadows including *R. bulbosus* and *S. minor* – usually cut for hay.

F Calcareous pastures (12%) characterised by calcareous grassland species and some heathland plants, *E. vagans* and *Helianthemum* sp. – not cut for hay

Competitor species show an increase in mean scores F–A with stress tolerators showing a decrease F–A. Table 3 summarises the directional change in meadow composition. Scores from only one quadrat are shown by an asterisk. Table 4 indicates the estimated mean percentage cover per quadrat of key species of these groups, as indicated by the TWINSPAN analysis. Although this shows relative abundance, the calculations are necessarily based on the median percentage scale.

Fig. 1 illustrates the TWINSPAN plots on DECORANA axes 1 and 2 and emphasises the major shifts that occurred in the vegetation, notably an almost complete shift from B to A over the two periods. The summary of directional change in 2009 is illustrated by arrows. If axis 1 represents the degree of calcareous influence and axis 2 represents fertility, then there is a directional increase in fertility towards more intensively used, fertile meadows, dominated by grasses, and a change towards less calcareous tolerating species. The major changes have taken place in groups A–D, which are at the most fertile end of the spectrum, although groups C–D/D–C show changes in both directions which might account for the stress tolerator figures. Groups E and F are relatively stable although the calcareous pastures have declined by 22%.

This change is due to the increased use of slurry, with the higher fertility causing competitive shading out of the sensitive species of open conditions. This confirms the conclusions reached in the sections above.

The mean pH values (by MAVIS) of the TWINSPAN classifications are 6.3 and 6.2 for meadow classes F and E respectively, falling to 5.8 for the other four categories; reflecting the decline of calcareous species.

Table 4
Mean percentage cover per quadrat of key species, groups A–F, with number of quadrats in parentheses using median percentage scale (highest percentages of species are underlined).

Species	A (16)	B (13)	C (16)	D (39)	E (66)	F (28)
Grasses	<u>49.8</u>	39.8	35.8	28.3	13.5	26.2
T. pratense	1.6	<u>15.0</u>	9.5	8.1	6.0	3.0
T. repens	2.8	<u>13.1</u>	4.7	2.6	2.5	1.6
C. fontanum	0.2	<u>1.0</u>	0.6	0.1	0.4	–
Sedum sp.	–	0.2	<u>1.6</u>	0.02	0.05	0.9
R. minor	–	0.4	–	<u>8.6</u>	6.0	2.5
L. corniculatus	1.0	2.5	4.4	<u>9.0</u>	4.4	7.8
G. dissectum	0.3	0.5	–	<u>1.3</u>	0.3	–
R. bulbosus	0.3	4.9	0.2	0.8	<u>5.6</u>	1.5
S. minor	1.0	2.5	1.9	4.5	<u>6.7</u>	4.9
E. vagans	–	–	–	–	–	<u>8.0</u>
Helianthemum sp.	–	–	0.1	0.1	1.1	<u>8.2</u>

Table 5
Summary DECORANA ordination plot.

	Neutral	Calcareous/neutral	Calcareous
Moist	*1*/2	*2*/1	*2*/2
Dry	*12*/6	*126*/120	*8*/16
Very dry	*8*/6	*12*/20	*10*/8

First number *italic* (1993 plots)/second number (2009 plots).

DECORANA of sites

Table 5 shows a summary of the DECORANA plot for the sample sites in 1993 and 2009 in a tripartite division of axes 1 and 2. The samples show a tendency towards fewer calcareous plots in 2009, thus confirming the results discussed above.

A paired t-test was performed to determine if there had been a significant change in samples between 1993 and 2009 on each of axes 1, 2 and 3. The mean differences in sample scores for $N = 89$ are shown in Table 6.

Therefore, on axes 1 and 3, one tail, ($p = 0.0005$), provides evidence ($t > t$crit) that there are significant overall changes in the sample composition between 1993 and 2009. The level of confidence is reduced on axis 2 ($p = 0.05$). There are, therefore, major shifts in the balance of the vegetation between the two sampling dates.

Discussion

In 1993, 35% of the meadows were still classed as herb-rich, diverse, infertile meadows. However, by 2009, these diverse meadows comprise fewer discriminating high diversity meadow species, with decreases in abundance of many species including *R. minor* and *R. bulbosus*. However, as the area of unencroached meadow land recorded by aerial photographs has also decreased (Dykes 2009), the core area of high quality meadows has probably remained stable; the major losses of species have been in the more fertile grasslands. These losses have implications for insect species which depend on the individual leguminous plants for larval feeding, e.g., *L. corniculatus* (Baura et al. 2006), and emphasises that measures are required to maintain a viable component of high value meadows.

The observed declines may be a function of the decrease in the application of manure and the shading out of sensitive species through increased use of slurry. Probably because of a shortage of labour, farmyard manure is no longer used, and is now dumped in

Table 6
t-test for paired samples on DECORANA axes, 1993 and 2009.

	Mean difference	SD	t	p
Axis 1	27.00	55.42	4.60	0.0005
Axis 2	−19.45	61.16	−3.00	0.05
Axis 3	41.09	60.75	6.38	0.0005

heaps on the village edges. Such manure releases nitrogen slowly compared with the use of slurry, which moves quickly into plants. The declines may also be a result of a decrease in small ruminants. The shift to species-poor meadows is probably indicative of the increasing intensity of management over the years, with diversification of income streams and a reliance on subsidies, although this trend is hard to quantify. The patterns of grazing in the higher altitude meadows may also have changed to a more extensive model, allowing cattle and sheep to roam across larger areas. As a result, these meadows have become low intensity pastures. This again leads to a loss of species, as shown by observational data and supported by current research (Busqué et al. 2008).

Small scale research projects in 68 matched meadows around the villages of Lon, Brez and Baró in the intervening years, confirm the changes in grazing in the valley, with the biggest changes observed between 2000 and 2002. However, there were local differences in grazing intensity. Excessive sheep grazing was reported in the Lon meadows, which showed the most apparent continuing decline in high diversity meadow indicators. The number of high diversity, infertile meadows has been shown to increase with greater distances from villages. In addition almost half were on steeper ground, presumably because of difficulties with machine access. The distance from the nearest track affects the level of management within very fertile and fertile meadows, but this was not influential for infertile meadows.

The detailed effects of management are difficult to quantify without interviewing farmers, which was not feasible in such a study. It is difficult to access information on the detailed changes in agricultural practices, although research by Rescia et al. (2008) in the Deva valley provides a good baseline. Interviews will only look at overall trends not specific fields and 'ground truthing' is necessary to determine botanical composition and an indication of management at this spatial level. The present results agree with the trends observed elsewhere in Europe in both lowland and mountain areas (Baura et al. 2006; Bunce et al. 2004; Hofer et al. 2011; O'Reilly 2010, for example).

The vegetation changes in the Deva valley over this period illustrate the ongoing changes from subsistence to subsidised farming, with European subsidies now accounting for more than 40% of farm income in the valley (Busqué et al. 2008). The livestock systems are now dominated by foreign cattle, which have displaced the local breeds (Rook et al. 2004) The mid valley areas have undergone cultural and landscape changes, with severe shrub encroachment in many higher, steeper meadows inaccessible by mechanised vehicles (Farino 2009). Goats, in particular, grazed marginal steep land in the past but they are now often housed in large modern barns for milk production for cheese. As a result, there has been an increase in forest and scrub land-cover on the steepest slopes of the whole valley (Dykes 2009). However, many grassland areas in the lower valley have not decreased because close proximity to settlements facilitates access and regulates management intensity (Reger et al. 2007).

Thus, the maintenance of semi-natural grasslands depends on traditional farming techniques (Baura et al. 2006) and the conservation of traditional landscapes is linked to biodiversity and sustainable management practices (Calvo-Iglesias et al. 2009). The future stability of the cultural landscape depends on economically and environmentally viable alternatives if the abandonment of the rural environment and reduced biodiversity of hay meadows is to be prevented. Knop et al. (2006) conclude that the Swiss agri-environment scheme, targeted at hay meadow conservation, preserves biodiversity. However, the development of Natura 2000 sites, based on European Directives, does not guarantee the maintenance and biodiversity for all areas because many habitats and species are outside protected areas (Mücher et al. 2009).

A communication paper on the future of the CAP (European Commission 2010) prepares the way for a revised budgetary and legal framework post-2013. It states that European agriculture needs to be competitive environmentally as well as economically, with one of the principal objectives being to maintain the territorial balance and diversity of rural areas where agriculture remains a major economic and social driving force. Such a balance is an 'important factor in maintaining a living countryside' (European Commission 2010). Recent legislation in Spain was introduced in 2007 (Law 45/2007: Sustainable Development of the Rural Environment) in an attempt to regulate and establish basic measures which favoured rural economic, social and environmental development. The law aims to improve quality of life for the inhabitants of the rural environment whilst promoting protection and appropriate use of ecosystems and natural resources (MARM 2011).

However, these recent CAP guidelines and the Spanish Sustainable Development of the Rural Environment Law have not been implemented in the valley, and research has highlighted the Deva valley as a social–ecological system in need of a participatory management system (Rescia et al. 2008). There needs to be policy implementation with the participation of farmers, who are the recipients of subsidies and the actual managers of the landscape, in order to handle landscape change in a socially desirable way. An agri-environment scheme might follow the framework established in England through the Environmental Stewardship Schemes (Natural England 2011), which follow the success of the Environmentally Sensitive Area (ESA) and Countryside Stewardship Schemes. Interestingly, the application of the ESA Scheme to the Picos de Europa has been suggested previously (Bunce et al. 1998). Such a policy could contribute positively to the maintenance of traditional landscapes and conserve upland hay meadows in the Picos de Europa. However, participatory management needs to be attractive to the valley's inhabitants if it is to be successful. At present, it is necessary to recognise that the pressure for conservation is mainly from extrinsic sources rather than from the farmers themselves.

Conclusion

The change in farming practices and consequent changes in vegetation composition of upland grasslands in the Deva valley are undisputed and are manifested in the decline of biodiversity observed in the present study. There has been a significant negative change in the abundance and number of calcareous and sensitive indicators, with corresponding increases in competitor species and a major change in the structure of the vegetation. According to the Government of Cantabria's Centre for Agricultural Research and Education (CIFA); despite the National Park, there is currently no implementation of the policies which are needed to maintain traditional agriculture management practices to safeguard the diminishing resources of herb-rich meadows, which are so important for nature conservation. At present, this is a much debated, central issue in Europe, which needs to be resolved; particularly because agri-environment schemes are costly. However, if the core practices of traditional farming are not maintained, this vital resource for nature conservation will be lost.

Acknowledgements

The authors appreciate greatly the two anonymous reviewers who spent so much time on improving the paper, and Freda Bunce's contribution to the final English editing.

Appendix A. Occurrences of species in 89 quadrats surveyed in both 1993 and 2009

N2009 and N1993: number of quadrats (out of 89) where a species was present in 2009 and 1993 respectively. 0 = species absent. Species with 10 or more records in 1993 and a decline of more than 50% in occurrence are in bold. There are no records from 2009 with an increase of 50% occurrence. Grass species identified only in 2009 are presented at the bottom of the table.

	N2009	N1993	Change
Acinos alpinus[a] Moench	0	0	0
Ajuga reptans	7	15	−8
Alchemilla glabra (agg.)[a] Neygenf.	0	0	0
Anagallis arvensis L.	1	0	1
Anthoxanthum odoratum	43	48	−5
Anthyllis vulneraria	7	14	−7
Aquilegia vulgaris[a] L.	0	0	0
Asphodelus albus	8	3	5
Astrantia major L.	0	1	−1
Bellis perennis	42	64	−22
Caltha palustris L.	1	0	1
Capsella bursa-pastoris Medik.	1	4	−3
Cardamine raphanifolia Pourr.	0	2	−2
Centaurea sp.	22	34	−12
Cerastium fontanum	15	21	−6
Colchium autumnale L.	1	3	−2
Dactylis glomerata	26	43	−17
Daucus carota	16	39	−23
Echium vulgare	8	5	3
Erica vagans	4	6	−2
Fagus sylvatica[a] L.	0	0	0
Erythronium dens-canis[a] L.	0	0	0
Gentiana verna L.	0	3	−3
Geranium dissectum	20	13	7
Geranium sanguineum	6	3	3
Helianthemum sp.	11	17	−6
Helleborus virdis L.	2	2	0
Heracleum sphondylium L.	2	2	0
Hieracium pilosella	16	31	−15
Hippocrepis comosa	6	5	1
Holcus lanatus	36	33	3
Juncus inflexus[a] L.	0	0	0
Juncus effusus L.	0	1	−1
Leucanthemum vulgare Lam.	1	0	1
Linum bienne	15	20	−5
Lithodora diffusa	5	3	2
Lotus corniculatus	55	60	5
Luzula campestris	11	27	−16
Malva moschata L.	2	4	−2
Medicago sp.	27	36	−9
Medicago sativa	11	8	3
Narcissus asturiensis[a] (Jord.) Pugsley	0	0	0
Narcissus triandrus L.	0	1	−1
Ononis repens L.	0	4	−4
***Origanum* sp.**	4	11	−7
Parentucellia latifolia (L.) Caruel	1	2	−1
Pinguicula grandiflora[a] L.	0	0	0
Plantago media	12	18	−6
Plantago lanceolata	80	82	−2
Potentilla montana Schur ex Nyman	4	0	4
Potentilla erecta	1	11	−10
Potentilla reptans	5	11	−6
Primula veris	7	8	−1
Primula vulgaris Hill	1	2	−1
Prunella vulgaris	4	10	−6
Quercus pyrenaica Willd.	4	0	4
Quercus ilex L.	0	2	−2
Ranunculus gramineus L.	0	2	−2
Ranunculus bulbosus	39	61	−22
Ranunculus ficaria	5	8	−3
Rhinanthus minor	24	24	0
Romulea bulbocodium (L.) Sebast. & Mauri	0	3	−3
Rosa sp.	2	3	−1
Rubus fruticosus	4	5	−1
Salvia verbenaca	6	4	2
Sanguisorba minor	44	55	−11
Saxifraga granulata	7	10	−3
Sedum sp.	8	11	−3
Sherardia arvensis	22	26	−4
Thalictrum minus[a] L.	0	0	0
Thymelea sp.	1	0	1
Thymus sp.	1	5	−4
Trifolium pratense	62	74	−12
Trifolium ochroleucon	10	0	10
Trifolium repens	62	44	18
Trollius europaeus[a] L.	0	0	0
Urtica dioica L.	1	1	0
Valerianella sp.	4	0	4
Grasses not specified in both years but presented here for **N2009**			
Brachypodium pinnatum P.Beauv.	20		
Briza media L.	4		
Bromus mollis L.	21		
Bromus sterilis L.	4		
Lolium perenne L.	11		
Nardus stricta L.	2		
Poa bulbosa var. *vivipara* Koch	8		

[a] These species were selected but did not appear in the quadrats sampled.

References

Baura, B., Cremeneb, C., Grozac, G., Rakosyc, L., Schileykod, A. A., Baura, A., et al. (2006). Effects of abandonment of subalpine hay meadows on plant and invertebrate diversity in Transylvania, Romania. *Biological Conservation*, *132*, 261–273.

Bergland, B. E. (1991). The Cultural Landscape during 6000 years in Southern Sweden – the Ystaad Project. In *Ecological Bulletin 41*. Copenhag: Muksgaard.

Buit, A., & Moonen, C. 1993. Vegetation and landuse in the Picos de Europa. Department of Physical Planning, Report, Waganingen Agricultural University.

Bunce, R. G. H., Pérez-Soba, M., Jongman, R. H. G., Gómez Sal, A., Herzog, F., & Austad, I. (2004). *Transhumance and Biodiversity in European Mountains*. Waganingen: IALE Publication Series nr1.

Bunce, R. G. H., Bell, M., & Farino, T. (1998). The environmentally sensitive area legislation in the United Kingdom and its potential application to the Picos de Europa mountains in north-west Spain. *Environmental Conservation*, *2*, 219–227.

Bunce, R. G. H., & Shaw, M. W. (1973). A standardised procedure for ecological survey. *Journal of Environmental Management*, *1*, 239–256.

Busqué, J., Bedia, J., & Cabañas, S. 2008. Research in rangeland pastoral systems in Cantabria (Spain). Presentation, Centre for Agricultural Research and Education (CIFA), Government of Cantabria.

Calvo-Iglesias, M. S., Fra-Paleo, U., & Diaz-Varela, R. A. (2009). Changes in farming system and population as drivers of land cover and landscape dynamics: the case of enclosed and semi-openfield systems in Northern Galicia (Spain). *Landscape and Urban Planning*, *90*(3–4), 168–177.

Dykes, N. (2009). Landscape changes in the Picos de Europa. MSc thesis, Manchester Metropolitan University.

Ellenberg, H., Weber, H. E., Dull, R., Wirth, V., Werner, W., & Paulissen, D. (1991). Zeigerwerte von Pflanzen in Mitteleuropa. *Scripta Geobotanica*, *18*, 1–248.

Ersten, A. C. D., Alkemade, J. R. M., & Wassen, M. J. (1998). Calibrating Ellenberg indicator values for moisture, acidity, nutrient availability and salinity in the Netherlands. *Plant Ecology*, *135*, 113–124.

European Commission (2010). Communication for the Commission to the European Parliament, The Council, The European Economic and Social Committee and the Committee of the Regions. The CAP towards 2020: Meeting the food, natural resources and territorial challenges of the future. Brussels: EC. Available at http://ec.europa.eu/agriculture/cap-post-2013/communication/com2010-672_en.pdf [accessed 16.10.11].

Farino, T. (1987). The Picos de Europa. In F. V. Granfield (Ed.), *Wild Spain*. London: Ebury, pp. 45–49

Farino, T. (1988). The glories of the meadows of the valleys of the Picos de Europa. *BBC Wildlife*,. July 1988.

Farino, T. 1992. Systematic list of the flora of the Picos de Europa. Report, Teresa Farino.

Farino, T. (2009). *Wild Spain: the plants animals landscapes*. London: New Holland.

Frame, J. (2000). *Improved grassland management*. Tonbridge: Farming Press.

Goldsmith, F. B., & García, G. (1983). *The meadows of the Picos de Europa Occasional paper 9*. London: University College.

Grime, J. P. (1979). *Plant strategies and vegetation processes*. Chichester: Wiley.

Hill, M. O. (1973). Reciprocal averaging: an eigenvector method of ordination. *Journal of Ecology*, *61*, 237–249.

Hill, M. O. (1979a). *TWINSPAN – a Fortran program for arranging multivariate data in an ordered two-way table classification of individuals and attributes*. New York: Cornell University.

Hill, M. O. (1979b). *DECORANA – a Fortran program for detrended correspondence analysis and reciprocal averaging*. New York: Cornell University.

Hill, M. O., & Carey, P. D. (1997). Prediction of yield in the Rothamsted Park Grass experiment by Ellenberg indicator values. *Journal of Vegetation Science*, *8*, 574–579.

Hindmarch, C., & Pienkowski, M. W. (Eds.). (2000). *Land management: the hidden costs*. Cambridge: Blackwell Science.
Hofer, G., Bunce, R. G. H., Edwards, P. J., Szerencsits, E., Wagner, H. H., & Herzog, F. (2011). Use of topographic variability for assessing plant diversity in agricultural landscapes. *Agriculture, Ecosystems & Environment*, *142*(3/4), 144–148.
Hopkins, A., Bunce, R. G. H., & Smart, S. M. (2000). Recent changes in grassland management and their effects on botanical composition. *Journal of the Royal Agricultural Society of England*, *161*, 210–223.
Jefferson, R. G. (2005). The conservation management of upland hay meadows in Britain: a review. *Grass & Forage Science*, *60*(4), 322–331.
Knop, E., Kleijn, D., Herzog, F., & Schmid, B. (2006). Effectiveness of the Swiss agri-environment scheme in promoting biodiversity. *Journal of Applied Ecology*, *43*, 120–127.
Losvik, M. H. (2001). Variations in species richness in hay-meadow vegetation at Voss West Norway, in relation to long-term management practices. *Norsk Geologisk Tidsskrift*, *55*, 77–84.
McCollin, D., Moore, L., & Sparks, T. (1999). The flora of a cultural landscape: environmental determinants of change revealed using archival sources. *Biological Conservation*, *92*, 249–263.
Marini, L., Scotton, M., Kilmek, S., & Pecile, A. (2008). Patterns of plant species richness in Alpine hay meadows: local vs. landscape controls. *Basic & Applied Ecology*, *9*(4), 365–372.
Metzger, M. J., Bunce, R. G. H., Jongman, R. H. G., Mücher, C. A., & Watkins, J. W. (2005). A climatic stratification of the environment of Europe. *Global Ecology and Biogeography*, *14*, 549–563.
MARM (Ministerio de Medio Ambiente y Medio Rural y Marino) (2011). Law for the Sustainable Development of the Rural Environment. Available at http//:www.marm.es [accessed 16.10.11].
Mücher, C. A., Hennekens, S. M., & Bunce, R. G. H. (2009). Modelling the spatial distribution of Natura 2000 habitats across Europe. *Landscape and Urban Planning*, *92*(2), 148–159.
Natural England (2011). Look after your land with environmental stewardship. NE124. Available at http://www.naturalengland.org.uk/es [accessed 28.11.11].
North Pennines Area of Outstanding Natural Beauty (AONB) Partnership (2011). Hay Time –North Pennines. Available at http://www.northpennines.org.uk/index.cfm?articleid=11799 [accessed 27.09.11].
O'Reilly, J. (2010). The state of upland hay meadows in the North Pennines. *British Wildlife*, *21*(3), 184–192.
Pedroli, B., van Doom, A., de Blust, G., Paracchini, M. L., Wascher, D., & Bunce, F. (2007). *Europe's living landscapes essays exploring our identity in the countryside*. Zeist: KNNV Publishing.
Prince, H.E. (1993). An evaluation of the conservation importance of hay meadows in the Deva Valley, Picos de Europa, Spain and the North-East area of the Lake District National Park, Cumbria, UK, using inter- and intra-area comparative techniques. Dissertation, University of Newcastle upon Tyne.
Reger, B., Otte, A., & Waldhart, R. (2007). Identifying patterns of land-cover and their physical attributes in a marginal European landscape. *Landscape and Urban Planning*, *81*, 104–113.
Rescia, A. J., Pons, A., Lomba, I., Esteban, C., & Dover, J. W. (2008). Reformulating the social-ecological system in a cultural rural mountain landscape in the Picos de Europa region (northern Spain). *Landscape and Urban Planning*, *88*, 23–33.
Rivas-Martinez, S., Diaz, T. E., Prieto, J. A., Loidi, J., & Penas, A. (1984). *La vegetación de la alta montana Cantabrica. Los Picos de Europa*. Lemesas: Leon.
Rook, A. J., Dumont, B., Isselstein, J., Osoro, K., Wallis DeVries, M. F., Parente, G., et al. (2004). Matching type of livestock to desired biodiversity outcomes in pastures – a review. *Biological Conservation*, *119*(2), 137–150.
Ruiz, M., & Ruiz, J. P. (1986). Ecological history of transhumance in Spain. *Biological Conservation*, *37*(1), 73–86.
Selman, P. (Ed.). (1994). The ecology and management of cultural landscapes. In *Proceedings of an IALE UK conference at Cheltenham 1993* Landscape Issues 11, 1, (pp. 64–72).
Smart, S. M. (2000). Ecological assessment of vegetation data from a nature reserve using regional reference data and indicator scores. *Biodiversity and Conservation*, *9*, 811–832.
Tutin, T. G, Burges, N. A., Chater, A. O., Edmonson, J. R., Heywood, V. H., Valentine, D. H., et al. (1993). *Flora europaea*. Cambridge: Cambridge University Press.
Waterhouse, T. (1996). Current trends in animal production practices in Northern Europe. In *Collected Papers of the European Network for Livestock Systems in Integrated Rural Development. Concerted Action EU: DEVI FAIR1 CTR50014*. Brussels: EU., pp. 7–9.