J Insect Conserv
DOI 10.1007/s10841-010-9332-0

ORIGINAL PAPER

# Grassland butterflies and low intensity farming in Europe

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Received: 24 March 2010 / Accepted: 18 August 2010


**Abstract** In this paper we describe the impact of the abandonment of traditional farming practices on butterflies and their habitats in traditional, often montane, pastoral systems. We link these declines to socioeconomic factors: illustrating how the failure of the CAP to support traditional farming leads to structural changes in farming enterprises—features which may be obscured by crude statistics on stock. We then call for the scheduled CAP reforms in 2013 to be radically realigned to support rather than destroy biodiversity so that any new EU agri-biodiversity commitments have an effective funding stream to support them.

**Keywords** Common Agricultural Policy · Grassland · Butterflies · Land-use change · Abandonment · Europe · Traditional management

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## Introduction

The European red list of butterflies lists a third of the 482 European butterfly species as having had population declines since 2000; 9% of species are considered threatened (van Swaay et al. 2010). The main processes are loss and degradation of habitat through agricultural intensification and land abandonment (and consequent fragmentation), much of it on grasslands (van Swaay 2002; van Swaay and Warren 2006; van Swaay et al. 2006, 2010). It is sobering to consider that much work in spatial ecology (see refs in Dover and Settele 2009) is not concerned with understanding processes in naturally fragmented systems, but with understanding and ameliorating anthropogenically mediated fragmentation that threatens biodiversity. It is perhaps ironic that many of the biodiverse systems in Europe which are now threatened by intensification and abandonment (Plieninger et al. 2006; Stoate et al. 2009; van Swaay et al. 2010) were created by human action in the first place as a side-effect of traditional farming practices (MacDonald et al. 2000; Plieninger et al. 2006). It is also easy, without data, to underestimate the scale of land-use change (and its impact on biodiversity) as people are adept at forgetting what landscapes were like when they were younger and hence unreliable witnesses when questioned. Younger members of a population are also likely to assume that there has been little change in biodiversity in their landscape because of a lack of communication between generations. Both phenomena are summarised by the term 'shifting baseline syndrome' (Papworth et al. 2009).

The Common Agricultural Policy (CAP) is the consumer of the lion's share of the European Union's budget and directly impacts on biodiversity. Wenzel et al. (2006) describe a shocking situation where seven remnant high quality fragments of calcareous grasslands, managed as nature reserves, lost more than 50% of their habitat specialist species whilst the generalists remained stable over the period 1972–2001. The results were interpreted as changes in the wider landscape surrounding the reserves impacting on the reserves either by reduced connectivity/loss of habitat or by affecting dynamics within the reserves. Designating nature reserves alone is not an answer to biodiversity conservation, the quality of the wider countryside is important too. Whilst the CAP does support agri-environment measures, it is clear that it is currently ineffective in protecting some of the most valuable biodiversity remaining on (high nature value) farmed land as well as in adjacent reserves (see van Swaay et al. 2010).

## High nature value farmland

The term high nature value farmland (HNV) was coined in the 1990s in an attempt to focus attention on traditional farming systems which support high levels of biodiversity, but which were likely to decline without additional support through the CAP (Beaufoy, undated). Projections made by Klijn (2004) suggest that marginal lands, including HNV, are threatened by wide-scale abandonment whilst Kleijn et al. (2009) point out that conservation is more cost-effective in "extensively farmed areas that still support high levels of biodiversity".

Existing EU policy explicitly promotes the support of low-intensity farming systems, primarily through Rural Development Programmes funded via Axis 2 of the CAP, but little progress has been made in the rebalancing of support from intensive farmland to HNV farmland (Beaufoy, undated) and there are conflicts within rural policy itself (Beaufoy et al. 2008). Paracchini et al. (2008) have refined initial attempts to map HNV farmlands (which used CORINE land cover data and information on farm systems) by incorporating biodiversity data from the Natura 2000 network, butterflies (Prime Butterfly Areas; van Swaay and Warren (2003)), birds (Important Bird Areas; Heath et al. (2000)) and national biodiversity datasets along with land cover and national-specific environmental data. Only Prime Butterfly Areas that included at least one of 27 species that use alpine, humid or dry grassland were used in the analyses (Paracchini et al. 2008).

Three criteria were used to identify HNV land

1. Farmland with a high proportion of semi-natural vegetation
2. Farmland with a mosaic of low- intensity agriculture and natural and structural elements, such as field margins, hedgerows, stone walls, patches of woodland or scrub, etc.
3. Farmland supporting rare species or a high proportion of European or World populations

The results gave an outline estimate of 32% of agricultural land as potentially of High Nature Value (using data from 26 of the EU-27 states) (Paracchini et al. 2008). This mapping approach has been criticised by Beaufoy et al. (2008) as not truly representing HNV farmland as it does not 'define' the HNV farming systems in member states, but is an "exercise in spatial targeting of support".

## Abandonment

Land-use change is a catch-all and can represent abandonment of management, change in timing or intensity of management, change in cropping, and destruction of habitat. HNV farmland is susceptible to two processes: intensification (conversion to intensive arable or increased grazing pressure in livestock systems) or, as much of it is on marginal land, abandonment. Such impacts can have catastrophic effects on grassland composition; for example, Foley (1987) identified eight separate factors accounting for the loss of 83 sites of *Orchis ustulata* L. in northern England: agricultural improvement (42%), building (22%), ploughing (8%), overgrazing (20%), scrubbing up (5%), afforestation (2%) and quarrying (1%). A further 20 sites were shown to have lost the species, but the causal factor could not be identified. Whilst the abandonment of farmed land can also be the end result of a complex of factors, the two main reasons are: (1) that it is no longer economic to farm (MacDonald et al. 2000) or (2) for social reasons (Stenseke 2006). The less accessible (Mottet et al. 2006) and least productive fields (MacDonald et al. 2000) are those most likely to be abandoned; this generality holds for whether the fields are in Europe (e.g. Sweden, Stenseke 2006) or Japan (Uematsu et al. 2010). Of 24 montane study areas in Europe, MacDonald et al. (2000) estimated 21 were suffering from abandonment. Few young people entering farming and rural depopulation are also common causes of abandonment (Rescia et al. 2008; Uematsu et al. 2010).

## Impacts of abandonment on the sward

Whilst the vegetation of lightly grazed meadows and traditional mown (hay) meadows is different (Erhardt 1985), continuity in management is essential in both cases for the maintenance of their characteristic flora. Abandonment of management leads to substantial declines of characteristic hay meadow species and succession through invasion by coarse grasses, shrubs and eventually trees (Erhardt 1985;

see also Jensen et al. 2001). Losvik (1999) found that overall plant species richness could be maintained, or even increase over a 10-30 year period, although in most cases succession to woodland would ultimately result. However, cessation of mowing for just 2–3 years reduces hay meadow plant species richness (Kull and Zobel 1991). Abandoned open meadows can become very nutrient poor, whilst those colonized by woody vegetation can become more fertile (Losvik 1999). The cessation of mowing, with its associated raking-off of the cut sward, will necessarily increase litter accumulation (Mašková et al. 2009) and may reduce seedling survival rates (Fowler 1988).

The abandonment of mowing need not immediately necessarily lead to conversion of an area to one dominated by woody vegetation; in the Pyrennes such meadows are often used, if only temporarily, as grazing meadows (Mottet et al. 2006). A switch from hay meadow to grazing meadow will maintain the open character of the field, but will have significant impacts on plant structure through the growing season and on plant composition and richness. For example, hay meadow and grazing management has been reported to result in different routes for the principal modes of reproduction by the toxic perennial false helleborine (*Veratrum album*) in subalpine Swiss grasslands (Kleijn and Steinger 2002). Essentially, hay meadow populations are mainly made up of small plants resulting from seed; whereas pastures have a higher proportion of large plants resulting from vegetative reproduction. The avoidance of grazing on the alkaloid-containing tissues is thought to result in increased vegetative growth whilst also promoting increased seed production (Kleijn and Steinger 2002). If the results of a study of *Scabiosa columbaria* by Reisch and Poschlod (2009) can be generalised, it also appears that plants in mown meadows flower earlier than in grazed meadows, and show genetic differentiation. Some meadows, such as those in the Picos de Europa, Spain, are artificially irrigated partly to prevent desiccation, but also to distribute waterborne macronutrients. Drainage channels are cut by hand through the meadows and water diverted from springs or streams as required. The abandonment of irrigation may precede abandonment of mowing, and is likely to modify plant communities in a similar way to the installation of drainage (Grootjans et al. 1996)

## Better to maintain than restore

Mottet et al. (2006) noted that abandonment of meadows was sometimes temporary, with management reinstated after 10–20 years. Restarting management may not be as good as maintaining it however: Stampfli and Zeiter (1999) reinstated mowing regimes in two hay meadow plots on the slope of Monte Generoso in Switzerland. The abandoned meadows had become dominated by *Brachypodium pinnatum* within 8 years of the cessation of mowing and 26% of the herb species lost. Over a 10-year period of restored mowing biomass increased, but growth was strongly affected by drought and was strongly correlated with relative humidity. Reinstating mowing did not lead to the recovery to pre-abandonment levels of species richness. Many of the meadow species in their study area had low persistence in the seed-bank (<5 years), and they concluded that if viable seed sources were not within a few metres of restoration management recolonisation would take a very long time and would also be affected by the microsite quality experienced by seeds. Eriksson and Ehrlen (1992) considered seed limitation to be an important, and underestimated, factor in recruitment in plant populations. Zobel et al. (1996) considered that mowing reduced shoot competition for light, allowing less-competitive species to survive, and removal of litter by raking improved seedling production. In effect, mowing constrains the process of competitive exclusion. The message of Kull and Zobel (1991) is clear: maintenance is simpler than restoration; when fully abandoned, estimates of timescales for restoration are of the order of 50 years due to impoverished seed banks and seed-rain.

## Impacts of abandonment on butterflies

Traditional grazing and hay management (mowing) are both valuable management techniques for butterflies and perhaps comparable: cessation of mowing may be compensated for by low intensity grazing, at least for many species (Dolek and Geyer 1997; Saarinen and Jantunen 2005). The expected trajectory for species richness of butterflies in abandoned traditional grasslands was outlined by Erhardt (1985): essentially, newly abandoned grassland increases in butterfly species richness as grazing/mowing pressure is relaxed (see Söderström et al. 2001), mainly a result of improved survival of juvenile stages, whilst plant species richness, which tends to correlate with butterfly richness (Cremene et al. 2005; Marini et al. 2009a, b), is maintained. As shrubs invade, plant diversity declines and shading increases. In a later paper on butterflies in the Swiss Jura Mountains Balmer and Erhardt (2000) indicated that the later successional stages of abandoned ('fallow') land, provided that it did not succeed to forest, was more valuable than extensively managed grassland for butterfly richness and called for the incorporation of the seral stage in conservation management as a rotation. This 'mosaic' approach to management intensity was also considered useful by Cremene et al. (2005), Kruess and Tscharntke (2002a), Marini et al. (2009a), and Schwarzwälder et al. (1997); the latter also considered that later seral stages in adjacent meadows were useful for complementation/supplementation (Dunning et al. 1992; Ouin et al. 2004). Stefanescu et al. (2009) reported changes in butterfly fauna in abandoned hay meadows over a 7-year period in northeast Spain, following a baseline year with hay management; a single continuously managed hay meadow acted as a control. The sward responded to abandonment typically through increased turf height, invasion by coarse grasses, brambles (*Rubus* spp.) and tree seedlings, loss of leguminous plants, colonisation by *Cirsium* spp. The principal results were that changes in the butterfly fauna were rapid with much change in species composition in the first 4 years, including the extinction of *Plebejus argus* along with its host plant *Lotus corniculatus*. Species richness did not change significantly in abandoned meadows, contrary to expectations, but species with specialist habitat requirements were replaced by generalists. The nearby, continuously managed control gained species; the assumption being that species relocated from the abandoned meadows.

With variations, elements of abandonment, e.g. succession following abandonment, leading to (often) initial increases in species richness, followed by decline as trees invade and displacement of grassland specialists by generalists, have been identified in many studies: e.g. Pöyry et al. (2005, 2006, 2009) in Finland, Kruess and Tscharntke (2002a, b) in Germany, Cremene et al. (2005) in Romania, Öckinger et al. (2006) and Nilsson et al. (2008) in Sweden. Whilst community trends may be similar, individual species' responses may differ in different geographic areas as Pöyry et al. (2005) caution.

The above studies were concerned, largely, with community effects; individual species studies reveal impacts of abandonment as well. A detailed study of *Mellicta athalia* ssp. *celadussa* in hay meadows, abandoned grasslands, and newly restored grasslands on Monte Generoso in Switzerland (Schwarzwälder et al. 1997) found distinct partitioning of activities by adult males between the different management regimes (mating in hay meadows and recently abandoned meadows, feeding in mature abandoned meadows); host plants and larvae were only found in the active hay meadows. Restoration of hay cutting in mature abandoned meadows did not result in colonisation by host plants after 2 years indicating that restoration management was a long-term process; Pöyry et al. (2005) found 'old-pasture' butterfly species had not recolonised restored pasture even after 5 years in Finland. Bergman and Kindvall (2004) considered that abandonment of grazing or mowing, and consequent succession, in meadows threatened the long-term survival of *Lopinga achine* in Sweden.

**An example from Spain (hay and grazing meadows)**

MacDonald et al. (2000) included in their paper a snapshot of the Picos de Europa, a prime butterfly area (van Swaay and Warren 2003): predominant farming: sheep and cattle; 26% of land area (180,000 ha total) used for agriculture; farm sizes 2–30 ha, % of farmers in the working population = 40% (45% of population in employment), 10 people per km$^2$, decreasing population with a "High dependence on farming for employment". They also considered that traditional land management practices and the land itself was being abandoned, with consequent impacts on the landscape. Rescia et al. (2008) defined the socio-ecological system in the Deva Valley of the Picos de Europa to be one of extensive mountain livestock farming. Cheese is a speciality product of the area (Álvarez 2006).

Rescia et al. (2008) assessed loss of open grassland in a study area at the head of the Deva Valley in the 'county' of Liébana, Picos de Europa: 51 meadow parcels were examined and 31 (61%) were shown, over the period 1957–2002, to have undergone changes in vegetative cover associated with succession (shrinkage and interspersion) whilst two were replaced by infrastructure. Dover et al. (in revision) working in the same general area (the municipality of Cameleño) estimated that most of the loss was associated principally with a decline in grazing meadows rather than hay meadows, though loss and shrinkage was evident in these too. In 1900 there were approximately 2,700 people in Cameleño, a decline set in around 1940, and by 1998 the population was about 1,100 (Gómez et al., undated). Within Liébana as a whole only the tourist town of Potes experienced a population increase over the same period. Between 1960 and 2003 Potes increased in population by 17.8% with all the other municipalities declining by between 49.6 and 83.6% (Gómez et al., undated) (Fig. 1). In Liébana as a whole, milk production declined by 43% in the 5 years 1995/1996 to 2001/2002 with only Tresviso increasing production (from 0 to 6,320 kg, equivalent to 0.18% of the total quota for Liébana in 2001/2002) (Fig. 2). Over the same period, the total number of dairy farmers in Liébana declined from 312 to 113 a reduction of 64%—only Tresviso had an increase in the number of farmers: up 1 from zero (Fig. 3) (Anonymous 2005; Gómez et al., undated). The number of livestock (meat) farmers in 2003 stood at 447 with a total headage (number of stock) for Liébana of 12,469 but with herd sizes varying between 10 and 35 per farm, depending on municipality (Gómez et al., undated). Over the period 1984–2003 livestock farms declined by about 50% (Fig. 4), with beef cattle numbers increasing by about 3% (see Rescia et al. 2008). The age structure of the farmers shows a top-heavy age structure compared with the population of Liébana (Rescia et al. 2008). Sheep farmers declined by

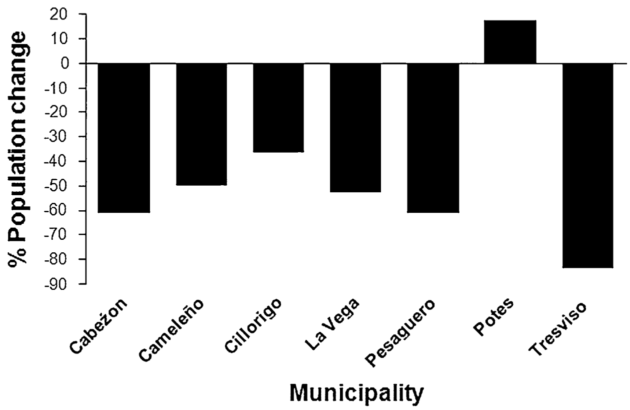


**Fig. 1** Percentage human population change in the seven municipalities of Liébana, Picos de Europa, Spain 1960–2003. Data from Padrón municipal cited in Gomez et al. (undated)

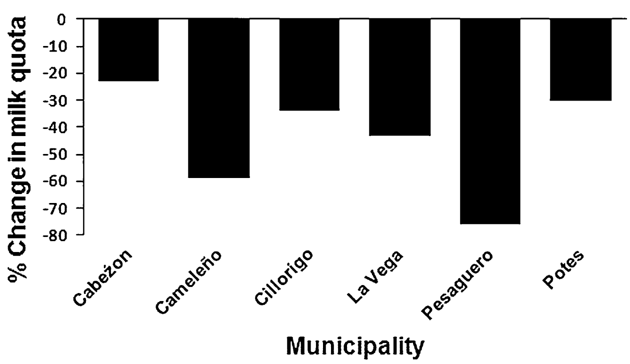


**Fig. 2** Percentage change in milk quota in six of the seven municipalities of Liébana, Picos de Europa, Spain between 1995/1996 and 2001/2002. Data from Consejeria de Ganaferia, Agricultura y Pesca cited in Gomez et al. (undated)

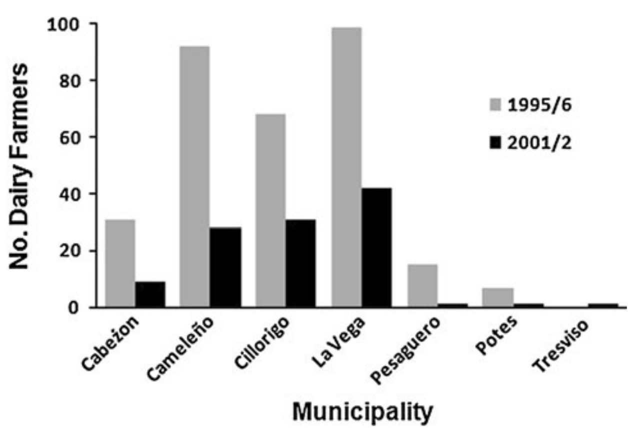


**Fig. 3** Number of dairy farmers in the seven municipalities of Liébana, Picos de Europa, Spain between 1995/1996 and 2001/2002. Data from Consejeria de Ganaferia, Agricultura y Pesca cited in Gomez et al. (undated)

65% to 161 over the period 1992–2003, and goat farmers by 45% to 63, sheep numbers declined by 79% over the same period (to 7,575 animals) whilst goats declined by

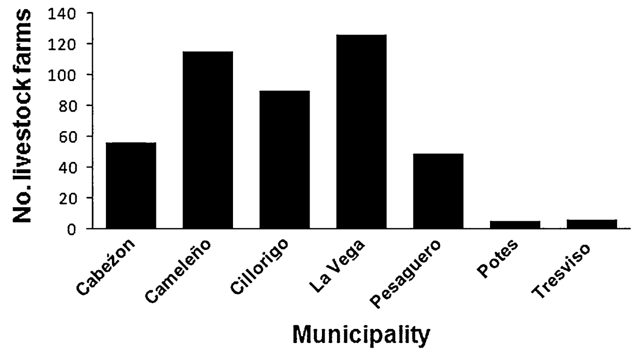


**Fig. 4** Number of livestock farms in the seven municipalities of Liébana, Picos de Europa, Spain in 2003. Data from Extension Agraria. Datos de la campaña de saneamiento 2003 cited in Gomez et al. (undated)

just 8% (to 4,972 animals) (data in Gómez et al., undated). Whilst all municipalities lost goat farmers, headage losses did not occur in Cameleño or Cillorigo (up 8.5 and 25% respectively) and this may reflect the needs of the cheese industry which is concentrated in these municipalities (Gómez et al., undated).

The results collectively demonstrate a collapse of traditional farming: the dairy industry has declined, remaining farmers have moved over to meat production. Massive depopulation of the rural areas has occurred, with tourism replacing farming as an economic activity. Shrinkage and loss of open grassland has been identified, with changes associated with the early stages of abandonment being picked up in the responses of the butterfly fauna (Dover et al., in revision, Dover et al., submitted). Whilst hay meadows appear relatively unaffected at present, the economics of cattle farming may result in an increase in barn-reared cattle, with hay imported from outside the valley—hay meadows are then likely to suffer the fate of grazing meadows.

## The example of Greece (shepherded grazing)

Traditionally, in Greece, grazing was unfenced and communal; sheep and goats were folded during the night, shepherded on mountain pastures during the day, and milked in the afternoon. However, observations suggest shepherded grazing in mountain regions has gone into a decline (SS, personal observation) and Zervas (1998) warned of problems of abandonment and overgrazing in Greek mountains. Hadjigeorgiou et al. (2005) cite some data on the depopulation and abandonment of mountain areas, where in the study area of Pindus Mountains a 46% decrease in cultivated land and a 43% decline in rough grazing was observed between years 1945 and 1992. The following data come from the Hellenic Statistical

Authority in Athens. National statistics (classified by altitude 0–300 m = lowland, 301–600 m = semi-mountainous, >600 m = mountainous) actually indicate an increase in the national flocks of sheep and goats over the period 1981–2000 (sheep up 2.3% to 8.75 million head; goats up 12.3% to 5.30 million head) although trends vary depending on location (Fig. 5). For sheep and goats the semi-mountainous areas have seen substantial headage increases (44.4 and 57.8% respectively) but declines are evident in both the lowlands and mountains for sheep (−12.1 and −14.7%), whilst goats have increased in the lowlands but decreased in the mountains (27.5 and −22.7%). Whilst goats have declined more than sheep and the declines in mountain areas are relatively modest, these figures do not reflect the *extent* of grazing declines observed in some areas. Despite the rise in sheep and goat numbers, there is a clear and substantial decrease in the number of farms over the same time period with sheep (−41.8% to 128,551) and goats (−60.6% to 138,251) (Fig. 5); the declines are less in the semi-mountainous areas for sheep, and goats (Fig. 6). A decrease in farms coupled with an increase in headage (Fig. 5) would suggest that enterprises would be intensifying stock production by feeding in barns rather than by shepherding or transhumance, especially as much of a sheep and goat farm's income derives from milk production requiring high hygiene standards and easy access to dairies (see Anonymous 2009; Kalantzopoulos et al. 2002). However, the national statistics do not demonstrate any increase in such 'home feeding' of animals; in the case of sheep and goats there are declines in both 'home-feeding' and transhumance, but large increases in the proportion of animals that were shepherded—especially for goats (Fig. 7). This seems extraordinary given that shepherded flocks made up 85.1 and 82.8% of the national sheep and goat flocks respectively. Recent legislative changes in Greece now require all dairy sheep and goat farms to have permanent sheds approved by the state authorities (Law N° 1579 of 1985 as amended by 3698 of 2008), in order to ensure milk and stock hygiene, animal welfare and manure management. However, this implies significantly higher costs for shed construction, restriction of shed locations near electricity and water supply networks, etc., therefore a greater emphasis on 'home-fed' or sedentarised flocks. It is possible that the historic method of data collection (registration of flocks) does not reveal the changes that are happening on the ground; the latest date that we have for comparisons is 2000 and much can happen in 10 years. The lack of up-to-date information on trends in husbandry, national flocks, their location relative to topography, and grazing pressure may hamper efforts to provide support for HNV farming (de Rancourt et al. 2006). For example, EU Sheep and Goat support has been '100% decoupled' for

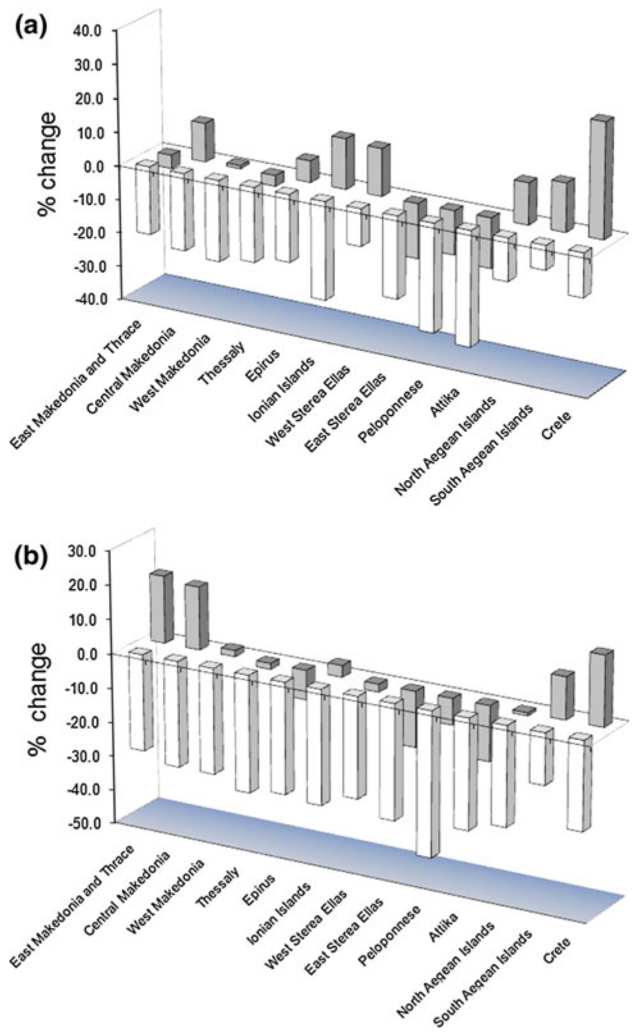


**Fig. 5** Percentage change in the number of farms (*white*) and headage (*grey*) for **a** sheep and **b** goats at different locations in Greece over the period 1991–1990. *Source*: Hellenic Statistical Authority, Athens

Greece i.e. converted to a "headage payment" based on historic rights (Stoate et al. 2009) so shepherds receive a payment based on their 2003 flock size. Whether these shepherded flocks still exist in some places is a moot point. Some mountain areas occupied by transhumant Vlachs are overgrazed (e.g Mt Smolikas), whilst other areas such as Northern Epirus, the Pindus Mountains and the Central Peloponnese are largely abandoned (SS and IH, personal observation). Anecdotally, shepherds on Mt. Chelmos indicate a ×10 decline in free ranging sheep and goats, but an increase in cattle.

Why is this important for butterflies? Pamperis (2009) gives the latest trend for 235 butterfly species found in Greece. Over the period 1983–2009, 24 species have declined by 15–25%, 7 by 25–50% and 1 by 50–75%, no species have been shown to increase although the geranium bronze *Cacyreus marshalil* has recently colonised. For

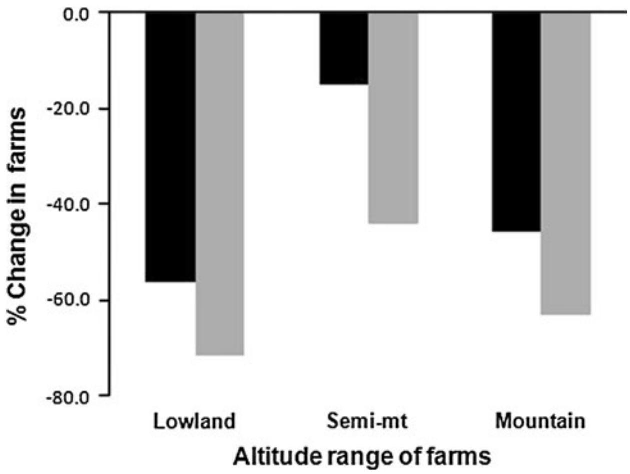


**Fig. 6** Percentage change in the number of Greek sheep and goat farms over the period 1981–2000 by altitude range. Hellenic Statistical Authority, Athens. *Black* sheep, *grey* goats, lowland = 0–300 m, semi-mountainous = 301–600 m, mountainous = >600 m


**Fig. 7** Percentage change in the number of Greek sheep and goats under different husbandry regimes 1981–2000. Hellenic Statistical Authority, Athens. *Black* sheep, *grey* goats

Greece ten sites are given as Prime Butterfly Areas in van Swaay and Warren (2003); most are mountain areas.

## Policy

Farm size is negatively correlated with both plant and butterfly species richness in Italian hay meadows; larger farms also utilise flatter fields compared with small farms (Marini et al. 2009b). The decoupling of farmer support under the CAP in 2005/2008 disadvantages small farmers and effectively promotes production in covered production areas (Stoate et al. 2009). There are clearly problems in maintaining traditional, extensive, sheep and goat grazing in montane areas (Anonymous 2009).

The entry of new states into the EU exposes them to the impact of the CAP; inflexible agri-environmental programmes can have disastrous impacts, including causing the extinction of some species, as happened to *Colias myrmidone* in the Czech Republic (Konvicka et al. 2008). Schmitt and Rákosy (2007) examined the impact of accession to the EU on Romanian agriculture and used butterflies as an indicator group. Their conclusion was that accession would result, as elsewhere, in intensification and abandonment of semi-natural grasslands with consequent negative effects on the country's butterfly fauna unless policy instruments directed funds towards their conservation and traditional management. Current agri-environment schemes are inadequate to prevent further biodiversity loss in the EU (Rounsevell et al. 2006). Giupponi et al. (2006) coupled the loss of small dairy farms in montane grasslands in Italy with biodiversity loss. They suggested that small dairy units might be retained by encouraging on-site/nearby processing units so that the monetary rewards from added-value products could be retained by local producers. Just such a system is evident in the Picos de Europa (Álvarez 2006; JWD, personal observation), and whilst it does seem to be halting the overall decline in goats in two of the seven municipalities in the county of Liébana, production is declining elsewhere. The implication is that any positive effect of processing plants would be very local and, whilst real, may be difficult to roll out across whole regions. Maintaining or even increasing milk production will not, of itself, maintain traditional grasslands as dairy stock can be housed in covered sheds—as happens in Pido in Camaleño (JD, personal observation)—support is needed for the traditional husbandry practices as well as for adding value locally.

Clearly changes are needed to the main support system in Europe (the CAP), and there does seem to be acknowledgement that reform is needed to support some producer systems, not least for their contribution to maintaining biodiversity (Anonymous 2009). Whilst agri-environment schemes designed to soften the impact of intensive agriculture are welcome and valuable, it is imperative that CAP reform in 2013 provides support for truly HNV farmland, and we suggest that the recommendations contained in (Anonymous, undated) by BirdLife International is a good starting point.

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