Agronomy Research **4**(1), 3–20, 2006

Review

Grassland for agriculture and nature conservation: production, quality and multi-functionality

A. Hopkins[1] and B. Holz[2]

[1]Institute of Grassland and Environmental Research (IGER), North Wyke, Okehampton, Devon EX20 2SB, United Kingdom
[2]LACOPE Project Management, Ingenieurbuero Holz and Partner Wollgrasweg 49, 70 599 Stuttgart, Germany

Abstract. European grasslands encompass a wide range of habitats that vary greatly in terms of their management, agricultural productivity, socio-economic value and nature conservation status, reflecting local differences in physical environment and economy, the effects of traditional practices and impacts of recent management. Widespread loss of biodiversity, as well as other environmental problems, have resulted from agricultural intensification or abandonment. Policies that have contributed to this have been progressively revised, initially by agri-environment schemes, and subsequently through changes in farm support payments and stricter regulatory frameworks, though many threats remain. We consider the agricultural implications of grassland biodiversity in terms of impacts on herbage production, feed intake and forage quality. Grassland biodiversity is both an externality of particular environments and farming systems and also contributes to objectives of multi-functional land-use systems. In addition to meeting species conservation and habitat protection, grassland biodiversity can contribute to enhanced value of agricultural products of regional, nutritional or gastronomic value, and to non-commodity outputs: agro-tourism, ecosystem functions linked to soil and water quality, and resilience to environmental perturbation. Needs and to conserve and improve the biodiversity potential of agricultural grasslands of typical moderate/high-input management, and for marginal, including communally managed large scale grazing systems, are considered using examples from contrasting areas of Europe. These include reindeer grazing in northern Fennoscandia, winter grazing in the Burren, Ireland, and cereal-fallow sheep grazing system of La Mancha, Spain.

INTRODUCTION

Intensification of management on both grassland and arable areas, and agricultural abandonment on many marginal grazing areas, have had profound impacts on the nature conservation value and landscape integrity throughout much of Europe. In regions where intensive livestock production is practised, biodiversity has been greatly reduced, not just in terms of the numbers of plant species within pastures or meadows, and the genetic diversity of those populations, but in terms of the loss of fauna in and beyond the farmed area. These consequent losses arise from eutrophication of the soil and sward, resulting in either reductions in the flora and fauna of associated habitats such as field margins, hedges, streams and ponds etc. affected by eutrophication or, in many instances, habitat loss. The effects of nutrient and other inputs such as inorganic fertilizers, purchased feed, slurry applications and pesticides, and of field operations including reseeding, herbicide applications, hedge and ditch removals, drainage work, silage in place of hay, and of greater livestock densities have all contributed to biodiversity loss and impacts on the wider rural environment (e.g. Soule & Piper, 1992; Haggar & Peel, 1993; Marrs, 1993; Nösberger et al., 1994). Managing for improved biodiversity and conservation objectives on such land is particularly challenging and needs to be seriously addressed.

On low-intensity livestock farmland, however, natural and semi-natural grasslands remain and are highly variable, often supporting considerable botanical diversity at a local scale, providing habitats for invertebrate and other faunal groups and delivering a range of ecosystem and socio-economic functions. These areas cover roughly 15–25% of the European countryside (EEA, 2004). Their variability reflects not only the underlying environmental conditions (soil type, geology, latitudinal and altitudinal ranges, and the effects of the pronounced west-east oceanicity-continentality gradient), but also the anthropogenic effects of pastoral farming systems. The role of past farming in creating and maintaining particular landscapes and biotopes is often not fully appreciated, even by ecologists (Bignall & McCracken, 1996). Settled agriculture in Europe developed over a long period in which the natural vegetation was either modified by livestock, supplanted by crops, or survived as remnants on field boundaries or sites that were marginal or inaccessible. In low- and medium-intensity farming areas the present-day rural landscape is therefore viewed either as having fragmented areas of the former wilderness habitat surrounded by improved agricultural land or, more logically, as a farmland biotope requiring management for biodiversity consistent with agricultural production and other ecosystem functions.

In this paper we consider the impacts of recent agricultural change and policy developments and the management, farming practices and techniques required on grassland and other pastoral ecosystems that can be implemented to help deliver nature conservation objectives. The links between biodiverse grassland systems and food product quality, grassland biodiversity and ecosystem functions, and the potential for grassland biodiversity in the context of wider community benefits are explored. Systems that represent typical agriculturally improved grassland are considered. Large-scale, including communally managed large scale grazing systems, are described using examples of semi-open landscapes from contrasting areas of Europe, based on a current European research project LACOPE (www.lacope.net).

1. Biodiversity and agricultural management of enclosed and improved grassland: some general principles

Plant species diversity in grassland, the actual species that are present, their relative abundance and the vegetative structure of the sward are largely determined by, (1) soil nutrient status and its modification by addition of fertilizers, liming and organic manures, including dung and urine from grazing animals; and, (2) defoliation and other disturbances, primarily through the intensity and frequency of grazing, or the timing and frequency of mowing, and by other natural environmental stresses (flooding, drought, fire, burrowing) or farming activities (cultivation, oversowing, drainage work, harrowing, herbicide applications etc.). The 'humped-back' model of Grime (1979) summarizes these phenomena in terms of species density in relation to stress/disturbance and in relation to increased standing crop. This interpretation is consistent with observations that increased nutrient additions, particularly nitrogen

(Charles & Haggar, 1979) and phosphorus (Janssens et al., 1998), lead to dominance by a small number of plant species (in temperate European lowland mesotrophic grassland these frequently include *Lolium perenne, Phleum pratense, Dactylis glomerata, Rumex obtusifolius;* on marginal and upland swards the shift may be towards *Agrostis capillaris* and *Holcus lanatus*). Although regular defoliation of swards is necessary to maintain biodiverse as well as species-poor grassland, frequent cutting and grazing arrest the flowering and seeding cycles of dicot species. Thus, agronomic practices that lead to agriculturally improved herbage production and/or forage composition are essentially incompatible with aims of management for sward biodiversity. Furthermore, losses of diversity resulting from soil nutrient inputs are largely irreversible because of high residual fertility of soils, thus limiting opportunities for enhancement of diversity by de-intensification of management. These aspects are discussed elsewhere in these proceedings (Isselstein et al., 2005).

Grassland biodiversity and agricultural land management: the policy context

Until the mid-20th century (more recently in some areas) agriculture was generally of low intensity and benign for nature, providing living space and habitat diversity alongside the utilization of land for food production. Agricultural intensification became accepted and justified by the compelling imperatives of increased food requirements and food security of the expanding urban populations in the post-World War 2 period, and of raising incomes and working conditions for (a declining number) of farm producers. International recognition of the negative impacts of modern farming on nature, water quality, human health and the wider environment increased incrementally from the 1960s (Carson, 1962) although the roots of these concerns and the needs to incorporate nature and non-material outputs with agriculture have earlier origins (Worster, 1994). The scientific understanding and public awareness of the scale and effects of agricultural chemical inputs and mechanized operations, the cost to taxpayers of supporting food surpluses in Europe in the late 1970s, and concerns over energy dependence all became major drivers of, what was to become, a slow policy change. Successive measures have been adopted, at least within the EU (with similar legislation also adopted in some non-EU countries), to incorporate nature and landscape conservation within agricultural land, (EC, 1985; 1992) and limitations set on emissions of nutrients and pesticides. EU Nature conservation policy is based on two main pieces of legislation: the Birds directive and the Habitats directive. Its priorities are to create the European ecological network (of special areas of conservation), called NATURA 2000 and, importantly, to integrate nature protection requirements into other EU policies such as agriculture and regional development (Europa, 2005). Discussion of this is beyond the scope of this paper (see Buller et al., 2000) for an account of its effects in different European countries), but an essential feature has been the implementation of measures that provide farmers with financial incentives to encourage management that protects biodiversity and landscapes combined with specific regulatory actions. For many countries, including the UK, protection of biodiversity within agricultural habitats also became a commitment under the terms of the Convention on Biological Diversity in 1992. The CAP is now increasingly aimed at delivering benefits to wider society including environmental protection and the conservation of nature and landscapes. This is not just seen as meeting environmental preferences, but as essential for developing the long-term potential of rural areas and sustaining livelihoods, encapsulated in the Killarney Declaration and the Malahide Commitments of 2004, and the 2010 targets of the European Biodiversity Strategy. The present structures imply a long-term commitment to maintaining biodiversity objectives within the farmed environment. For grassland scientists this poses a number of challenges in terms of how these objectives can be met within the context of profitable and sustainable farming, with quality food production and wider ecosystem and socio-economic benefits.

Grassland biodiversity and agricultural output and utilization

Herbage production and sward biodiversity

The potential for greater herbage production from grass mixtures compared with monocultures was claimed by Charles Darwin in the 19th century, and underpinned the advocacy of complex sown mixtures for grassland reseeding during the early twentieth century (Elliot, 1946). Advances in plant breeding and grassland agronomy, and the need for managing swards for specific objectives such as high quality silage, subsequently contributed to a progressive simplification of sown grassland and a reduction in grassland species diversity, particularly on intensively managed land (Frame et al., 1995; Casler, 2001). Management inputs have effectively been used to substitute for some of the attributes such as spatial and seasonal complementarity of different species and functional groups, and positive species interactions. It is paradoxical, though timely, that in the context of multifunctional land use and sustainable management, that there should be further consideration of the relationships between species diversity and production.

Several recent research projects in Europe and North America have shown that greater plant species diversity can lead to increased plant productivity, with associated nutrient retention and ecosystem stability (Hector et al., 1999; Tilman et al., 2001). Based on multi-site European experiments, Hector et al. (1999) found that 29 of 71 common species had significant though small effects on productivity, with one species, *Trifolium pratense*, having the greatest effect. Thus, increased productivity with species richness is not a simple one of species numbers - since productivity can saturate at a relatively low number - but of functional groups, of which the presence of legumes, long recognized by grassland agronomists as essential components for production, was a key feature (Spehn et al., 2002). These findings and concepts link evolutionary biology to ecosystem functioning and have both direct and indirect implications for agricultural grassland management. But since they are derived from measurements made on soils without fertilizers or other agricultural inputs, and are based on cut plots on prepared sites, we must consider further other situations typical of agricultural grassland.

In multi-site field experiments in the UK in the 1980s, grassland production from permanent swards was compared with that of newly sown *Lolium perenne* at a range of fertilizer N rates. First-year sown *Lolium* swards had greater herbage yields than identically managed permanent swards of more diverse composition; this applied across the fertilizer response range and at all sites. In subsequent years differences in productivity were, in most cases, higher on the permanent swards under nil-N inputs, and were similar on the two sward types under inputs of 150 kg fertilizer N ha^{-1} $year^{-1}$, but with some relative advantages of permanent grassland being apparent on sites that had the oldest and most botanically diverse swards. Production advantages to sown swards occurred when fertilizer inputs above 150 kg fertilizer N ha^{-1} $year^{-1}$ were applied (Hopkins et al., 1990; 1992) from which we must conclude that the sward diversity/ productivity correlation applies mainly under low-input situations. In low-input situations, relatively high herbage production can, in some cases, be realized from species-diverse grasslands; but in the context of agricultural management it is relevant to examine further how this translates into animal nutrition, through effects of herbage on intake and nutritional quality.

Grassland biodiversity and herbage intake

Grazing animals on mono-specific swards have limited choices: to graze or not, or if the sward has some structural variation, to select between different fractions of leaf and stem within the canopy. In a two-species sward, such as a grass and a legume (e.g. *Lolium perenne* and *Trifolium repens*), the choice is greatly extended as the two species have different nutritional and structural characteristics; the responses to this choice vary over diurnal and seasonal timescales (Rutter et al., 2004). From studies of grazing behaviour at this simple two-species level of sward biodiversity, sheep and cattle show a partial preference for *ca*. 70% clover in their diet (Nuthall et al., 2000; Champion et al., 2004). Relative intake of grass and clover has been shown to be considerably different than their proportions in the sward on offer (Champion et al., 2004; Rutter et al., 2004; 2005), implying selectivity and searching out of a preferred feed (in this case, white clover, particularly in daytime, and higher fibre grass in the evening). Furthermore, the animals' desired proportions of clover, and, at least for sheep, the total herbage intake, are affected by the spatial scale (patch size) of clover and grass within the sward.

In sown multi-species swards, or permanent swards of diverse botanical composition, livestock may be presented with an array of choices of species and plants at different growth stages, reflecting differences in content of carbohydrate, N, fibre, and possibly also of minerals, condensed tannins and other secondary metabolites. The implications for grazing preferences and intake rates on biodiverse grassland are therefore considerable, though at present poorly understood. In one example (using goats) selection from amongst a choice of different grass species tended to maximize intake rate, with only a small amount of residual variation explained by the individual preference for each grass species (Illius, 1999). Information of intake and feeding preference of non-legume dicot species in diverse swards is sparse, although a number of studies have shown that intake of fodder-based rations increases as the proportion of grasses declines relative to that of legumes and fine herbs (Jans, 1982; Lehmann & Schneeberger, 1988). The consequences of this effect show up in milk yield of dairy cows, with a reported 40% greater milk production potential from a green fodder diet with a grass: herbs+legume ratio of 40:60 compared with a ratio of 90:10. Similar evidence of higher intake from botanically rich green fodder, compared with maize silage, has been reported for fattening bulls (Lehmann & Schneeberger, 1988). There are, however, plant morphological characteristics and sward responses to environmental stress that can limit intake on some types of biodiverse pastures. Tallowin et al. (2002) reported on the agronomic constraints of using Cirsio-Molinietum fen meadow in the UK for summer cattle grazing. Low animal growth rates, low herbage energy value, and some mineral imbalances and deficiencies were identified and these increased from summer to autumn. Thus, we can identify situations where biodiverse pastures provide resources for high intake of nutritionally adequate forage and others where this is not the case. Intake of biodiverse forage will depend on the characteristics of the plants species present and their growth stage. Many grassland dicot species have evolved adaptations as potential defence strategies against herbivory, including anti-feedant secondary metabolites, structural reinforcement (e.g. spines or toughened leaves) and adaptive growth forms (e.g. basal rosettes and lignified stems) (Herms & Mattson, 1992). There is, therefore, an *a priori* inference that herbivory and thus intake of some dicot species will be lower than for grasses and forage legumes, especially under continuous as opposed to rotational or short-term grazing, or under environmental stress. The issue is further complicated by the consideration that some plant secondary metabolites may have evolved for other plant survival strategies (e.g. to attract pollinators) and thus not necessarily deter herbivory. Other metabolites contribute to the 'goals' of grazing animals for acquisition of nutrient requirements, including supplying fibre needed for rumen function.

Grassland biodiversity and herbage quality

The feeding value of herbage depends on intake, nutrient content and nutrient availability of the herbage once ingested. One of the strongest agronomic reasons for reseeding botanically diverse permanent swards is to create a grass crop, consisting of a few species (sometimes one species) and varieties with defined and predictable nutritional values at a given growth stage. This is particularly important for situations such as high-quality silage, where conserved grass forms the main ration of winter-housed livestock (Beever et al., 2000) and under some grazing situations also. Many of the plant characteristics that affect intake by livestock, such as stem: leaf ratio, differ between grass species and vary with growth stage, and are reflected in herbage nutrient content. Comparisons of herbage sampled from identically managed pure sown L. perenne swards and permanent swards of mixed species composition have shown that L. perenne usually has higher digestible organic matter content (digestibility value) at the same sampling dates (Hopkins et al., 1990). The presence of some legumes, particularly Trifolium repens, can maintain the overall sward digestibility over a longer period because its leaves and petioles are replaced as it matures. The presence of other leafy herbs in a diverse sward might be expected to confer similar advantages, particularly deep-rooting species that maintain active leaf growth in summer during periods of soil moisture deficit; conversely, other herbs that develop a high proportion of stem and high fibre content as they mature would result in a net reduction in digestibility.

The nutritional quality of herbage has wider dimensions than the simple token measure of digestibility, and species diversity in grassland is the basis for a wide variation in biochemical properties (Rychnovska et al., 1994). There are trade-offs between widely understood measures of herbage quality and other traits that affect agricultural utilization. One widely recognized instance of nutritional advantages for livestock linked to sward composition is through condensed tannins (proanthocyanidins) present in a number of pasture species including trefoils (Lotus spp.). These can improve protein digestion, reduce bloat, reduce the burden of intestinal parasites and possibly reduce methane emissions from livestock (Aerts et al., 1999; Waghorn et al., 2002). Horses in particular are often considered to benefit from having access to diverse pastures containing herbs (Allison, 1995). A number of the frequently occurring herb species of mesotrophic grassland contain higher concentrations of mineral elements (Ca, Na, K, Mg) than in the associated grass components (Hopkins, 2004), and this function may be important on sites low in particular soil nutrients. While the content of most vitamins in herbage is not considered to be a significant issue for ruminant nutrition that of vitamin E has livestock health implications and affects milk and meat quality (Beever et al., 2000). The content of vitamin E in relation to pasture type, species, and plant age, as with many other aspects of plant species and animal health, remains relatively poorly understood. Potential benefits linked to plant biochemical diversity are unavailable to livestock under more intensive production systems, although this must be balanced against the possible exposure to toxic or other injurious plants that can occur in botanically diverse swards.

Grassland biodiversity as both an output and an input of multi-functional land management

Biodiversity, where associated with agricultural production, has largely been regarded as a positive externality to the process of food production, and as a product (environmental good) of a particular (usually low input) farming system and environmental influences such as hydrology, pH and past ecological succession; in this context it may provide a range of benefits to wider society without necessarily conferring significant direct benefits to the producer. Furthermore, farm output under management required to maintain biodiversity may be inadequate to meet the income expectations of farmers in the 21st century. The basis of most environmental management agreements implemented since the 1980s is that society at large, through taxation, contributes to the cost of environmental goods and compensates the farmer for additional costs or income foregone. Development of long-term management strategies for integrated sustainable agriculture and conservation requires a better understanding of the potential values to farmers and wider society of farmland biodiversity. There is a need to identify and better understand situations where biodiversity is an input to, as opposed to simply an output of, agriculture and other land use functions; these can include food quality, agro-tourism, and ecosystem functions.

Food quality

There is increasing emphasis on the marketing of niche food products by geographical origin, method of production, gastronomic value and nutritional and health properties, and thereby improving financial returns for farmers and the wider rural economy. Production in which grassland biodiversity is an input to the livestock production food chain is embedded in some speciality systems, notably in mountain areas of Europe (Peeters & Frame, 2002). The diet of ruminant animals can affect not only taste but also the chemical composition of meat and dairy products produced, with consequences for human health; examples include the ratio of mono-and poly-unsaturated fats, cardio-protective omega-3 fatty acids, and content of minerals and anti-oxidants (O'Keefe & Cordain, 2004; Wood et al., 2004). Further understanding of particular sward types and plant species components of ruminant diets, including their utilization by local livestock breeds, in relation to the quality and value of meat and milk products may provide increased economic opportunities for producers. There are also marketing opportunities that target the ‘green consumer’ who is prepared to pay a premium for produce linked to a environmentally acceptable production system, in the same way as certificated organic produce. Floristically diverse grasslands also have the potential to provide nectar sources for honey bees and thus an additional high-value consumer product.

Agro-tourism

The growth of tourism and increased popularity of outdoor recreation also confers direct economic advantages to areas where conservation of biodiversity, wildlife and scenic landscapes have been achieved. The potential for landscape and wildlife interest to contribute to the rural economy is affected by agricultural management and grassland species composition, and these in turn affect habitat quality, e.g. for freshwater fish, bird life, etc. (Vickery et al., 2001). The value to local rural economies of green tourism can often exceed that directly attributable to farming (Pretty, 2002). While this income may not always benefit farm businesses directly, it can provide diversification opportunities for individual farmers, e.g. direct retailing of local produce, provision of accommodation, facilities for angling, game shooting etc. A number of web-based marketing initiatives have been adopted that link on-farm biodiversity with agro-tourism, and partnerships between conservation organizations and farms such as the Green Gateway scheme in south-west England (Devon Wildlife Trust, 2005) have been set up.

Ecosystem services

Biodiverse grasslands, and the management required to maintain them, may also deliver or contribute to other ecosystem services such as catchment management and carbon sequestration. Farmland vegetation has an important role in surface catchment hydrology, with potential effects on rates of run-off from slopes and subsequent discharge from rivers and streams, and the vegetation and its management can further affect water quality in terms of transport of sediments, micro-organisms and nutrients. Thus, we can consider differences in sward structure, and the presence of trees, shrubs and species that improve stem flow or affect soil structure and thus percolation, as well as wetland areas such as reed beds that act as filters (Worrall, 1997) as having important economic implications. In this context the role of scrub perhaps needs to be reconsidered as a component of managed grassland, including intensively farmed grassland, rather than simply an environmental problem associated with inappropriate management and under-utilization (FACT, 2003). The need for a better understanding of how different sward types and their associated vegetation affect surface hydrology and water quality is gaining importance in the context of climate change. Increased frequency of high intensity rainfall events leading to downstream flooding, and changes in the seasonal distribution of rainfall with more summer droughts are projected to occur with greater frequency under future climate change scenarios (IPCC, 2001). Allied to these changes, botanically diverse swards may provide greater resilience to droughts or other environmental perturbations partly through a wider gene pool (Tilman & Downing, 1994; Dodd et al., 2004). Carbon sequestration of soils is accountable under the Kyoto Protocol and there is significant potential to increase C sequestration by changes in grassland management (Jones & Donnelly, 2004; Sousanna et al., 2004). Options for achieving increased C sequestration include increasing the area of long-term grasslands by reducing short-term leys, maize and arable crops, as well as maintaining existing permanent grass, particularly peat grasslands as carbon sinks are important (Freibauer et al., 2004).

2. Biodiversity and multi-functionality of large-scale grazing systems: general principles

In Europe there still exists a variety of livestock systems which are based on migration. Transhumance (Bunce et al., 2004), and the year-round migration of cattle (e.g. in Mediterranean countries) and reindeer herds (Northern Fennoscandia) are organized on a communal or co-operative basis. There are also grazing systems which are privately or co-operatively managed, making use of marginal land, rough grazing land, stubble fields in agricultural landscapes and natural complexes. The succession cycles of vegetation communities with smooth gradients (ecotones) between fallowed and pastured land are an intrinsic part of large-scale grazing systems, with a gradient from young stages with open soil (disturbed patches) to a closed vegetation cover.

A common feature of many high biodiversity value grazed habitat complexes is their strong dependence on large-scale grazing systems. The size which can be considered as large depends on the regional nature and the regional traditions; the range is from < 1 km² (e.g. commonages in Connemara, West Ireland) to several 1000 km² (e.g. reindeer grazing migration between the northern Swedish lichen tundra and the Norwegian mountain chain in the Dividalen region). Large-scale grazing causes high structural diversity (γ-diversity) at different scale levels which provides a broad range of habitat requirements for high α- and β-diversity (diversity of species and communities).

From an ecological point of view these pastoral systems have also contributed to the development of unique landscapes that have almost disappeared from the more intensively managed surrounding areas. They can provide and maintain habitats for species which depend on ecotones between, e.g., forest and grassland, and on specific landscape features and disturbance regimes. These systems have a socio-economic meaning for the region in which they operate, as they are closely linked to its provision of a unique landscape, which in turn affects the tourism potential of the locality. Landscapes develop as a complex, and grassland management corresponds with the changes in the landscape context and the socio-economic context. Much of the species richness and biodiversity of Europe was developed and has been maintained by pasturing. After the Ice Age, the natural re-invasion of species and the human impact by pasturing developed simultaneously. By the end of the Middle Ages the small areas with arable land were enclosed and the landscape used for pasturing, as documented by contemporary painters. Regional and social identity was influenced by pastured landscapes, and in many regions of Europe this identity is still bound on landscapes formed by pasturing systems, even where they have lost their original economic basis.

Grazing systems organized on a communal or co-operative basis, either permanently or seasonally, can utilize different vegetation types and landscapes, including forests, pastures and mires, and even some arable land, and generally use larger land areas than individual family farms. The emergence of co-operative structures enabled exploitation of additional grazing resources, remote from the villages and/or with difficult access. Co-operative grazing systems enable grazing by big flocks or herds, with low costs, and their biodiversity effects vary with scale.

(1) At the landscape scale, large-scale grazing produces parkland or grove-land. The scattered northern Europe timberline and the alpine timberline are products of grazing, mixing forest patches together with tundra or sub-alpine meadows. Similar landscapes are

documented in the lowlands on historic maps and occur as rare remainders of commonages in Upper Bavaria.

(2) At the medium or ecotone level, grazing produces spatial smooth gradients, e.g. the shrub belt and tall herb belt between forests and meadows, or gradients between raised bog hummocks and fen vegetation. Large-scale grazing also produces time gradients by effecting cycles of young and older succession stages in a space- time gradient. In most landscapes these time cycles are additionally maintained by management (burning, tree harvesting).

(3) At the micro or patch level, the trampling action of grazing animals produces micro habitats and bare soils which serve as secondary habitats for species of young and dynamic ecosystems. Many plants need bare soils to germinate. The size of the grazing units depends on the density of accessible grazing resources within the region.

Table 1. Hierarchical typology of large-scale grazing systems: LACOPE project

Group 1: Year-round grazing systems

(1) Northern-most Fennoscandia: long distance migration between winter- and summer-grazing grounds along climatic gradients.

(2) Upland sheep and cattle grazing in hyper-Atlantic regions of Ireland, UK and south-west Norway, with two variants: (i) year-round grazing, managed during summer as rough grazing on blanket bogs and heath lands in a hyper-Atlantic environment, combined with improved grazing grounds near farm-steads, and (ii) large-scale grazing period in winter (inverse transhumance) in regions with high yearly average temperatures (Bunce et al., 2004).

(3) Mediterranean year-round grazing with sheep, cattle, pigs or goats, and movements driven by high temperature /drought effects on summer feed, with three variants: (i) Polygono cereal-fallow system of La Mancha; (ii) Mediterranean transhumance (Montados and Dehesas agro-silvo-pastoral systems) with short displacements or long distance migration between village-based winter grazing and summer grazing in the Mediterranean uplands; and (iii) short-distance mountain-foothill migration in Mediterranean areas.

Group 2: Seasonal grazing combined with an indoor period

This group is sub-divided into five regional types:

(4) Middle-European transhumance systems with temporal indoor feeding (classic transhumance)

(5) The indoor feeding period in central Europe caused by harsh winter weather. Livestock systems are farmstead-oriented and production of winter forage is optimized. Traditional access to grazing rights in remote areas. Examples include: sheep transhumance at the Swabian Jura, south-west Germany (Luick, 1997).

(6) Allmende system with summer grazing on marginal sites or/and on rough grazing areas in the periphery of villages and farm-steads; stationary grazing on commonages, currently co-operatively or privately managed (Lederbogen et al. 2004). The Allmende pastures form a spatial continuum with the farm-steads or are located in the vicinity so that products are processed at the permanent farm-stead.

(7) Short distance migration combined with temporal indoor feeding; ‘modern’ type of migrating cattle and sheep flocks.

(8) Mediterranean short distance migration with seasonal indoor feeding

Regions where co-operative pastoral systems have survived in Europe do not generally have a good potential for agriculture, when considered in terms of productivity or profitability on a per hectare basis, due to their climate and topography. In Table 1 a hierarchical typology of large-scale grazing systems is presented. Today, the existence of these structures is threatened because of changes in agricultural land-use practices and inappropriate governmental policies, which have resulted in intensively used, even overexploited regions on the one hand, and abandonment of marginal ones on the other. Many of the species that depend on open or semi-open landscapes are now seriously endangered. Therefore, to maintain these systems, and the habitats and species that depend upon them, the challenge lies with the difficult adjustments in terms of modern economics, of the resource exploitation to the needs of the local population. The following examples, studied in the LACOPE project (www.lacope.net) illustrate, for contrasting areas of Europe, co-operative pastoral systems linked to particular habitats, and the management practices and techniques that are needed to maintain the production system and the nature conservation value of the area.

Examples of large-scale grazing systems and their biodiversity and agricultural / socio-economic implications

Reindeer herding in northern Fennoscandia

The characteristics of this system are long-distance migration between winter- and summer-grazing grounds and the utilization of feed resources exclusively from semi-natural landscapes. The migration follows continental climate gradients and is necessitated by the need to save forage resources and to protect the lichen cover from trampling in the continental winter areas.

Traditional Sámiland, Sapmi and the whole of Fennoscandia are very heterogeneous landscapes, and migration from a forested valley to unforested mountains is similar to movement from tundra to taiga. Since the Ice Age the reindeer (Rangifer tarandus tarandus) has been the dominant grazer in Fennoscandia in the Arctic and Sub-arctic zones. The area is characterized by a sharp gradient in climate and grazing conditions, created by the high mountains. Their western slopes and summits are strongly influenced by the wet Atlantic climate. Reindeer can graze the western habitats in summer, but in winter they have no access to the vegetation, due to the thick snow cover. On the winter grazing area different conditions prevail on the leeward slopes of the mountains and on the extensive Precambrian plateau east of the mountain chain. The climate is dry with precipitation primarily as rain in summer; winters are cold and snow cover is thin. Extensive lichen heaths, providing lots of winter food for reindeer if properly used, thus characterize the area. The thin, powdery snow typical of the area makes winter grazing easy. However, the lichens are extremely brittle during dry summer days. Thus, even low numbers of reindeer can destroy the lichen cover if they are present in the area under summer conditions.

During the period 1960–1990 reindeer management experienced major technological, economical and political changes. The production system changed from a subsistence pastoralism to a motorized and market-oriented industry, moving away from near-complete dependence on animal and human muscle power. In parallel, the Sámi society came under the ‘protection’ of the modern welfare state, giving access to extended schooling, housing, health care and social security. In short, a traditional livelihood encountered modernity, which became a serious challenge to Sámi communities and resource management. Moreover, internal regulations for the promotion of agricultural settlements further limited the extent of land-use for reindeer management. The encroachments of modern society have led to a fragmentation of reindeer management land and thus a marginalization of traditional livelihoods.

These changes have relevance for biodiversity. Some of the grazing is due to lemmings and voles, so grazing would not disappear if reindeer were removed, but a comparison of both sides along a reindeer fence suffices to show that habitats change when grazed or left ungrazed by reindeer. Intense grazing contributes to maintain what Zimov et al. (1995) call the 'steppe stage' of the tundra (more palatable graminoids, less mosses, less ericoids (in Fennoscandia) and/or unpalatable tussock graminoids (in Beringia) (Olafsson, 2001; Olafsson & Oksanen, 2002). The interaction between reindeer and rodent grazing is currently being investigated. Long-tailed jaegers are used as surrogates of the threatened arctic foxes as the latter are too rare for sufficient collection of field data. Both species need higher rodent densities for successful breeding. From the data obtained so far, neither positive nor negative grazing impacts on jaegers can be detected, but all the data so far are from Finnmarksvidda where grazing impacts on rodent habitats are modest. The plant target species are rare arctic-alpine plants which grow on calcareous soils. Their distribution (and rarity) is connected to the occurrence of these soils above the timberline. Grazing-induced erosion creates more habitat for rarities by spreading the lime and nutrient-rich material of the dolomite rocks to wider areas. Furthermore, these plants rely on local disturbances for successful reproduction. The wider habitat amplitude created by grazing influences metapopulation dynamics: more suitable habitat means lower risk of local extinction plus shorter dispersal distances and higher propagule production, increasing the recolonization rate of habitat patches, from where a local population has become extinct.

Contemporary Sámi reindeer herd management is a low-intensity, low-profit industry within a robust culture based on vulnerable resources. The present destabilizing factors include direct encroachments on pasture land, disturbances of pasturing and animals by other activities (forestry, tourism, mining etc.) leading to fragmentation and increased need of technical facilities (UNEP, 2001). This development also has wide implications in terms of property rights, and long-term effects of encroachments and disturbances are undermining existing property rights. A further destabilizing factor has been the replacement of animal and human muscle power by modern transport, and the integration of the herders into the surrounding society. Herder families could hardly resist being a part of the monetary based society, and the cost problem associated with new herding technology is to some extent also a treadmill effect (Riseth, 2000).

Overgrazing is the main internal problem. This danger is greatest for the lichen pastures of winter, than for the green pastures used most of the year. Serious overgrazing may require up to 20-30 years of recovery time. Green pastures can stand much heavier pasture utilization, changes in vegetation to more productive and pasture tolerant species, e.g. from heather species to grass species, can be promoted (Olofsson, 2001). Moreover, mat-forming lichens will have their highest production level at an intermediate grazing intensity of winter pasturing. Accordingly, some rotation in winter pasture use between years is a favourable strategy, which also is a natural strategy for the animals.

The Burren in Ireland

This area has a traditional management involving a period with large-scale grazing in winter – an ‘inverse transhumance.’ In terms of biodiversity the Burren is a particular hot spot of species richness on karstic limestone hills. Winter grazing is released to the uplands where the ground is warm enough for continuous vegetation growth during winter. Livestock, mainly suckler cows and sheep, are sent out for large-scale grazing in an non-herded management system. During summer the farm land in the valley is used and the vegetation in the uplands can recover.

Negative impacts on species diversity have been documented on those parts of the uplands that have no rest from grazing during summer time, and where additional feeding with silage during winter times has lead to nutrient accumulation and local trampling effects. The balance of livestock density between winter grazing and summer grazing has changed because of higher productivity of grassland management in the valley. For generations the grazing rights in the Burren have been partly rented by “rancher” farmers from the larger surroundings of the Burren. This causes a competitive situation with respect to feed resources between external and local farmers who are more interested in winter grazing. The optimized management for the Burren relies on a good co-operation between nature conservation interests (Wildlife Committee of the Heritage Council) and both groups of farmers (Burren representatives of the Irish Farmers' Association) (Dunford, 2002).

The Polygono system of La Mancha sheep grazing in Spain

The traditional cereal-fallow system with sheep as the dominant herbivores (one-year winter cereal, one-year fallow) is evolving towards either continuous cereal cropping, or towards a more intensive ploughing regime during the fallow year. Both trends, favoured by increasing mechanization of the agricultural farming practices, are reducing the availability of grazing resources by shortening the growing period of fallow-associated vegetation. The lack of specific subsidies is diminishing the crop fields managed through a rotation of crops between cereal and forage legumes (vetch), with or without a fallow year in-between. This kind of crop rotation is relevant for the economy of the grazing system as a potential source of local fodder (Caballero, 2003). The current trend towards a reduction of grazing determines results in an increasing area of ‘eriales’ which are no longer grazed, and in particular the more distant areas from the sheepfolds are being progressively abandoned, favouring shrub encroachment and probably a loss in herb species diversity (though increased shrub species diversity). The territory of a village is divided into allotments connected with contractual grazing rights for sheep. The land (95% comprises arable fields) is owned by cereal producers and partly in public property. Short distance migration through mostly flat landscapes are typical for the region of La Mancha. A dry summer season frequently brings a bottle neck in feed supply. For this reason the sheep owners search for opportunities to fill the diet gap with additional forage from legume fields and other crops. Food resources are mainly based on agricultural residues (cereal or legume stubbles and fallow land), non-arable land such as natural grassland, shrub-steppe vegetation (eriales) and Mediterranean forest and shrubland. Parcels of olives, vineyards and irrigation are, by law, excluded from grazing use by pastoralists who rent the grazing allotments (polígonos de pastos).

In terms of biodiversity relevance one of the target species for protection in the context of large scale grazing systems in La Mancha is the Great Bustard (Otis tarda). The population of the Great Bustard in the Iberian Peninsula is linked to the cereal-farming regions with dominance (census 2001–2002) in Castile-Leon (10500 individuals), Extremadura (6000 individuals), Castile-La Mancha (4500 individuals), the Madrid region (1150 individuals distributed in 13 leks), and Portugal (some 1500 individuals). Approximately 60% of the world's population of Great Bustard is concentrated in the Iberian Peninsula (www.proyectoavutarda.org). Since hunting this bird has been prohibited in Spain since 1980, the population is recovering, although the future of this species is linked to farm practices and urban development in the Spanish cereal regions (Alonso et al., 2003).

There are a number of destabilization effects which are threatening this land-use system. Recent subsidy-driven tendencies in Spain's agriculture have induced a continuous disconnection of the two land uses which were formerly closely linked through traditions and dependency (organic fertilization). Irrigation leads to reduced periods of set aside. Subsidies for vineyards and olive production have caused an increase of enclosed land for grazing. Marginalization processes and scattered property structure in the arable fields makes it nearly impossible to turn towards a concerted strategy for stabilization of the La Mancha sheep grazing system. Forage deficits appear mainly in winter. Shepherds are landless and have no direct access to land for producing additional forage (Caballero et al., 2003).

Outlook and conclusions

The impacts of agricultural change on biodiversity and landscape during recent decades are well understood, and reforms of agri-environmental policies are now providing frameworks for incorporating biodiversity and other environmental objectives into agriculture. Many threats remain, in both the now-fragmented areas of agriculturally improved productive lowlands, and also in the marginal areas of Europe where traditional systems are disappearing or lands are abandoned. Challenges for researchers and policy makers include appraisal of the value of elements of biodiversity within the rural economy, including links with quality of farm products, the value of tourism in relation to farmland and rangeland management, impacts of biodiversity on livestock health and nutrition, and the role of biodiversity in whole ecosystem management, including soil and water conservation. The current food security provision that Europe now enjoys cannot necessarily be assured in future decades in the face of issues such as climate change, world population growth and uncertainties over supplies of fossil energy and water. Thus, many research challenges remain if we are to successfully manage agri-environmental ecosystems to deliver food and biodiversity benefits.

REFERENCES

Aerts, R.J., Barry, T.N. & McNabb, W.C. 1999. Polyphenols and agriculture: beneficial effects of proanthocyanidins in forages. *Agriculture, Ecosystems and Environment* **75**, 1–12.

Allison, K. 1995. *A guide to herbs for horses.* J.A. Allen, London, 48 pp.

Alonso, J.C., Martín, C.A., Palacín, C., Magaña, M.A. & Martín B. 2003. Distribution, size, and recent trends of the Great Bustard (Otis tarda) population in Madrid region, Spain. *Ardeola* **50**, 21–29.

Beever, D.E., Offer, N. & Gill M. 2000. The feeding value of grass and grass products. In Hopkins A. (ed.): *Grass – its production and utilization*, 3rd edition. Blackwell Science, Oxford, UK. pp. 140–195.

Bignall, W.M. & McCracken, D.I. 1996. Low-intensity farming systems in the conservation of the countryside. *Journal of Applied Ecology* **33**, 413–424.

Buller, H., Wilson, G.A. & Holl, A. 2000. *Agri-environmental policy in the European Union*. Ashgate, Aldershot, UK, 291 pp.

Bunce, R.G.H., Pérez-Soba, M., Jongman, R.H.G., Gómez Sal, A., Herzog, F. & Austad, I. (eds.) 2004. Transhumance and Biodiversity in European Mountains. Report of the EU-FP5 project TRANSHUMOUNT (EVK2-CT-2002-80017). IALE publication series nr 1, pp 321.

Caballero, R. 2003. A set of guidance for the management of grazing units in the cereal-sheep system of Castile-La Mancha (south-central Spain). *Journal of Sustainable Agriculture* **21**, 11–28.

Carson, R. 1962. *Silent spring.* (Reprinted 2000). Penguin Books, Harmondsworth, UK, 336 pp.

Casler, M.D. 2001. Breeding forage crops for increased nutritional value. *Advances in Agronomy* **71**, 51–107.

Champion, R.A., Orr, R.J., Penning, P.D., Rutter, S.M. 2004. The effect of the spatial scale of heterogeneity of two herbage species on the grazing behaviour of lactating sheep. *Applied Animal Behaviour Science* **88,** 61–76.

Charles, A.H. & Haggar, R.J. (eds.). Changes in sward composition and productivity. BGS Occasional Symposium No. 10. British Grassland Society, Hurley, UK, 253 pp.

Devon Wildlife Trust. 2005. Devon Wildlife Trust website http://www.devonwildlifetrust.org/index.php?section=people:greengateway. (Accessed April 2005).

Dodd, M.B., Barker, D. J. & Wedderburn, M.E. 2004. Plant diversity effects on herbage production and compositional changes in New Zealand hill country pastures. *Grass and Forage Science* **59**, 29–40.

Dunford, B. 2002. *Farming and the Burren.* Teagasc (The Irish Agriculture and Food Development Authority), pp 108. www. teagasc.ie., ISBN 1-84170-321-4.

EC (1985) Regulation 797/85. Official Journal of the European Communities, L93, 30/3/85.

EC (1992) Regulation 2078/92. Official Journal of the European Communities, L215, 30/7/92.

EEA (2004) High nature value farmland – characteristics, trends and policy challenges. EEA Report No 1/2004, European Environment Agency, Copenhagen.

Elliot, R H. 1946. *The Clifton Park system of farming and laying down land to grass.* Faber, London, 261 pp.

Europa (2005) NATURA 2000. http://europa.eu.int/comm/environment/nature/home.htm. Accessed April 2005.

FACT (2003) The scrub management handbook: guidance on the management of scrub on nature conservation sites. Forum for the Application of Conservation Techniques and English Nature, Wetherby, UK, and on-line at http://www.english-nature.org.uk/pubs/Handbooks/upland.asp?id=8 (Accessed April 2005).

Frame, J., Baker, R.D. & Henderson, A.R. 1995. Advances in grassland technology over the past fifty years. In Pollott G.E. (ed.): *Grassland into the 21st Century.* BGS Occasional Symposium No. 29, British Grassland Society, University of Reading, Reading, pp. 31–63.

Freibauer, A., Rounsevell, M.D.A., Smith, P. & Verhagen, J. 2004. Carbon sequestration in the agricultural soils of Europe. *Geoderma* **122,** 1–23.

Grime, J.P. 1979. *Plant strategies and vegetation processes.* Wiley, Chichester, UK, 222 pp.

Haggar, R.J. & Peel, S. (eds). Grassland Management and Nature Conservation. BGS Occasional Symposium No. 28. The British Grassland Society, University of Reading, Reading, 336 pp.

Hector, A., Schmid, B., Beierkuhnlein, C., Caldeira, M.C., Diemer, M., Dimitrako-poulos, P.G., Finn, J.A., Freitas, H., Giller, P.S., Good, J., Harris, R., Hogberg, P., Huss-Danell, K., Joshi, J., Jumpponen, A., Korner, C., Leadley, P.W., Loreau, M., Minns, A., Mulder, C.P.H., O'Donovan, G., Otway, S.J., Pereira, J.S., Prinz, A., Read, D.J., Scherer-Lorenzen, M., Schulze, E.D., Siamantziouras, A.S.D., Spehn, E.M., Terry, A.C., Troumbis, A.Y., Woodward, F.I., Yachi, S. & Lawton, J.H. 1999. Plant diversity and productivity experiments in European grasslands. *Science* **286**, 1123–1127.

Herms, D.A. & Mattson, W.J. 1992. The dilemma of plants: to grow or to defend. *The Quarterly Review of Biology* **67**, 283–335.

Hopkins, A. 2004. Productivity and nutrient composition of multi-species swards. In Hopkins A. (ed.): *Organic Farming – science and practice for profitable livestock and cropping. BGS Occasional Symposium No. 37*. The British Grassland Society, University of Reading, Reading, pp. 117–120.

Hopkins, A., Bowling, P.J. & Johnson, J. 1992. Site-specific variability in the productivity and nutrient uptake of permanent and sown swards. *Proceedings of 14th General Meeting European Grassland Federation*. June 8–11, Lahti, Finland, pp. 199–203.

Hopkins, A., Gilbey, J., Dibb, C., Bowling, P.J. & Murray P.J. 1990. Response of permanent and re-seeded grassland to fertilizer nitrogen. 1. Herbage production and herbage quality. *Grass and Forage Science* **45**, 43–55.

Illius, A. W., Gordon, I. J., Elston, D. A. & Milne, J. D. 1999. Diet selection in goats: a test of intake-rate maximization. *Ecology* **80,** 1008–1018.

IPCC (2001) Climate Change 2001: the scientific basis (J.T. Houghton et al., eds). Cambridge University Press, Cambridge, UK, 881 pp.

Isselstein, J., Jeangros, B. & Pavlu, V. 2005. Agronomic aspects of extensive grassland farming and biodiversity management (these proceedings).

Jans, F. 1982. Sommerfuetterung. Die bedeutang der botanischen zusammensetzung des gruenfutters. UFA Review, 3.

Janssens, F.A., Peeters, A., Tallowin, J.R.B., Bakker, J.P., Bekker, R.M., Fillat, F. & Oomes M.J.M. 1998. Relationship between soil chemical factors and grassland diversity. *Plant and Soil* **202**, 69–78.

Jones, M.B. & Donnelly A. 2004. Carbon sequestration in temperate grassland ecosystems and the influence of management, climate and elevated CO_2. *New Phytologist* **164**, 423–439.

Lederbogen, D., Rosenthal, G., Scholle, D., Trautner, J., Zimmermann, B. & Kaule, G. 2004. Allmendweiden in Südbayern: Naturschutz durch landwirtschaftliche Nutzung. – Schriftenreihe Angewandte Landschaftsökologie, Heft 62, Landwirtschaftsverlag, Münster. 469 S. & Anhang.

Lehmann, E. & Schneeberger. 1988. Efficient utilization of nutrients in grassland systems including permanent grassland. *Proceedings of the 12th General Meeting of the European Grassland Federation*. 4–7 July, Dublin, Ireland, pp. 59–70.

Luick, R. 1997. Extensive pasture systems in Germany – realizing the value of environmental sustainability. In: Livestock Systems in European Rural Development. First International Conference of the LSIRD Network, Nafplio, Greece. Macaulay Land Use Research Institute, Scotland, pp. 81–92.

Marrs, R.H. 1993. Soil fertility and nature conservation in Europe: theoretical considerations and practical management solutions. *Advances in Ecological Research* **24,** 241–300.

Nösberger, J., Lehmann, J., Jeangros, B., Dietl, W., Kessler, W., Bassetti, P. & Mitchley, J. 1994. Grassland production systems and nature conservation. In Mannetje L.'t and Frame J. (eds): *Grassland and Society. Proceedings of the 15th General Meeting of the European Grassland*

Federation. June 6–9, Wageningen, The Netherlands, pp. 255–265
Nuthall, R., Rutter, S.M. & Rook, A.J. 2000. Milk production by dairy cows grazing mixed swards or adjacent monocultures of grass and white clover. *Proceedings of the 6th BGS Research Meeting.* Aberdeen, 11–13 September. British Grassland Society, Reading, UK, pp. 117–118.
O'Keefe, J.H. & Cordain, L. 2004. Cardiovascular disease resulting from a diet and lifestyle at odds with our Palaeolithic genome: How to become a 21st-century hunter-gatherer. *Mayo Clinic Proceedings* **79**, 101–108.
Olofsson, J. & Oksanen, L. 2002. Role of litter decomposition for increased primary productivity in areas heavily grazed by reindeer: a litterbag experiment. *Oikos* **96**, 265–272.
Olofsson, J. 2001. Long-Term Effects of Herbivory on Tundra Ecosystems. *Doct. dissertation.* Umeå University, Sweden.
Peeters, A. & Frame, J. 2002. Quality and Promotion of Animal Products in Mountains. *Proceedings FAO/CIHEAM, Inter-Regional Cooperative Research and Development Network for Pastures and Fodder Crops.* 13–17 September 2002, Luz-Saint-Saveur, France, 147 pp.
Pretty, J. 2002. Change in agricultural policy and its consequences: will conservation keep farmers in business?. In Frame, J. (ed.): *Conservation Pays? BGS Occasional Symposium No. 36*. The British Grassland Society, University of Reading, Reading, pp. 15–25.
Riseth, J. Å. 2000. Sámi reindeer management under technological change 1960–1990: implications for common-pool resource use under various natural and institutional conditions. A comparative analysis of regional development paths in West Finnmark, North Trøndelag, and South Trøndelag/Hedmark, Norway. *Dr. Scientarium Theses* 2000:1. ISSN 0802-3222. ISBN 82-575-0411-4. Ås: Department of Economics and Social Sciences, Agricultural University of Norway.
Rutter, S.M., Cook, J.E., Young, K.L. & Champion, R.A. 2005. Spatial scale of heterogeneity affects diet choice but not intake in beef cattle. *20th International Grassland Congress*, *Glasgow Satellite Workshop*. Scotland, 3–6 July 2005. (In press).
Rutter, S.M., Orr, R. J., Yarrow, N.H. & Champion, R.A. 2004. Dietary preference of dairy cows grazing ryegrass and white clover. *Journal of Dairy Science* **87**, 1317–1324.
Rychnovska, M., Blazkova, D. & Hrabe, F. 1994. Conservation and development of floristically diverse grassland in central Europe. In Mannetje L.'t and Frame J. (eds): *Grassland and Society. Proceedings of the 15th General Meeting of the European Grassland Federation.* June 6–9, Wageningen, The Netherlands, pp. 266–277.
Soule, J. D. & Piper, J.K. 1992. *Farming in Nature's Image – an ecological approach to agriculture, 2nd edition*. Island press, Washington, USA, 305 pp.
Soussana, J.F, Loiseau, P., Vuichard, N., Ceschia, E., Balesdent, J., Chevallier, T. & Arrouays, D. 2004. Carbon cycling and sequestration opportunities in temperate grasslands. *Soil Use and Management* **20**, 219–230 Suppl.
Spehn, E.M., Scherer-Lorenzen, M., Schmid, B., Hector, A., Caldeira, M.C., Dimitrakopoulos, P.G., Finn, J.A., Jumpponen, A., O'Donnovan, G., Pereira, J.S., Schulze, E.D., Troumbis, A.Y. & Korner, C. 2002. The role of legumes as a component of biodiversity in a cross-European study of grassland biomass nitrogen. *Oikos* **98**, 205–218.
Tallowin, J.R.B., Smith, R.E.N., Goodyear, J., Bullock, J.M. & Lehain, P. 2002. Sustainable management of lowland purple moor-grass and rush pastures: constraints and opportunities for livestock farmers. In Frame J. (ed.): *Conservation Pays? BGS Occasional Symposium No. 36.* The British Grassland Society, University of Reading, Reading, pp. 47–50.
Tilman, D., Reich, P.B., Knops, J., Wedin, D., Mielke, T. & Lehman, C. 2001. Diversity and productivity in a long-term grassland experiment. *Science (Washington)* **294**, 843–845.
Tilman, G.D. & Downing, J.A. 1994. Biodiversity and stability in grasslands. *Nature* **367**, 363–

365.
UNEP (2001). GLOBIO. Global Methodology for Mapping Human Impacts on the Biosphere (C. Nellemann et al.) The Arctic 2050 Scenario and Global Application. UNEP/DEWA/TR.01-3. ISBN:92-807-2051-1.
Vickery, J.A., Tallowin, J.R., Feber, R.E., Asteraki, E.J., Atkinson, P.W., Fuller, R.J. & Brown, V.K. 200.1 The management of lowland neutral grasslands in Britain: effects of agricultural practices on birds and their food resources. *Journal of Applied Ecology* 38, 647–664.
Waghorn, G.C., Tavendale, M.H. & Woodfield, D.R. 2002. Methanogenesis from forages fed to sheep. *Proceedings of the New Zealand Grassland Association* **64**, 167–171.
Wood, J.D., Richardson, R.I., Nute, G.R., Fisher, A.V., Campo, M.M., Kasapidou, E., Sheard, P.R. & Enser, M. 2004. Effects of fatty acids on meat quality: a review. *Meat Science* **66**, 21–32.
Worrall, P., Pederdy, K.J. & Millett, M.C. 1997. Constructed wetlands and nature conservation. *Water Science and Technology* **3**, 205–213.
Zimov, S.A., Chuprynin, V.I., Oreshko, A.P. , Chapin, F. S., Reynolds, J.F. & Chapin, M.C. 1995. Steppe-tundra transition: a herbivore-driven biome shift at the end of the Pleistocene. *American Naturalist* **146**, 765–794.