# Grazing and biodiversity: from selective foraging to wildlife habitats

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# Grazing and biodiversity: from selective foraging to wildlife habitats

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**Abstract.** Livestock grazing in low-intensity farming systems is a key aspect in the conservation of Europe's biodiversity, which reaches high levels of species richness in semi-natural grasslands. With the demise of traditional grazing systems, the design of viable low-intensity grazing systems for the future requires a good understanding of grazing impacts on biodiversity. Here, I review various scale-dependent aspects of selective grazing and how they may affect biodiversity. Insects such as butterflies are well-suited to elucidate small-scale impacts of grazing intensity. They highlight the importance of viewing grazing impacts in a framework of spatial heterogeneity and successional dynamics. In order to optimise these successional dynamics, grazing management may adopt techniques such as rotational grazing and strategic placement of mineral licks. However, we still lack a good evidence base on the effects of targeted grazing practices on biodiversity. The challenge to solve this gap can be met by a combination of creative field experiments that focus on the mechanisms of biodiversity responses and adaptive management that builds on a continuous feedback from sound monitoring.

**Keywords.** Grazing impact – Low intensity farming – Conservation – Butterflies – Rotational grazing.

***Pâturage et biodiversité : du broutage sélectif à l'habitat de la faune sauvage***

***Résumé.*** *Le pâturage du bétail dans les systèmes d'élevage à faible intensité est un aspect déterminant pour la conservation de la biodiversité en Europe, qui atteint de forts niveaux de richesse en espèces dans les prairies semi-naturelles. Avec la disparition des systèmes de pâturage traditionnels, la conception de systèmes de pâturage à faible intensité viables pour l'avenir requiert une bonne compréhension des impacts du pâturage sur la biodiversité. J'examine ici plusieurs aspects de pâturage sélectif dépendants de l'échelle ainsi que la manière selon laquelle ils peuvent affecter la biodiversité. Des insectes tels que les papillons sont appropriés pour élucider les impacts à petite échelle de l'intensité de pâturage. Ils mettent en lumière l'importance d'envisager les impacts du pâturage dans un cadre d'hétérogénéité spatiale et de dynamique de succession. En vue d'optimiser ces dynamiques de succession, la gestion du pâturage peut faire appel à des techniques telles que le pâturage rotationnel et l'emplacement stratégique de pierres à lécher. Toutefois, il nous manque encore une bonne base de preuves quant aux effets des pratiques de pâturage ciblé sur la biodiversité. Le défi lié à cette lacune peut être résolu par une combinaison d'expérimentations créatives aux champs focalisées sur les mécanismes de réponse de la biodiversité et de gestion adaptative, visant à tirer parti d'un feedback continu à partir d'un suivi robuste.*

***Mots-clés.*** *Impact du pâturage – Élevage à faible intensité – Conservation – Papillons – Pâturage rotationnel.*

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## I – Introduction

Low-intensity land use has played a major role in shaping Europe's biodiversity (Poschlod, 2015). A high proportion of plant and animals species are linked to the open landscapes that have developed over several millennia of the traditional land use systems preceding the era of industrial agriculture that relies on the inputs of chemical fertilisers (Bignal and McCracken, 1996). Grazing by various types of ungulate livestock has been an important driver determining the structure and composition of plant communities and the associated animal diversity that characterise these semi-natural communities. It has even been argued that these landscapes and characteristic species closely resemble the natural herbivore-dominated communities from which they have been derived (Vera 2000; Bakker *et al.*, 2015).

At present, the conservation value of the species-rich communities from traditional land use systems has been recognised under the umbrella of High Nature Value (HNV) farming systems (Oppermann *et al.*, 2012). With the continuing pressure to increase agricultural productivity, HNV farming still faces a growing threat of marginalisation, leading to abandonment and, ultimately, loss of biodiversity (Balmer and Ehrhardt, 2000). Stimulating rewilding with the restoration of wild native herbivores has been suggested as an option to maintain the biodiversity of HNV farming systems (Merckx and Pereira, 2015), but with conflicting claims of land owners and challenging socio-economic problems for the affected rural communities, such initiatives have not yet proved successful on a large scale. With increasing concerns for the planetary boundaries of the earth system for human influence (Steffen *et al.*, 2015), a renewed emphasis on the beneficial ecosystems services and robust sustainability of low-intensity farming systems seems called for (Maes *et al.*, 2012; Rodríguez-Ortega *et al.*, 2014). In developing these efforts, it will be important to include biodiversity to the full extent, as the ecosystem service approach tends to focus on a minority of species that perform the core of ecosystem services, such as pollination, which fails to meet the requirements of rare species (see Kleijn *et al.*, 2011).

Integrating biodiversity aspects into the optimisation of grazing systems requires a good understanding of the impact of different land use types and intensities across species communities. Although significant steps have been made to deepen our understanding of biodiversity dynamics in agro-ecosystems (e.g., Tscharntke *et al.*, 2012), this remains a challenge for low-intensity grazing systems, given their typically large spatial extent and high landscape complexity (WallisDeVries *et al.*, 1998; Plachter and Hampicke, 2010). In this paper, I will review some main aspects of grazing impact on biodiversity. I will especially focus on the role of spatial scale as both the grazing process and functional aspects of habitat quality are strongly scale-dependent (Fig. 1; see WallisDeVries, 2002). Also, in addressing conflicts between grazing impact and biodiversity, it is imperative to view the various management options at the proper spatial scale. In dealing with biodiversity, I will rely especially on examples from butterflies and other insects, as arthropods are well-suited to illustrate the impacts of grazing across a range of spatial scales (Van Klink *et al.*, 2015a). The case studies that are treated here derive not only from montane but also from lowland ecosystems, such as coastal salt marshes and heathlands, but as these also pertain to low-intensity grazing systems, the emerging insights should be considered equally applicable to montane and alpine ecosystems.

## II – Spatial scales in grazing behaviour

The environment of free-ranging livestock can be viewed as mosaic of units at different spatial scales, ranging from the landscape level of a herd's home range (order of magnitude 1-100 km$^2$) down to the dm$^2$ scale of individual bites. Evidently larger spatial scales should also be considered with respect to seasonal ranges, such as in mountainous environments, and viable populations or even species persistence of wild herbivores. The grazing process operates across this range of spatial scales (Fig. 1; Bailey *et al.*, 1998; Rook *et al.*, 2004).

Local depletion and foraging selectivity are the main motivations for short-term movements within a feeding bout between bite locations (head movements), between feeding stations (a single step) and between patches (series of steps) within a feeding site (WallisDeVries *et al.*, 1999), where animals graze for a longer period of several hours. At larger scales of time and space, decisions are made on the selection of plant communities, landscape types and seasonal ranges. These involve trade-offs between the energy gains of staying against the temporary costs of travel to other feeding sites with more abundant resources. Here, other currencies besides energy and protein, such as water or minerals (particularly phosphorus and sodium; McNaughton, 1990; WallisDeVries and Schippers, 1994) may play a role in the selection process. Selection can occur at each scale level and may accumulate across scale levels.

Selectivity is also affected by social interactions between herd members leading to intraspecific competition, which increases with stocking rate (Lawrence and Wood-Gush, 1988) and is affected by social status (Hewitson *et al.*, 2007). In complex environments, the experience of herd members may also result in foraging decisions for the entire herd at higher spatial scales (Prins, 1996).

Foraging selectivity may differ as a result of variation in body size, morphology and digestive physiology between species (Hofmann, 1989; Cromsigt *et al.*, 2009) and breeds (Rook *et al.*, 2004). Variation in body composition and, hence, basal metabolic rate between species (Richmond *et al.*, 1977; Christopherson *et al.*, 1978; 1979) and breeds (Wright and Russel, 1984; Webster, 1985) may also lead to differences in foraging selectivity, as voluntary intake and growth rate are positively related to energy requirements (Ketelaars and Tolkamp, 1992). Thus, the higher energy requirements of dairy breeds compared to beef breeds (Thompson *et al.*, 1983; Solis *et al.*, 1988) and of early maturing than late maturing breeds (Mason, 1971) is likely to lead to a greater selectivity for patches with high intake rates of digestible dry matter, although well documented examples are rare (see Rook *et al.*, 2004). Thus, in a comparative study of two Aberdeen Angus genotypes, Cid *et al.* (1997) found that, at the end of the growing season, sward height structure was more heterogeneous in paddocks grazed by the more selective early maturing genotype, at a similar overall mean herbage mass. This greater patchiness may have subsequent impacts on biodiversity.

## III – Spatial scales in wildlife habitats

Arthropods have been found to be generally more vulnerable to grazing intensity than plants (Van Klink *et al.*, 2015a), which warrants a focus on arthropods as biodiversity indicators in grazing systems. Amongst arthropods, butterflies are appropriate organisms to illustrate spatial scales of wildlife habitats from bite level to home range level (Fig. 1). Relevant spatial scales for plants are typically smaller and those for birds are mostly much larger (WallisDeVries, 2002). Moreover, the ecological relations of butterfly species in Europe have been comparatively well studied in comparison to other insect groups (Thomas, 2005). In order to understand habitat requirements for butterflies, as well as for other species, a resource-based perspective (Dennis *et al.*, 2003) has proved fruitful. Here, the habitat is defined as the full set of essential resources and conditions required by an organism to complete its life cycle. Thus, butterflies will need to find, amongst others, nectar plants, shelter and roosting sites as adults, food plants in a suitable microclimate for oviposition and larval development. As cold-blooded animals, they are sensitive to microclimatic conditions, especially during the larval stages. In temperate regions, this means warm and sheltered conditions at the microscale of decimetres in early stages to metres for late instars. This restricts many species to low-productive environments with low dead:green ratios of plant biomass in spring (WallisDeVries and Van Swaay, 2006). The distribution of all essential resources and conditions may show a complete spatial overlap in a single patch at a scale of <1 ha, but they may also be spatially disjunct. In that case, the daily mobility of the butterfly (x100 m in many species) will determine whether the distance between separate resources may be bridged to obtain a functionally adequate habitat (Vanreusel *et al.*, 2007).

The dispersal capacity of butterflies varies greatly between species. However, in many species that are considered habitat specialists in the modern anthropogenic landscape (WallisDeVries, 2014), dispersal is limited to a few kilometres. As the density of individuals is often low and yearly fluctuations in population size are large due to climatic variation and parasitoids, local populations can maintain themselves only when the size of a habitat patch is in the order of hectares (Schtickzelle and Baguette, 2009). In the long run, this is insufficient for a viable population because of the extinction risk due to stochastic events. A network of patches is then required, all the more so because in the modern fragmented landscape individual patches are often much smaller than one hectare. Many butterflies thus typically require a metapopulation structure at a landscape scale for long-term population persistence Schtickzelle and Baguette, 2009).

In the light of the overwhelming diversity of arthropod life histories, it is essential to distinguish species according to their life history traits (e.g. WallisDeVries, 2014) in order to understand contrasting responses between (groups of) species (see Van Klink *et al.,* 2013; WallisDeVries *et al.,* 2016).

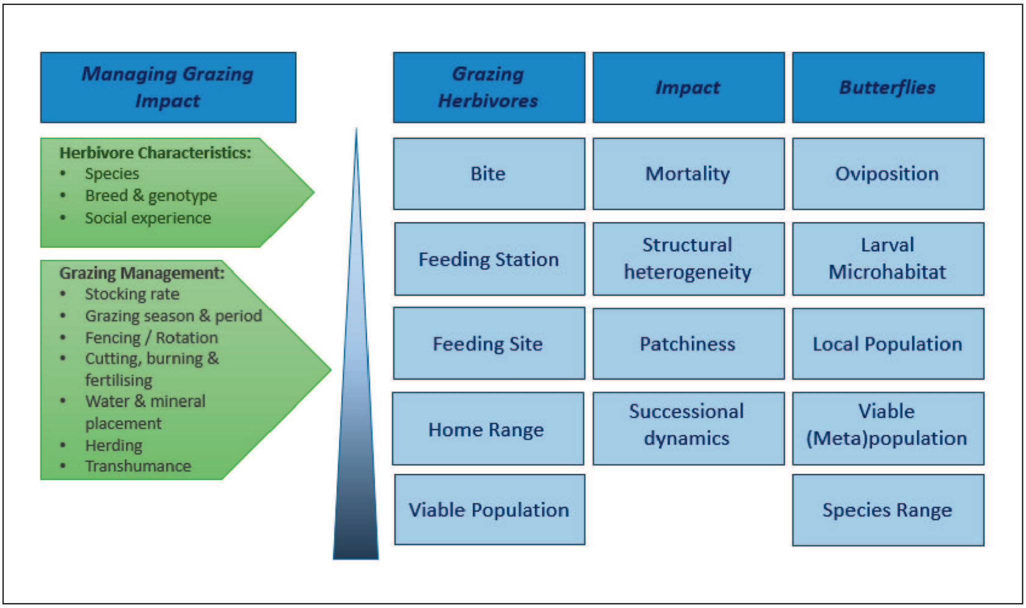

**Fig. 1. Options to manage grazing impact in relation to spatial scales of grazing and its impact on butterflies and their habitat.**

## IV – Spatial scales of grazing impact

Four main types of impact of grazing ungulates on butterflies and other arthropods can be identified (Van Klink *et al.*, 2015a): (i) disturbance and unintentional predation, (ii) reduction of plant resource availability by defoliation or trampling, (iii) increase in resource availability for dung-dependent insects and (iv) changes in habitat quality through alterations of plant diversity, vegetation structure and abiotic conditions. The first two impacts are detrimental, but the third is beneficial and the fourth may be either detrimental or beneficial. Beneficial impacts may be expected when grazing (a) increases resource availability by suppressing competitors of hostplants or by enhancing plant regrowth for herbivore species and by increasing dung availability for coprophagous insects or (b) improves microclimatic conditions by affecting vegetation structure. These impacts vary with spatial scale (Fig. 1).

The level of the bite is the basic unit determining not only the impact on vegetation structure but also the direct impact on the least mobile stages in the butterfly life cycle. Van Noordwijk *et al.* (2012) provide rare evidence of mortality in overwintering *Melitaea cinxia* caterpillars due to grazing events. Although grazing during periods of insect activity is likely to cause less severe mortality, evidence on substantial mortality due to cutting (Humbert *et al.*, 2009) suggests that this impact should not be underestimated. Furthermore, travel between feeding stations by grazing animals may cause disturbance and also increase dispersal activity in smaller animals, as has been documented in grasshoppers (Berggren, 2004). Trampling can not only directly kill arthropods, but also affect habitat conditions indirectly through soil compaction and the creation of bare ground (Van Klink *et al.*, 2015b). Whereas dung represents a crucial resource delivered by grazing livestock to dung beetles and other coprophagous insects, these insects may also suffer mortality through trampling at high stocking rates (see Van Klink *et al.*, 2015a).

Selective foraging has been supposed to maximise arthropod diversity through the creation of a patchy vegetation mosaic (Dumont *et al.*, 2012). Indeed Cherrill and Brown (1992) have shown that the grasshopper *Decticus verrucivorus* requires a combination of short and tall vegetation to find food, shelter and optimal thermoregulation. Grasshopper species richness has also been shown to increase with greater patchiness under cattle grazing (Jerrentrup *et al.*, 2014). Structural heterogeneity also appears a prerequisite for butterfly species such as *Melitaea cinxia* (WallisDeVries, 2006) and *Maculinea alcon* (WallisDeVries, 2004). However, this may apply especially to the larger and more mobile arthropods. For small and sedentary species, grazed mosaics may rather function as a patchwork of suitable and hostile habitats, following the principles of island biogeography (Cole *et al.*, 2010; Van Klink *et al.*, 2013). For such small species, horizontal patchiness can also result in large edge effects that may compromise the vertical architectural complexity which strongly determines the variation in arthropod niches (Tscharntke and Greiler, 1995). Indeed, Van Klink *et al.* (2013) did find a higher overall species richness of arthropods in tall, ungrazed salt-marsh vegetation than in homogeneous short vegetation grazed by sheep, but species richness was similar in tall vegetation and in patchy mosaics. Still, in heathland insects it has been shown by WallisDeVries *et al.* (2016) that an array of species depends on intensively grazed vegetation in dry heathland, particularly thermophilous species and species depending on prostrate plants with low competitive ability. In contrast, species requiring more humid microclimates and those associated with large tussock grasses or grazing-intolerant herbs were found mostly in lightly grazed wet heathland. Such species-specific responses to grazing intensity may be placed in a perspective of vegetation succession, with contrasts between species from early and late successional stages.

Another effect of selective grazing is an alteration of plant species composition in the vegetation. Grazing may both increase plant species diversity by preventing the encroachment of dominant grasses and shrubs as well as reduce it by overgrazing of palatable species (Olff and Ritchie, 1998). Here, differences between livestock species may affect the outcome. For example, the great ability of sheep forage selectively on herb species may reduce overall species richness (Scohier and Dumont, 2012) and flower abundance (Scohier *et al.*, 2013), with negative effects on various insect groups. Thus, sheep grazing has caused losses of populations of the rare butterfly *Euphydryas aurinia* in the United Kingdom and Ireland, due to overgrazing of its host plant *Succisa pratensis* (see WallisDeVries, 2002).

The less selective grazing behaviour of cattle rather tends to benefit plant species richness by suppressing dominant grasses (see Rook *et al.*, 2004). Horses tend to select for grasses and because of their ability to bite down close to the ground, they may create greater contrasts in patchiness than cattle (Nolte *et al.*, 2014). The higher daily activity of horses than cattle generates considerably greater trampling effects on the vegetation, with potentially negative effects on tall flowering plants and associated insects, but with particularly detrimental effects on nesting birds (Mandema *et al.*, 2013).

At larger spatial as well as temporal scales, grazing may lead to successional mosaics of grassland, scrub and woodland, with unpalatable or thorny shrubs acting as grazing refuges for tree recruitment (Olff *et al.*, 1999). In such woodland pastures, there is not only an impressive floristic diversity (WallisDeVries *et al.*, 1998) but also a high faunistic diversity, e.g. in butterflies (Bailey *et al.*, 1998). Overgrazing then may lead to a virtual disappearance of the structural heterogeneity of shrub and woodland edges and, hence, an impoverished arthropod diversity (Van Klink *et al.*, 2016).

## V – Targeting grazing impact

Targeting grazing impact in order to optimise plant and animal diversity against the socio-economic requirements of maintaining viable livestock grazing systems is a complex puzzle. Fortunately, there are many options to manipulate grazing intensity and grazing patterns. This especially involves reducing grazing intensity of overgrazed or preferred sites on the one hand, and increasing the attractiveness and use of underutilised or avoided sites on the other hand (Bailey *et al.*, 1998).

A first series of measures to modify grazing impact consists of selecting appropriate species, breeds and, especially in complex environments, individuals with social and local experience (see section II; Rook *et al.*, 2004). Although traditional breeds do not offer biodiversity benefits *per se* (WallisDeVries *et al.*, 2007), their often lower metabolic requirements may render them more prone to graze less digestible tall and tussock grasses or exploit less accessible terrain. Besides the choice of grazing animal, combinations of livestock species may also influence resource exploitation. Thus, goats can be more successful in controlling shrub encroachment than sheep (Osoro *et al.*, 2013). Mixed grazing by sheep and cattle also may lead to an enhanced arthropod abundance and, hence, greater breeding abundance of meadow pipits *Anthus pratensis* than sheep grazing alone (Dennis *et al.*, 2008). The benefits of mixed grazing on insect diversity appear to be greater at low plant diversity than at high plant diversity, especially through its impact on structural heterogeneity (Zhu *et al.*, 2012).

Manipulating stocking rate is an obvious and effective measure to modify grazing intensity. In production-oriented grazing systems, extensification may provide rapid biodiversity benefits to insect communities (WallisDeVries *et al.*, 2007) that may extend over longer time periods, although benefits may be lost with further extensification (Jerrentrup *et al.*, 2014), due to the encroachment of competitive plants.

Differentiation in grazing intensity at small to moderate spatial scales (size order 0.1-1 ha) may be achieved by rotational grazing. This may result in increased flower abundance and higher densities and species richness of butterflies and other pollinators (Farruggia *et al.*, 2012; Scohier and Dumont, 2013). However, there is an, as yet uninvestigated, risk that subsequent grazing may turn the temporarily ungrazed refuges to ‘ecological traps’ when these are not only used for foraging but also for reproduction. Van Klink *et al.*, (2016) emphasize the value of edges between grazed and ungrazed areas for arthropod diversity; at high grazing intensity, creating such grazing refuges by fencing may be an easy alternative to a reduction in stocking rate. When space is less limiting, rotation with extended grazing intervals over a greater number of paddocks may be considered. Morris *et al.* (2005) reported on a case study where such a complex rotation system was successfully applied on chalk grassland. In any case, such rotational grazing systems do show promise to reconcile objectives for livestock production and biodiversity to a considerable extent (Farruggia *et al.*, 2012).

At larger spatial scales, selective foraging in heterogeneous environments may result in overgrazing of vulnerable patches and undergrazing of areas where grazing is desired, e.g. to suppress dominant grasses or shrubs. Probo *et al.* (2014) investigated a rotational grazing in alpine mountains, with a rotation over 6 paddocks of c. 100 ha each that were grazed by cattle in 3-week periods. In the rotational system, cattle were indeed showing a less aggregated grazing pattern and exploited steeper slopes to a greater extent than in a continuous grazing system. Another option for grazing less attractive vegetation is by enclosing livestock in temporary night camps or by placement of mineral supplements. In such areas, pressure grazing on dominant dwarfshrub vegetation resulted not only in a greater cover of bare ground, but also a higher cover and pastoral value of the herbaceous vegetation as well as an increased dung beetle diversity (Tocco *et al.*, 2013). Probo *et al.* (2013) also experimented with the placement of mineral supplements and again found that the increased use by cattle reduced shrub cover and increased forage pastoral value in shrub-encroached alpine pastures. Manipulating access to water may be used in a similar way as mineral supplement placement to manipulate livestock distribution (Ganskopp, 2001).

Further options to increase the attractiveness of grazing areas are by prescribed burning, cutting and fertilising (Bailey *et al.*, 1998). However, with all these options, negative effects on biodiversity should be taken into consideration. Thus, burning may be detrimental to reptiles (Lyet *et al.*, 2009) and insects (Swengel, 2001), although superficial burning in late winter may prove beneficial (e.g. Panzer, 2002). Small-scale application of such measures can therefore be recommended for all these measures.

Grazing patterns over truly large spatial scales, exceeding the landscape scale (>10-100 km$^2$), are currently rare in European livestock grazing systems. Transhumance systems, as described by Ruiz and Ruiz (1986) in Spain or by Poschlod and WallisDeVries (2002) in Germany are no longer functioning, although they are thought to have played a major role in the dispersal of plant seeds and they are likely to have contributed to further aspects of landscape heterogeneity and biological diversity. Thus, Dolek and Geyer (2002) describe the challenges and the potential of a shepherded grazing system with sheep (and goats) as a practical low-intensity land use type to maintain biodiversity in calcareous grasslands of Bavaria. With ongoing abandonment of marginal lands for high-intensity agriculture, there may be new potential to reinstate such large-scale grazing systems in a modern of sustainable land use (Cubbage *at al.,* 2012).

## VI – Conclusion

Over recent decades, considerable progress has been made to elucidate the impacts of low-intensity grazing on biodiversity. Taking into account spatial-dependent processes in grazing systems is a crucial step in this development. With a growing emphasis to understand the mechanisms behind the emerging patterns, we are getting better equipped to optimise grazing management in relation to the conservation, and even restoration, of biological diversity. Trait-based approaches have become an important tool of the equipment.

However, in the face of the variation and complexity of low-intensity grazing systems, the evidence base on the effects of targeted grazing practices on biodiversity is still weak. Also, many studies that rely on counts of the mobile stages of adult arthropods cannot reliably distinguish between short-term concentration effects and long-term population responses (Scheper, 2015). The challenge to solve this knowledge gap can be met by a combination of (a) research involving creative field experiments that focus on the mechanisms of biodiversity responses and (b) practical implementation of adaptive management that builds on a continuous feedback from sound monitoring. Finally, we still face the major challenge to adequately incorporate the biodiversity benefits of low-intensity grazing systems in the larger framework of sustainable land use (Maes *et al.,* 2012; Siepel *et al.,* 2013; Fraser *et al.,* 2014).

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