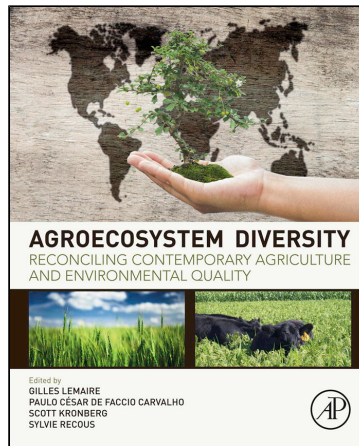


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Local and Landscape Scale Effects of Heterogeneity in Shaping Bird Communities and Population Dynamics: Crop-Grassland Interactions

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INTRODUCTION

Farmland landscapes support very high biodiversity (Pimentel et al., 1992), including functional species that provide ecosystem services (Tscharntke et al., 2005) and flagship species for wider ecosystems. However, over the past 50 years, biodiversity has strongly declined in agricultural areas, with major losses in plants, amphibians, reptiles, arthropods, mammals, and birds, and these losses have been attributed to agricultural intensification (Robinson and Sutherland, 2002; Inger et al., 2015). Agriculture intensification refers to the combination of rapid land use changes with, e.g., the replacement of natural habitats by crops, and more intensive use of existing farmland (Krebs et al., 1999;

Robinson and Sutherland, 2002; Stoate et al., 2001). About 50% of all European bird species live in rural landscapes (Tucker, 1997), but farmland birds have declined in Europe much faster (–57% on the European Union farmland bird indicator between 1980 and 2013, EBCC, 2017) than in other ecosystems, i.e., birds were more or less stable in forests during the same period (EBCC, 2017). Farmland bird specialist species, even extremely common species such as the skylark *Alauda arvensis*, have declined by more than 50% in the past 30 years (Gregory et al., 2005, Voříšek et al., 2010). Although the main service expected in farmland is obviously food production, species and habitat conservation in agroecosystems are important issues, since farmland species provide other ecosystem

services and are often the elements of biodiversity that are most accessible to humans close to urban areas (Power, 2010). Biodiversity loss has therefore additional consequences for ecosystem function and, ultimately, societal repercussions.

AGRICULTURAL INTENSIFICATION, HABITAT HETEROGENEITY, AND BIODIVERSITY

Recent empirical evidence strongly supports the hypothesis that plant and animal diversity decreases with increasing crop yield, a good proxy of agricultural intensification (see Geiger et al., 2010 and references therein), although acting through a range of specific ecologic processes that may not all be known. Processes proposed to date include loss of seminatural habitats such as field margins and hedgerows, intensification of in-field management (increased use and efficacy of pesticides and fertilizers), loss of fallow habitats (often crop stubbles) in winter due to increased winter sowing, and loss of crop diversity itself: only 30 species now provide more than 95% of the total human food consumption, and 75% of the genetic diversity of cultivated plants is already extinct (Rahmann, 2011). Those three processes all contributed to habitat homogenization (Tscharntke et al., 2005) through loss of habitat heterogeneity, at different spatial scales (i.e., countries, regions, farms, fields, and within fields; Benton et al., 2003). Mixed (arable and pastoral) farmland landscapes were replaced by homogeneous areas of either crop production or grassland, together with an increase in mean field size and the temporal simplification of arable crop rotations. Though, it is currently uncertain whether biodiversity loss at the regional scale results more from cropping intensification at the field

scale, from loss of diversity within cropping system, or from the loss of natural elements in the landscape.

In their review of biodiversity in farmland, Benton et al. (2003) argued that habitat heterogeneity in farmlands is associated with higher biodiversity, and that the recent losses of farmland biodiversity are therefore due to homogenization of farmland at multiple spatial and temporal scales. Habitat heterogeneity is supposed to affect biodiversity at the level of communities (species richness and composition), but also at the level of populations (growth rate, dispersal, and connectivity). Landscape heterogeneity has been shown to increase species richness in butterflies (Weibull et al., 2000), birds (Berg et al., 2015), and plants (Belfrage et al., 2015). Landscape heterogeneity could influence species richness through either compositional or configurational heterogeneity (Fahrig et al., 2011). The former refers to the proportion of each cover type in a landscape, while the latter considers the spatial arrangement of these cover types and their neighborhood relationships. Heterogeneity can have a negative effect on individual species due to the decrease of the area of preferred habitat type (Benton et al., 2003; Hiron et al., 2015). Pickett and Siriwardena (2011) also showed that ground-nesting birds prefer homogeneous landscapes. But our ability to detect such effect is limited, first because there will always remain a hidden heterogeneity that cannot be directly detected (Vasseur et al., 2012), secondly because results will strongly depend on landscape characterization and spatial grain (see Fahrig, 2011). In general, heterogeneity effects depend on the species considered (Teillard et al., 2014), with species-level responses not necessarily reflecting those of communities.

More complex mechanisms may be further involved, such as resource complementation (Dunning et al., 1992), niche diversity, or species

interactions (Danielson, 1991). More diverse communities are supposed to be more stable since ecologic functions are supported by more potential species and may also support more complex relationships involving edge effects and patch size. Relationships between biodiversity and heterogeneity may also be nonlinear: Concepcion et al. (2008) proposed, for instance, a sigmoidal response of biodiversity to landscape heterogeneity, with a threshold below which biodiversity does not increase and saturation at high level of landscape heterogeneity. Allouche et al. (2012) extended this view with a quadratic response, positive until an intermediate level of heterogeneity and then negative effect above this level, due to stochastic extinction of individual species when patch size becomes too small. Landscape heterogeneity also has important consequences for individual behavior and population dynamics, in particular through habitat selection, a process that links individual behavior and its demographic consequences. Habitat selection is a hierarchic behavioral process that involves multiple spatial but also temporal scales (Jones, 2001). In particular, when species require more than one resource to complete their life cycle sequentially or simultaneously (e.g., for nesting and foraging), their abundance is predicted to be higher when those resources are both present within the home range, a process called landscape complementation (Dunning et al., 1992). For example, winter cereals are optimal for the nesting of skylarks only early in the breeding season (Donald, 2004; Eggers et al., 2011), but they then shift to other crops once cereals are too tall and dense to allow nesting (Miguet et al., 2013) for other breeding attempts, a necessary condition for maintaining a stable population in intensive farmland habitat (Wilson et al., 1997; Siriwardena et al., 2001).

Quantifying landscape heterogeneity effects is, however, a difficult task in farmland landscapes, since it results from the interaction of green infrastructure (i.e., grasslands, seminatural elements) abundance and distribution, with crop configuration and composition (Fahrig et al., 2011). The resulting effect ultimately depends on the scale at which species respond to the landscape. Relationships between habitat heterogeneity and biodiversity may further depend on the level of agricultural intensification, yet very few empirical tests of this have been performed so far. Batary et al. (2011) found that in simple landscapes, less intensive agricultural practices improved biodiversity, though other studies found a higher effect of such practices on biodiversity in more complex landscapes (Duelli and Obrist, 2003), or no interaction at all (Winqvist et al., 2011). A major limitation of almost all studies to date is that they have considered landscape heterogeneity as the percentage of seminatural habitats (or seminatural elements) present (e.g., Chiron et al., 2010 on birds), or with arable fields being considered a single land use (Pickett and Siriwardena, 2011). Sometimes, the quantity of arable land or grasslands in the landscape has been used as a proxy of habitat-type diversity (Roschewitz et al., 2005; Rundlöf and Smith, 2006). Therefore, most studies so far have ignored an important source of landscape complexity, namely crop heterogeneity. Indeed, annual crop rotations, as well as the temporal variability in crop growth rates, result in heterogeneity in time and space that contributes to farmland landscape heterogeneity in addition to seminatural habitats. A few studies have shown an effect of crop diversity having a landscape complementation effect on biodiversity (Miguet et al., 2013; Kragten, 2011). An effect of configurational heterogeneity has also been

shown (Fahrig et al., 2015), but effects of crop heterogeneity on wildlife populations have received relatively little attention so far, and the few studies that investigated crop heterogeneity did not attempt to identify the processes responsible for its effects (e.g., Siriwardena et al., 2012).

THE COMPONENTS OF FARMLAND HABITAT HETEROGENEITY AND THEIR EFFECT ON BIRDS

A commonly tested prediction is that complex landscapes can support more species due to a spatial (or temporal) landscape complementation effect (Fahrig et al., 2011). Landscape complementation may occur in particular between annual arable crops and permanent or temporary grasslands. At a landscape level, cereal-based agroecosystems are characterized by high spatial discontinuity and high temporal turnover (due to crop rotations, harvesting, mowing, and plowing). Plowing, introducing temporally asynchronous alterations of habitat quality, is a main human disturbance in these systems (see Gaba et al., 2014), barely considered so far, conversely to other components of heterogeneity. Though, in landscapes dominated by intensive cereal cropping, multiannual forage crops such as meadows, mown grasslands, and alfalfa can be considered “perennial habitats” because they are usually retained for 3–4 years (or even longer). They differ radically from all annual crops in terms of mechanical disturbance (soil tillage, sowing, cutting for harvest), and they also generally receive fewer pesticide inputs. Highly fragmented in space and time, intensive farmland landscapes interspaced with grasslands provide animal and plant populations with a spatial patchwork of habitats. Many studies have found, unsurprisingly, that

grasslands were indeed important reservoirs of biodiversity within agricultural landscapes (Henckel et al., 2015; Bretagnolle and Gaba, 2015). Such undisturbed or at least less disturbed habitats may act as refuges or breeding habitats for many taxa, in particular prey of birds (insects, small mammals, or plants). From the refuge habitats, the prey population can then invade annual crops. Such source sink or metapopulation dynamics have been repeatedly suggested to maintain biodiversity in fragmented and unpredictable agroecosystems (Mouquet et al., 2006).

Indeed, grassland presence within intensive cereal landscapes is of utmost importance, not only for biodiversity perspective but also for many other agronomic and environmental services (see Lemaire et al., 2014). Due to the modern separation of crop and livestock production within agricultural businesses, current crop rotations are often only composed of annual crops. The inclusion of temporary grasslands (perennial forage crops) into crop rotations may have particularly strong impacts, especially in weed communities (Meiss et al., 2010; Gaba et al., 2010), just as areas of arable land use within pastoral systems can have positive effects on many species (Robinson et al., 2001). Weed persistence and abundance in farmland landscapes represent a critical keystone element for higher trophic levels (Bretagnolle and Gaba, 2015), including birds that feed mainly on weed seeds in winter (Moorcroft et al., 2002). This is also the case for grasshoppers (Orthoptera: Acrididae), which play a major trophic role, being the primary invertebrate herbivores in grassland habitats, and also because they are prey for other invertebrates, e.g., spiders, and vertebrates, notably, farmland birds that use them as food for chicks (Barker, 2004; Bretagnolle et al., 2011). Like many invertebrates, grasshoppers need perennial habitats

due to the fact that their survival from one year to the other is achieved by eggs that are deposited in the soil. Maintaining grasshopper populations in the landscape can be achieved only through perennial habitats such as grasslands or field boundaries incorporating suitable grassy habitats. In addition, patches of perennial crops, as well as field margins and other noncrop habitats, may act as shelters for wildlife within the “matrix” of less favorable annual crops. With respect to biodiversity and trophic networks, the presence, abundance, and distribution of these perennial habitats may have strong impacts on metapopulation and meta-community dynamics of various organisms (Hanski, 1999).

To summarize, testing the effect of cropland heterogeneity first needs the amount of seminatural habitat to be accounted for (Concepción et al., 2012), given its overwhelming effect. Fahrig et al. (2015) found a general positive effect of complex landscape configuration, independent of the effect of seminatural components. However, the effect of crop configuration for birds is particularly difficult to separate from the effect of seminatural habitat composition, especially hedges, because variations in the arrangement of field types are often confounded with lengths of such permanent field boundary vegetation, and the strongly structuring effect of hedges and small forest fragments for farmland passerines has been largely demonstrated (Hinsley and Bellamy, 2000). The latter can be positive (Macleod et al., 2004; Whittingham et al., 2001) or sometimes negative, e.g., for species that prefer more open habitats (Mason and MacDonald, 2000; Miguët et al., 2013). After controlling for this influence, a positive effect of crop compositional heterogeneity on birds has consistently been found (Firbank et al., 2008; Lindsay et al., 2013; Miguët et al., 2013), as in plants (Marshall, 2009) and insects. In addition, landscapes with a more complex configuration (smaller field size, with a higher linear length

of field border) were found to stabilize community temporal variation (Henckel et al. submitted), and overall in birds, there may be a stronger effect of landscape configuration than composition (Fahrig et al., 2015).

SELECTED EXAMPLES FROM FLAGSHIP SPECIES

Black-Tailed Godwit Conservation in the Netherlands

Before humans started large-scale modifications of the landscape, the black-tailed godwit, *Limosa limosa*, is thought to have nested in bogs and littoral grasslands that naturally occurred in the riverine delta of what now is the Netherlands (Blankers and Kleijn, 2011). As natural habitats were increasingly cultivated, black-tailed godwits switched to extensively managed anthropogenic grasslands. Over the last century, the largest numbers and highest densities were found on agricultural grasslands in polder areas with high water tables and clay or peat soils in the western and northern parts of the country. Black-tailed godwits are typical of many ground-nesting farmland birds, in that they require large, open areas free of buildings or vertical landscape elements that can be used as a perch by birds of prey. At a large scale, they therefore require structurally simple landscapes that, to the human eye, are monotonous and of low diversity. Heterogeneity is nevertheless critically important (Verhulst et al., 2011) but is expressed in more subtle ways than gross landscape character. In the black-tailed godwit, habitat heterogeneity is mostly driven by water level that, in combination with natural relief, can create complex spatiotemporal gradients in soil moisture that, in turn, result in heterogeneity in vegetation height and composition. This creates a robust system in which godwits, and other so-called meadow birds, are able to find good

nesting and chick-raising habitat under a variety of environmental conditions. With the acceleration of agricultural intensification since the 1960s, and particularly the large-scale drainage of agricultural grasslands in the 1970s and 1980s, the grasslands became increasingly monotonous, both in space and time, and the black-tailed godwit population started to decline rapidly. In the period 1990–2008, the national godwit population declined with approximately 3% per year, but in more recent years the decline seemed to increase to almost 7% per year in the period 2004–08 (Van Paasen and Teunissen, 2010). Conservation of black-tailed godwits is being done by means of national agri-environmental schemes on farmland, as well as in protected areas, with €21 million being spent on so-called meadow bird agreements with farmers and €4 million on meadow bird conservation in spatially much more restricted protected areas (2008 data; van Paassen and Teunissen, 2010). In the succession of Dutch agri-environmental programs up to 2016, meadow bird agreements at best delayed the first seasonal activities of farmers. This potentially protects clutches and chicks of nesting meadow birds (Beintema and Muskens, 1987; Schekkerman et al., 2008). However, because these schemes did not address key factors affecting habitat quality for meadow birds, such as fertilization, ground water level, openness of the landscape, and disturbance during the breeding season, they were largely ineffective (Kleijn et al., 2001; Verhulst et al., 2007; Schekkerman et al., 2008). Reproductive success, in particular chick survival, the key factor determining population dynamics, is higher in protected areas than on farmland with or without meadow bird agreements (Berendse et al., 2004; Kentie et al., 2013), but population trends are negative in many reserves as well. There are examples that show that, even under the difficult environmental conditions of modern agricultural landscapes, godwits can successfully and

sustainably be conserved. These case studies suggest that, for this to be possible, areas have to be large (>200 ha), have to be free of vertical structures (e.g., trees, woodlots, buildings), need to have limited predator abundance, and need to have a significant proportion of the land area with raised water levels to create optimal chick-rearing habitat (Kleijn and Lammertsma, 2012). In the Netherlands, this is generally only possible by implementing agri-environment schemes on farmland in a buffer surrounding protected areas and by managing both the protected area and adjacent farmland as if they formed a single unit. Such efforts are generally only successful when an ornithologically, as well as agriculturally, skilled coordinator is present who oversees and coordinates all activities in the area and who has the policy tools and authority with farmers to optimize black-tailed godwit management by means of implementing last-minute management adjustments (Kleijn and Lammertsma, 2012).

Little Bustard and Other Threatened Top Predators

Insects and small mammals make up the bulk of biomass in food supply for higher trophic levels in agroecosystems. For example, the common vole, *Microtus arvalis*, represents a key resource in the trophic chain in agroecosystems (Lambin et al., 2006). In western France, the Montagu's harrier, *Circus pygargus*, appears to be highly dependent on common vole abundance, since its population density displays a numerical response to the cyclic dynamics of this prey species (Salamolard et al., 2000; Millon and Bretagnolle, 2008). Thus, the persistence of this flagship raptor species is conditional upon the maintenance of common vole populations. And the abundance, in particular the amplitude of the peak vole years, appears to be related to the presence of the grasslands at the landscape scale (Bonnet et al., 2013), although there is a

complex interplay between dispersal and density dependence processes, likely linked to grassland habitat quality (Pinot et al., 2016).

Similarly, differences in grasshopper availability appear to be critical to little bustard, *Tetrax tetrax*, productivity. The little bustard population of western France, which winters in Spain (Villers et al., 2010), has undergone one of the steepest declines documented to date for a bird species in Europe: 7800 males in 1978 to 390 in 1996 (a decrease of 95% in 18 years: Inchausti and Bretagnolle, 2005; Bretagnolle et al., 2011b), and 250 in 2016. There is a strong positive relationship between annual average grasshopper abundance (calculated as the mean abundance over the surveyed grasslands for a given year) and total annual productivity of little bustard, as estimated by the number of fledglings counted in postnuptial groups (Bretagnolle et al., 2011). The case is not unique, as many bird chicks in cereal systems feed primarily on insects (e.g., Rands, 1986; Baines et al., 1996; Panek, 1997) and especially on Orthoptera. In western France, different agri-environment schemes (AESs) have been implemented to protect the little bustard. Most of these schemes were based on grassland restoration and modification of grassland management (Berthet et al., 2012, 2014). The strong decline in little bustard has been linked to a reduction in the areas of perennial habitats that are suitable for breeding, but also to strong decreases in insects (particularly grasshoppers) for feeding (Inchausti and Bretagnolle, 2005; Bretagnolle et al., 2011b). The latter resulted simultaneously from a decrease in the grasslands in which the grasshoppers breed, as well as the intensive use of insecticides and herbicides (reducing food availability for these insects). The conservation strategy for little bustard was therefore to counteract the loss of habitat and the low availability of food resources. Protection measures providing food resources and favorable nesting plots (reducing agricultural activities to

minimize the risk of destruction of nests and incubating females), and more generally to encourage farmers to restore perennial vegetation covers, were developed. The little bustard population, which had shown an initial decrease by a factor of five in just 8 years (about 13% per year since 1996), has then recovered in no more than 5 years (Bretagnolle et al., 2011b). More recently, however, the wheat price increased (especially in 2008 and 2009), in addition to changes in AESs and rules following the last Common Agricultural Policy (CAP) reform (2014), and these two factors resulted in strong decrease of grasslands (by up to 45%) and AES contracts. Little bustard populations in western France started to decrease again.

Skylark and Heterogeneity at Multiple Scales

The skylark *Alauda arvensis* is one of the most iconic farmland birds across Europe, and its decline has been symbolic of wider declines in farmland biodiversity since the 1990s. Skylarks both breed and overwinter in grassland and arable habitats, feeding and nesting on the ground. Given that sufficient food is actually present, the species critically needs access to bare ground for foraging on seeds in the winter (adults) and on insects in the breeding season (for feeding to chicks): variation in accessibility can override food abundance (e.g., Atkinson et al., 2005). Agricultural intensification has potentially affected skylarks through (1) a loss of seed-rich winter habitat with autumn sowing, and increasingly effective herbicides and efficient harvesting, (2) reduced food availability and accessibility in pastures with high nitrogen inputs that have denser swards and much reduced weed floras, (3) loss of insect food for chicks from broad spectrum pesticides and loss of weed flora supporting them, and (4) loss of breeding opportunities, especially late in the breeding season, due to the high vegetation

density in autumn-sown cereal crops (Donald, 2004; Wilson et al., 1997; Siriwardena et al., 1998a). However, of these potential effects, only one can be limiting for populations in a given location and season, and conservation solutions need to identify the limiting factor and put in management to reverse it. Demographic analysis has shown that skylark breeding success per nesting attempt is good, in common with most other farmland passerines, suggesting that effects on this demographic rate, such as effects on chick food, have not been important (Siriwardena et al., 2000). The paucity of ring-recovery data means, however, that knowledge of variation in skylark survival is poor, but the survival of many farmland granivores has been affected negatively by reduced seed availability (Siriwardena et al., 1998b), and while skylarks can eat green vegetation in winter cereal crops, this probably does not provide sufficient resources for them (Green, 1978). Analyses of population trends reveal a strong correlation with areas of winter cereal stubbles (Gillings et al., 2005): this could show the effect of seed availability in stubble on survival, but also could reflect the benefits of spring cropping allowing higher numbers of breeding attempts per pair per year. Evidence from studies at the field and farm scales supports an important role for numbers of attempts in skylark demography (Wilson et al., 1997; Donald, 2004). To inform potential solutions, both the potential winter and spring drivers of skylark population decline can be regarded as reflecting effects of a loss of heterogeneity in space or time. Multiple breeding attempts per year require both nest cover and access to food to be maintained over several months. In arable fields, this means fine-scale heterogeneity, so these resources can be found within the areas of individual territories, as well as this structure then being replicated in different fields through the growing season. Autumn-sown crops are poor for late season breeding, but they provide better cover than spring crops early in the breeding season,

so a heterogeneous cover of each is beneficial (Wilson et al., 1997; Donald, 2004). In grassland, such structural heterogeneity needs to persist through the season, which means extensive management because improved pasture and silage fields will feature swards that are too dense and grow too fast.

Conservation management cannot just replace historical land use, because it has to work in the context of modern agriculture. Management manipulating heterogeneity in farmed habitats may be the key to population recovery. Limited crop areas with high seed density, such as individual fields within standard crop or grassland mosaics, can potentially support flocks made up of an entire local breeding population, plus winter migrants. Hence, heterogeneity at the landscape scale, interspersing winter-sown crops with seed-rich winter fallows, should address issues of winter limitation. For breeding birds, heterogeneity benefits need to be more dispersed, to allow territoriality. Hence, a key AES option to address the breeding season problem in arable farmland is “skylark plots” (uncropped areas of a few square meters in winter cereal crops), which effectively replace interfield heterogeneity with that within fields, providing access to nesting sites (Morris et al., 2004). This conservation option probably requires further development because its efficacy has been found to vary with habitat context, suggesting interactions with the facilitation of predation and specificity to farming systems (Morris et al., 2004; Berg and Kvarnbaeck, 2011); the evidence shows also that the option can succeed when placed in the right context (Morris et al., 2004; Schmidt et al., 2017; Dillon et al., 2009). It is therefore worthy of establishment at the landscape scale, as in English AESs, but needs to be monitored and revised if necessary. However, it can be unpopular with farmers who dislike actions that interfere with crop management, so there is also a role for enhanced engagement and communication.

CONCLUDING REMARKS AND FUTURE PROSPECTS

Many species of birds have undergone sharp declines in Europe (Gibbons et al., 1993; Potts, 1997). In Western Europe, about 1% of the avifauna of lowland landscapes disappears annually (Donald et al., 2001; Julliard et al., 2004). In France, between 1989 and 2003, bird populations (all species combined) have declined by 3%, while birds using agricultural habitats declined by 25% (Julliard et al., 2004). It has been repeatedly suggested that the environmental impact of intensive agriculture may rely more on oversimplification of landscapes at all spatial scales (fields, farm, region) than intensification of production from a given cropped area. We provided here some evidence that indeed landscape simplification has a strong impact on biodiversity, though the balance between local factors (e.g., intensive use of agrochemicals) and landscape scale factors is still debated.

Understanding the causes of historical population changes to aid prediction of the impact of future land use change and to suggest management actions relies on knowledge of the relationships between the abundance of species and habitat characteristics. In terms of conservation, AES and NATURA 2000 are the two main leverage tools available (if not the only ones) to try to mitigate the devastating effects of intensive agriculture on biodiversity. Grasslands have a critical role in shaping the distribution and abundance of organisms of different trophic levels including plants, grasshoppers, small mammals, and birds. Despite the evidence that increasing landscape diversity and/or restoring grassland habitats in intensive cereal systems can have major benefits, AESs have not commonly targeted the conservation and management of permanent and temporary grasslands, due to the difficulties in restoring a market for livestock where it has disappeared (Berthet et al., 2012). Organic farming is an interesting alternative (which is often included in

AESs, for example, in France), although neither AESs nor organic farming always provide the positive effects on biodiversity that they are intended to deliver (Kleijn and Sutherland, 2003). The management of grassland habitat in such ecosystems is therefore critical for both the maintenance of ecosystem services such as those depending on functional biodiversity, as well as for the conservation of threatened species. For the latter, grassland must be managed at the regional rather than the local scale because many bird species forage on vast areas and occur at rather low densities, so that their population dynamics can operate at the regional scale. Moreover, most studies have dealt with only a few taxa, and there is a lack of studies investigating the effects of biodiversity loss on ecosystem services, e.g., soil conservation, nutrient cycling, groundwater purification, pollination, or biologic control (but see Geiger et al., 2010). Therefore, restoring crop diversity is itself not sufficient to increase biodiversity, but managing both Semi-Natural Elements (SNE) and crop diversity should be targeted. Grassland-arable cropping integration could be an important mechanism for diversifying agricultural systems.

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