Integrated land use modelling of agri-environmental measures to maintain biodiversity at landscape level

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Abstract

Integrated land use models (ILM) are increasingly applied tools for the joint assessment of complex economic-environmental farming system interactions. We present an ILM that consists of the crop rotation model CropRota, the bio-physical process model EPIC, and the farm optimization model FAMOS[space]. The ILM is applied to analyze agri-environmental measures to maintain biodiversity in an Austrian landscape. We jointly consider the biodiversity effects of land use intensity (i.e. nitrogen application rates and mowing frequencies) and landscape development (e.g. provision of landscape elements) using a rich indicator set and region specific species-area relationships. The cost-effectiveness of agri-environmental measures in attaining alternative biodiversity targets is assessed by scenario analysis.

The model results show the negative relationships between biodiversity maintenance and gross margins per ha. The absence of agri-environmental measures likely leads to a loss of semi-natural landscape elements such as orchard meadows and hedges as well as to farmland intensifications. The results are also relevant for external cost estimates. However, further methodologies need to be developed that can jointly and endogenously consider the complexities of the socio-economic land use system at farm and regional levels as well as the surrounding natural processes at sufficient detail for biodiversity assessments.

Keywords

Integrated farm land use modeling, biodiversity indicators, agri-environmental policy, landscape elements

1 Introduction

The United Nations declared 2010 as the *Year of Biodiversity* to raise public awareness on the role of biodiversity in supplying ecosystem services to humans. It shall also make aware of the objectives of the Rio *Convention on Biological Diversity*. The convention calls for a significant reduction of biodiversity losses from national to global scales by the year 2010, which may not be achieved without unprecedented additional efforts (Convention on Biological Diversity, 2010). Among the most important drivers, i.e. land use, atmospheric CO_2 concentration, nitrogen deposition, acid rain, climate, and biotic exchanges, land use have had and will have in the 21st century the most important although bi-directional effects on biodiversity globally (cf. Sala et al., 2000). One direction is that agricultural land use is responsible for severe losses through the conversion of natural habitats to farm land as well as for the on-farm losses induced by production intensification (Pimm and Raven, 2000; Secretariat of the Convention on Biological Diversity, 2006). The other direction is that crop and animal breeding have enriched genetic diversity and extensive agricultural land use has created cultural landscapes of high ecological values and unique semi-natural habitats (Wrbka et al., 2004; Fischer et al., 2008; EEA, 2009). However, ongoing processes in agriculture such as intensification and abandonment of farm land can threaten these high nature value (HNV) landscapes (Benton et al., 2003; Tscharntke et al., 2005) and may reduce the ecosystem services to the society (Björklund et al., 1999). Intensification of farm land is frequently accompanied by high agro-chemical inputs. Semi-natural landscape elements such as field margins or hedges have been removed as a consequence of field consolidations to alleviate mechanization. Fragmented farm land has been negatively perceived by stakeholders such as "the blackest of evils, to be prevented by legislative action as one would attempt to prevent prostitution or blackmail" (Farmer, 1960, p. 225; cited in Bentley, 1987 p. 31). In contrast, ecologists and agronomists have often alerted to the loss of valuable landscape elements as the consequence of field consolidations (Krebs et al., 1999; Benton et al., 2003) with biodiversity as “the big looser of technological changes in agriculture” (Giampietro, 1997 p. 161). Many species have been able to adapt to changing environments during the previous millennia of agricultural development. However, adaptation is limited with fast and large scale changes such as during agricultural industrialization (Tucker, 1997). Its scale and dynamics of pressures may even increase under global change phenomena such as climate and demographic changes (Tilman et al., 2001). Furthermore, abandonment of marginal agricultural lands as observed in several parts of Europe (Höchtl et al., 2005; Strijker, 2005) often lead to substantial losses of HNV farm land (Tasser and Tappeiner, 2002; EEA, 2009). Consequently, the Common Agricultural Policy (CAP) for instance has adopted biodiversity policies in recent reforms such as the birds and habitats directives, the NATURA 2000 networks, or agri-environmental measures (European Commission, 2006). Accompanying monitoring and evaluation are already integral elements of many policies. They require scientific analysis tools to investigate complex systems such as agricultural land use and ecosystem effects ex-post as well as ex-ante (Pain and Pienkowski, 1997; Mattison and Norris, 2005). Integrated land use models are able to analyze such complexities by linking thematic data and disciplinary models.

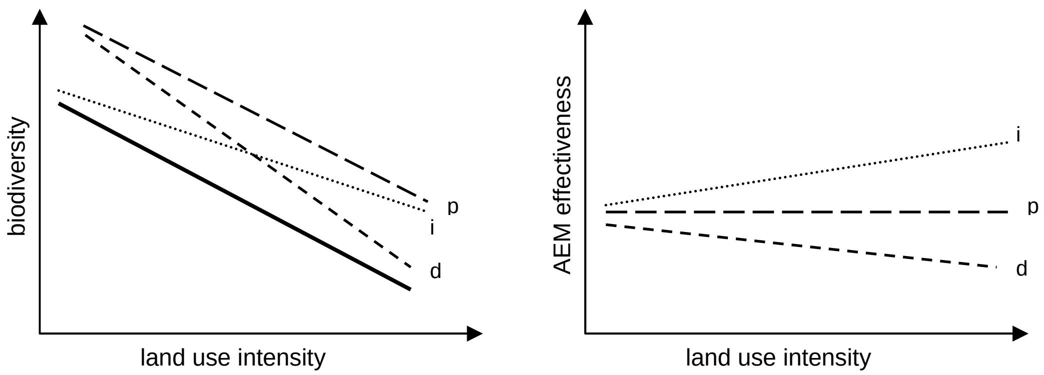
In this article, an integrated farm land use modeling framework (IMF) is applied to analyze impacts of agri-environmental measures on biodiversity at landscape level. Opportunity costs of biodiversity provision at farm and landscape levels are assessed for an Austrian case study landscape. We do not attempt to model the development of single species but rather apply surrogate indicators, correlations, and sensitivity analysis for species and habitat diversity. We also provide a literature review on landscape ecological foundations for biodiversity in agricultural landscapes and show how they have been applied in land use models (section 2). In section 3, we present the IMF including the data requirements and indicator set applied for biodiversity assessment. Section 4 describes the case study region and scenarios. It is followed by a presentation (section 5) and discussion (section 6) of model results, their policy implications, and remaining methodological challenges.

2 Biodiversity from a landscape ecological and agricultural economics perspective

2.1 Biodiversity and agricultural land use

Reviews on landscape ecological studies identify a vast amount of concepts, definitions, and indicators with respect to biodiversity and highlight the need for well defined value systems, corresponding research objectives, and indicators (Duelli and Obrist, 2003; Clergue et al., 2005). A basic categorization applicable to different spatial levels separates structural, functional, and compositional attributes of biodiversity (Noss, 1990). The latter represents the frequently applied concept of biodiversity i.e. species or habitat diversity in agricultural landscapes (Duelli and Obrist, 2003). In our analysis, we refer to this concept of biodiversity due to its central role in conservation policies.

Species and habitat diversity in agricultural landscapes may be influenced by a number of natural site conditions such as slope gradients, soil quality, and climate (Kleijn et al., 2009), but agricultural land use seems most relevant with respect to the magnitude of effects and controllability. Particularly two aspects are seen as important, which are land use intensity at the field level (e.g. application rates of agro-chemicals, mowing frequencies and livestock densities of meadows and pastures) and the composition and configuration of landscape elements at the landscape level (e.g. extent and distribution of semi-natural farm land, diversity of agricultural crops) (Benton et al., 2003; Tscharntke et al., 2005; Billeter et al., 2008; Concepción et al., 2008; Kleijn et al., 2009). Landscape complexity or landscape structure refers to the spatial distribution of ecotopes such as fields, hedges, or trees in a landscape (cf. Wrbka et al., 2004).



Source: Own figure based on Concepción et al. (2008) and Tscharntke et al. (2005)
Notes: p (land use intensity and landscape complexity are independent), d (impacts of landscape complexity on biodiversity are relatively decreasing with land use intensity), i (impacts of landscape complexity on biodiversity are relatively increasing with land use intensity).

Figure 1: Hypothetical relationships between biodiversity and land use intensity under lower (solid line) and higher (broken lines) landscape complexities (left) and the corresponding effectiveness of agri-environmental measures (AEM) (right)

Empirical studies indicate that land use intensity and landscape complexity are interacting, which determines biodiversity and the effectiveness of agri-environmental measures (Tscharntke et al., 2005; Concepción et al., 2008; Smith et al., 2010). Such relationships are demonstrated in Figure 1. It shows a hypothetical linear relationship between land use intensity and biodiversity (left graph, solid line). Increases in landscape complexity, such as attained by agri-environmental programs (broken lines), can shift the curve and/or alter its slope. A parallel shift would reflect a proportional higher but in relative terms a constant impact of landscape complexity on biodiversity. The relative impact of landscape complexity on biodiversity is increasing with land use intensity as shown in (i) or decreasing as shown in (d). Consequently, the rate of species diversity through extensification decreases (increases) with increasing (decreasing) landscape complexity, which has been shown for arable weed species (Roschewitz et al., 2005). In addition, landscape complexity also determines the relative effectiveness of agri-environmental measures that regulate land use intensity (Figure 1, right; cf. Concepción et al., 2008).

2.2 Biodiversity assessment in economic land use optimization models

There are several strategies to include biodiversity aspects in economic land use models and we review some contrasting examples. Thereby, we only focus on optimization models due to their importance for ex-ante policy analysis and the methodology applied hereafter.

One way is to directly include biodiversity objectives together with others in a multi-objective function. The challenge here is to find representative preference systems to rank and weight multiple societal objectives either prior to the model application or to the selection among multiple model results (e.g. Groot et al., 2007; Holzkämper and Seppelt, 2007). Alternatively, biodiversity maintenance can be directly included with constraints to guarantee minimum provision levels (e.g. van Wenum et al., 2004). The challenge here is to represent minimum provision levels in spatial contexts and to appropriately account for synergies and trade-offs between species and habitats to avoid model solution infeasibilities. Others have applied economic land use optimization models for alternative scenarios and have sequentially evaluated scenario results with respect to biodiversity effects (e.g. Brady et al., 2009). Consequently, the corresponding land use and biodiversity effects of predefined policy objectives may only be assessed with multiple model runs.

Any of these methodological options relies on functions between land use and biodiversity either directly or indirectly. Direct functions can portray rather simplistic relationships between biodiversity and single management criteria such as nitrogen application rates and biodiversity (Groot et al., 2007) or dose-response functions of nitrogen deposition (Fraser and Stevens, 2008). Münier et al. (2004) have applied a database on ecotopes to assess species diversity. Ecotopes represent homogenous biodiversity response units consisting of bio-physical and land use management characteristics. A frequently applied concept is some kind of species-area relationship that relates the expected number of species to its habitat area (Brady et al., 2009; Nelson et al., 2009). More elaborated approaches combine economic land use models and stand-alone biodiversity models. These models have been developed as simulation models for single species to estimate population developments under changing habitat quality (e.g. Johst et al., 2002; Wätzold et al., 2008), or as regression models based on empirical field data for several species or taxonomic groups (e.g. Gottschalk et al., 2007, 2010; Holzkämper and Seppelt, 2007). Indirect or surrogate indicators can replace direct biodiversity functions. They are frequently applied in cases where detailed data on species-habitat relationships are lacking and build on the experiences of empirical case studies from landscape ecology. Both land use intensity and landscape structure may be covered by such indicators (e.g. Pacini et al., 2003; Reidsma et al., 2006).

3 Materials and methods

3.1 Overview on the research methodology

We apply an IMF to assess the impacts of selected agri-environmental measures on biodiversity at field and landscape level. The IMF consists of the farm optimization model FAMOS[space], the crop rotation model CropRota, and the bio-physical process model EPIC (Environmental Policy and Integrated Climate; Williams, 1995; Izaurralde et al., 2006). CropRota provides farm specific crop rotations, which are integrated in EPIC together with crop management data and geo-referenced field and climate data to simulate field specific bio-physical impacts. Further details on these two model components, data, and validation are presented in Schönhart et al. (2009, 2010a). Crop rotations and crop yields are inputs to FAMOS[space], which explicitly considers alternative land use intensities as well as landscape elements. Biodiversity effects of land use choices are evaluated with a set of field and landscape indicators. Because the composition and configuration of a landscape is influencing ecosystem processes and habitat quality, we apply landscape metrics at the end of the model chain to quantify the spatial biodiversity impacts of landscape development scenarios. Neighborhood metrics are used to analyze the settings of specific landscape elements and their roles as ecological networks. Figure 2 gives an overview on the research approach.

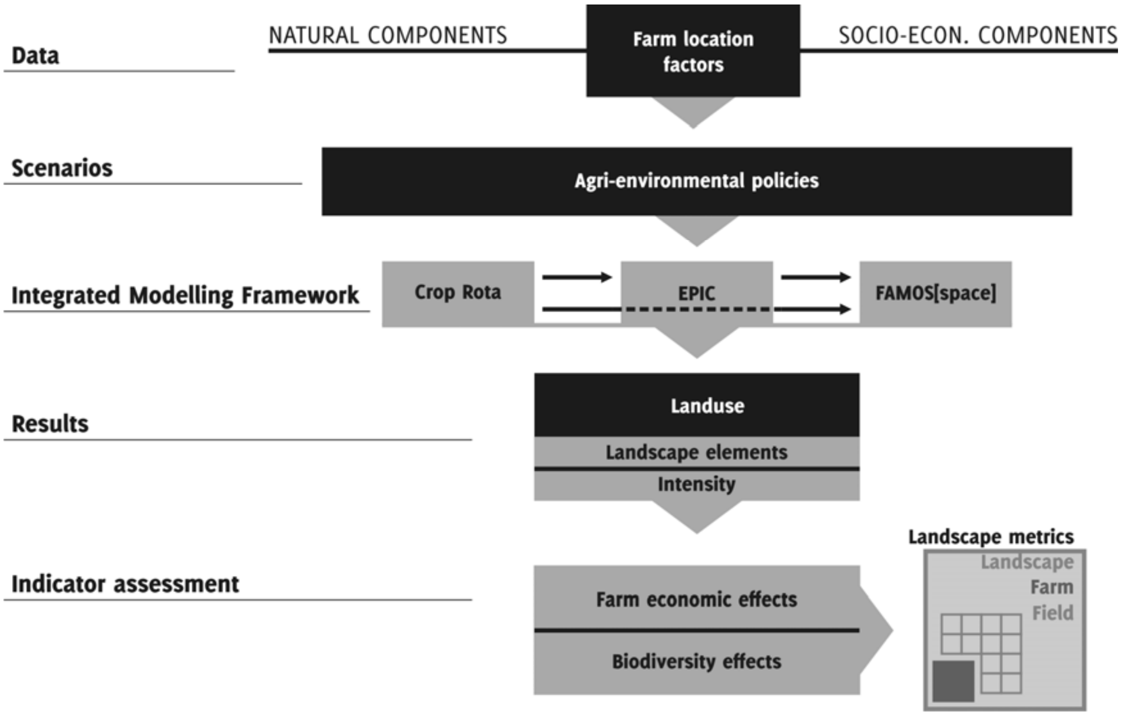


Figure 2: Overview on the research approach

3.2 Land use intensity and landscape elements in FAMOS[space]

FAMOS[space] is developed in GAMS (General Algebraic Modeling System, www.gams.com) and is based on the FAMOS model (Farm Optimization System, Schmid, 2004). It has been expanded towards environmental and landscape structure analysis by integrating spatial field contexts. A loop procedure allows for sequential and independent simulations of farms in a landscape. The model is described in detail in Schönhart et al. (2010b). Here, we only discuss its representation of land use intensity and landscape elements.

FAMOS[space] is a mixed integer linear farm programming model. It maximizes total farm gross margin (*GROS*) subject to farm specific resource endowments and field properties (farm location factors) by finding optimal production and management activities. Equation (1) portrays the objective function in FAMOS[space]. *OPUT* represents farm output variables and *PROD* alternative farm production activities for livestock and land use. Prices, costs, and subsidies are represented by ρ, χ, and ν.

$$\max.\, GROS = \sum(OPUT \cdot \rho_{OPUT}) + \sum(PROD \cdot \nu_{PROD}) - \sum(PROD \cdot \chi_{PROD}) \tag{1}$$

Fields are the spatial decision units in FAMOS[space] and provide the basic structure for all further indicator assessments. A field’s distance to the farmstead, soil quality, size, weather, and slope conditions determine crop production costs and yields. The alternative land use activities (*PROD*) on a field consist of crops and forages as well as landscape elements (hedges and orchard meadows).

Orchard meadows are an ecologically valuable agro-forestry system wide-spread over central Europe that consists of tall fruit trees dispersed over managed meadows or pastures (Herzog, 1998). The products such as fruits or cider can be sold on markets. We assume the long-term average fruit price for orchard fruits of 60.7 €/t and average harvest productivities and yields. For further details on the implementation of orchard meadows in FAMOS[space] see Schönhart et al. (2010b). Other landscape elements in FAMOS[space] are hedges. Hedges do not usually provide marketable outputs but a number of social benefits such as reductions in wind erosion and nutrients leaching as well as provision of nesting and feeding grounds to farm land birds (Hinsley and Bellamy, 2000). Furthermore, they are widely acknowledged for their role in connecting habitat patches in fragmented agricultural landscapes (Baudry et al., 2000). Establishment costs for hedges depend on their design, which is related to a purpose, e.g. wind protection or habitat improvement. In Lower Austria they vary between 10,000 €/ha and 20,000 €/ha including maintenance costs during the first years according to the Agrarbezirksbehörde Niederösterreich, a public authority responsible for hedge establishment (personal communication, 8 November 2005). Farmers may be granted subsidies covering up to 90 % of these establishment costs. Roth und Berger (1999) estimated establishment costs of about 9,000 €/ha for smaller hedges to increase habitat quality. In our analysis, we assume costs of 12,000 €/ha including maintenance and do not consider any establishment subsidies. The hedges as well as orchards are assumed to remain for a 30-years period. Annuities have been calculated using a discount rate of 5 %. Transitions from cropland to grassland and vice versa seem unlikely and are not considered in the model, because forage production options are possible on croplands, and permanent grassland conversions to cropland are prohibited by cross compliance legislation. Transitions between landscape elements and other land uses are possible on pre-defined sites, which have been identified from historical surveys.

Land use intensity in FAMOS[space] is considered by crop rotation choices, nutrient application rates (N, P, K) as well as mowing frequencies. The model can choose among four intensity levels – high intensity (HI), medium intensity (MI), low intensity (LI), and organic farming.

3.3 Landscape data and indicator selection

The IMF operates on a high level of detail with respect to field, farm, and landscape location factors. Consequently, it requires farm resource and landscape element data from field to landscape levels (for a description of the data sources see Schönhart et al., 2010a). Besides the common set of economic and farm resource data, high resolution field data are of crucial importance as well. They are extracted from the geo-referenced IACS (Integrated Administration and Control System) database and merged with other thematic IACS and statistical data sources. Instead of applying the concept of artificial landscapes (cf. Brady et al., 2009), actual fields have been integrated as polygons to portray the landscape as detailed as possible with respect to their production and ecological functions. Field data are complemented by landscape element data to derive current and potential sites for landscape element establishments. Maps on landscape elements have been generated by a semi-automated segregation process based on ortho- and arterial photos (cf. Schauppenlehner et al., 2010; Schönhart et al., 2010b), from which potential sites are drawn considering landscape planning criteria.

We apply a broad set of surrogate indicators that indicate the biodiversity effects from alternative agricultural land uses. Their choice has been guided by empirical studies on the relationship between habitat quality and biodiversity. Indicators include an intra-patch dimension at the field level and a matrix dimension at the landscape level (Dauber et al., 2003). Field level intra-patch indicators, such as habitat type and land use intensity describe field management effects (Table 1). Habitat type is based on the concept of hemeroby, which is an indicator for the naturalness of habitats and frequently applied in empirical and model-based biodiversity assessments (Zechmeister and Moser, 2001; Zechmeister et al., 2002, 2003; Zebisch et al., 2004; Schreiber, 2010). Land use activities from FAMOS[space] are classified according to the hemerobic states as presented in Zechmeister et al. (2002) and aggregated to the landscape level. Nitrogen application rates can serve as important biodiversity indicator (Zechmeister et al., 2003; Schmitzberger et al., 2005; Kleijn et al., 2009). It is complemented by mowing frequencies of permanent grasslands (Zechmeister et al., 2003) to describe land use intensity.

At the landscape level, matrix indicators based on landscape metrics describe the extent, composition, and spatial configuration of different habitats (Bennett et al., 2006). ‘Extent’ relates to the total area of habitat types in a landscape and is approximated by the intra-patch indicator for habitat quality. The prominent "mosaic concept" in landscape ecology (cf. Duelli, 1997) pronounces landscape composition and configuration. Composition or habitat variability refers to the number (richness) and relative areas (evenness) of habitats in a landscape (Duelli, 1997; Bennett et al., 2006), which both can be expressed by the Shannon diversity index (SDI) (cf. Gottschalk et al., 2007, 2010; Brady et al., 2009). SDI-categories are different cropland and grassland activities (e.g. wheat or corn production, orchard meadows) as well as landscape elements. Two other indicators for landscape composition are the total number of patches (NP) and the mean patch size (MPS). However, composition does not sufficiently describe the spatial configuration of habitats in the landscape, which is important for network elements such as hedges. In this analysis, habitat configuration is indicated by the total length of patch edges (TE) between the two land use categories cropland and grassland and landscape elements (orchard meadows, hedges). For instance, edge length is an indicator for plant species diversity on grasslands (Marini et al., 2008). Furthermore, we assess the network of landscape elements as it can be important for example to habitat specialists and larger mammals (cf. Steffan-Dewenter, 2003; Pereira and Rodríguez, 2010). Therefore, we sum the area with a distance of more than 50 m from the next landscape element as an indicator for the distribution of landscape elements in a landscape.

Table 1: Overview on the type and measurement of biodiversity indicators

spatial level	indicator	description
intra-patch	habitat value	mean hemerobic state
	nitrogen use intensity	mean nitrogen application rate (kg/ha)
	mowing intensity	mean mowing frequency of permanent grassland (cuts/a)
matrix	landscape diversity	Shannon diversity index (SDI) $SDI = -\sum_{i}^{S}\left[\left(PROD_i / PROD_I\right)\cdot \ln\left(PROD_i / PROD_I\right)\right]$
	patch number	Total number (TP) of different land use patches
	patch size	Mean size of different land use patches (MPS) (ha)
	edge length	total length of edges (TE) between landscape elements and grassland or cropland (km)
	habitat connectivity	total area with a distance > 50m from landscape elements (ha)

Notes: All indicators are analyzed at the landscape level. $PROD_i$ refers to the area of a land use activity i and $PROD_I$ to the area sum over all land use activities. S is the number of different i.

Intra-patch and matrix indicators differ by the spatial level of indicator application - either at single fields, subfields, or the landscape. However, model results on biodiversity effects are only presented at the landscape level.

3.4 Biodiversity data and sensitivity analysis

The surrogate biodiversity indicators are supplemented by correlations between selected indicators and plant species diversity, as the latter is seen as useful indicator for overall species richness (Sauberer et al., 2004). Data on plant species diversity are extracted from published field study data. Schmitzberger et al. (2005) investigated cropland at different locations in Austria and relate nitrogen application rates to arable weed diversity. Zechmeister et al. (2003) correlate total plant species richness (vascular and bryophyte plants) in grasslands based on data from Austrian wide samples. Furthermore, the scenario values for habitat quality (hemeroby) of the landscape are correlated to the species number of bryophyte plants based on an Austrian wide assessment (cf. Zechmeister and Moser, 2001). Due to similar climatic and land use conditions, we assume that all three studies are an appropriate approximation for relative changes in biodiversity depending on different management intensities. We have translated absolute values to relative changes to reduce biases from varying site conditions. A site is assumed to reach its maximum in species diversity with a hemerobic state of five and a rate of 15 kg/ha nitrogen fertilizer application on grassland and zero kg/ha on cropland.

Landscape complexity and land use intensity are interacting at the landscape level, which may also determine the effectiveness of agri-environmental programs (compare to section 2 and Figure 1). The possibilities for functional relationships are numerous and are a potential source of uncertainty. Hence, we apply a sensitivity analysis to show the impact of different functional relationships discussed in section 2.1. We assume three hypothetical linear functional relationships based on observations from Schmitzberger et al. (2005) and Zechmeister et al. (2003) and analyze the effects of nitrogen application rates (kg/ha) and landscape complexity (SDI) on relative plant species diversity. In all three functional forms, landscape complexity is assumed to be effective between the lowest SDI value and the largest possible in the landscape. The SDI value either increases the upper (at 0 and 15 nitrogen kg/ha) or lower level (at 150 nitrogen kg/ha) of the relative plant species diversity. Table 2 lists the different functional forms of the sensitivity analysis. For example, gl_i_0.5 is a functional relationship of type (i) for grassland (gl), i.e. landscape complexity is assumed to be more effective on biodiversity at higher land use intensities. In the scenario, the relative plant species diversity is increased by 50 percentage points at high land use intensities (150 nitrogen kg/ha) and a normalized

SDI value of 1, while it remains unchanged at low intensities (15 nitrogen kg/ha) with the lowest normalized SDI of 0.53, which occurred in the reference scenario (cf. following section).

Table 2: Sensitivity analysis on functional forms between land use intensity, landscape complexity and biodiversity

functional relationships	change of relative biodiversity value (percentage points)	nitrogen application rate (kg/ha)
gl_i_0.5	50	150
gl_i_1.0	100	150
gl_d_0.5	50	15
gl_d_1.0	100	15
gl_p_0.5	50	15 and 150
gl_p_1.0	100	15 and 150
cl_i_0.5	50	150
cl_i_1.0	100	150
cl_d_0.5	50	0
cl_d_1.0	100	0
cl_p_0.5	50	0 and 150
cl_p_1.0	100	0 and 150

Legend: gl (grassland), cl (cropland); functional relationships: p (land use intensity and landscape complexity are independent), d (impacts of landscape complexity on biodiversity are relatively decreasing with land use intensity), i (impacts of landscape complexity on biodiversity are relatively increasing with land use intensity).

4 Case study landscape and model scenario descriptions

The IMF is applied to a landscape in the Lower Austrian ‘Mostviertel’ region, which is characterized by a rather homogenous northern part with respect to landscape structure and relief and a southern part that features the traditional landscape element of the ‘Mostviertel’ region, namely orchard meadows on gentle hills. We model 20 conventionally producing farms specialized in cash crop or livestock production or a mixture of both. The farms manage about 430 agricultural fields with 546 ha in total, of which are 399 ha cropland and about 147 ha permanent grassland. We have chosen a smaller portion of adjacent fields out of the total modeled farm land for the designation of potential landscape element sites due to data limitations. Fields outside are assumed to have neither existing nor potential landscape element sites.

In our case study analysis, we assess the joint effects of landscape structure and land use intensity as a consequence of agri-environmental measures. We have developed a reference scenario (REF) and an agri-environmental policy scenario with different measures (S1-S6). The latter introduces agri-environmental measures with alternative levels of land use intensities and landscape elements (Table 3), which are seen as important to maintain farmland biodiversity such as farmland birds (Tucker, 1997). Landscape elements such as hedges and orchard meadows can be grown on existing sites or may be established on new sites, which both sum up to the potentially available sites.

Table 3: Overview on the case study scenarios

scenario	description	
	landscape elements	*land use intensity*
REF	no intervention	no intervention (nitrate directive binding)
S1	no removal of existing sites	no intervention (nitrate directive binding)
S2	no removal of existing sites at least 50% of potentially available sites on each farm	low or medium intensity
S3	100% of potentially available sites on each farm	low or medium intensity
S4	100% of potentially available sites on each farm	low intensity
S5	100% of potentially available sites on each farm	low intensity, at least 25% extensive grassland
S6	100% of potentially available sites on each farm	low intensity, at least 75% extensive grassland

Both, hedges and orchard meadows are considered as valuable semi-natural elements for habitat and biodiversity provisioning in rather intensively managed grassland landscapes of Austria to which the case study landscape belongs (Wrbka et al., 2005). In the case study landscape, 1.8 ha orchard meadows but no hedges are currently cultivated. New orchard meadows can be established in the model on historical orchard meadows land, which amounts to 4.1 ha (cf. Schönhart et al., 2010b). The establishment of hedges is often regarded to increase the ecological value of a landscape while simultaneously allowing profitable agricultural land use (Briemle et al., 2000). In landscapes with a high share of orchard meadows, hedges increase the network among frequently fragmented orchard meadows patches (Weller, 2006), while species in hedges such as birds may benefit from the vicinity of extensively used grasslands as feeding grounds (Herzog et al., 2005). We have identified potentially available sites for hedges along field edges and in the case of large fields throughout fields according to their proximity to other semi-natural areas such as forests and orchard meadows. The hedge width is set to 3 m and can double where farmers establish hedges at the same field boundary. This leads to a total hedge area of 3.3 ha, which sums up to a total landscape elements area of 9.2 ha or 1.7 % of the total agricultural land. The ecologically effective distance criterion is assumed to be 50 m (cf. Herzog et al., 2005).

5 Results

The main results of our case study analysis with respect to the biodiversity indicators are presented in Table 4. Without policy interventions, the average nitrogen application rate among all farms is 145 kg/ha, which is below the maximum levels permitted by the nitrate directive. In the reference scenario (REF) all orchard meadows have been removed and neither new orchard meadows nor hedges are established. The introduction of an agri-environmental measure to promote landscape element maintenance in scenario S1 has only minor effects on most indicators due to the small share of existing orchard meadows in relation to the total farm land (= 0.3 %). However, effects on individual farms can be economically important as farm gross margins decrease by 280 €/ha orchard meadows on average, despite the already considered clearing costs for orchard meadows in REF. The establishment of additional landscape elements and medium to low land use intensities (MI, LI) in S2 and S3 lead to decreasing average nitrogen application rates mainly on cropland (Figure 4 (a)). The average hemerobic state, SDI, and NP increase, and MPS decreases, which indicates a more heterogeneous landscape. Total farm gross margin ($GROS_{landscape}$) is moderately lower on average with -2 % in S3 compared to REF. Direct and opportunity costs increase with further reductions in land use intensity (LI) in S4 and reduce $GROS_{landscape}$ by 13 percentage points compared to S3. From S4 onwards, a small share of agricultural land becomes abandoned. The introduction of minimum extensive grassland areas mown only once a year (25 % of all permanent grassland in S5 and 75 % in S6) further reduce land use intensity to average nitrogen application rates of 77 kg/ha in S5. In the model, farms partially compensate the forage yield losses in quantity and quality by cultivating temporary grassland on their croplands and by forage purchases. $GROS_{landscape}$ in S6 is 24 % below REF, which can even be up to 42 % for single farms. In S6, all potentially available sites are covered by landscape elements, the land use intensity is reduced to the low level (LI) and 75 % of the permanent grassland is extensified. Consequently, landscape heterogeneity further increases with an SDI in S6 of 73 % of the maximum possible value compared to 53 % in REF. Figure 3 presents maps for the scenarios REF and S6.

Figures 4 (b) and (c) correlate nitrogen application rates and the hemerobic value with the reductions in $GROS_{landscape}$ and the relative biodiversity changes of plants (cf. section 3.4). According to Figure 4 (b), plant species on cropland increase from about 10 % in REF to 50 % in S6 and grassland species from 60 % to 90 % as a consequence of grassland extensification. From a hemeroby perspective, changes in species number show similar magnitudes resulting from extensification and landscape element creation if the higher share of cropland and therefore its higher weight compared to grassland is acknowledged. Clearly Figures 4 (b) and (c) cannot be simply aggregated as hemeroby among others is a function of nitrogen application.

The results of the sensitivity analysis are presented in Figure 5 (i), (d), and (p) (cf. section 3.4 and Table 2). The figures show the influence of both, the shape of the hypothetical functional relationships as well as the assumed quantitative influence of the SDI on plant species developments on grassland and cropland. Despite their different shapes (cf. Figure 1), all three functional relationships cause similar effects on relative plant species diversity due to the simultaneous changes of land use intensity and landscape complexity in the scenarios. In general, the sensitivity of landscape complexity is lower at higher land use intensities and therefore becomes more important during extensification. The highest changes of relative plant species diversity are observed for the parallel shift (p) of 100 percentage points at low and high land use intensities (cl_p_1.0, gl_p_1.0) and nearly double relative plant species diversity on cropland and increase grassland values by 50 %.

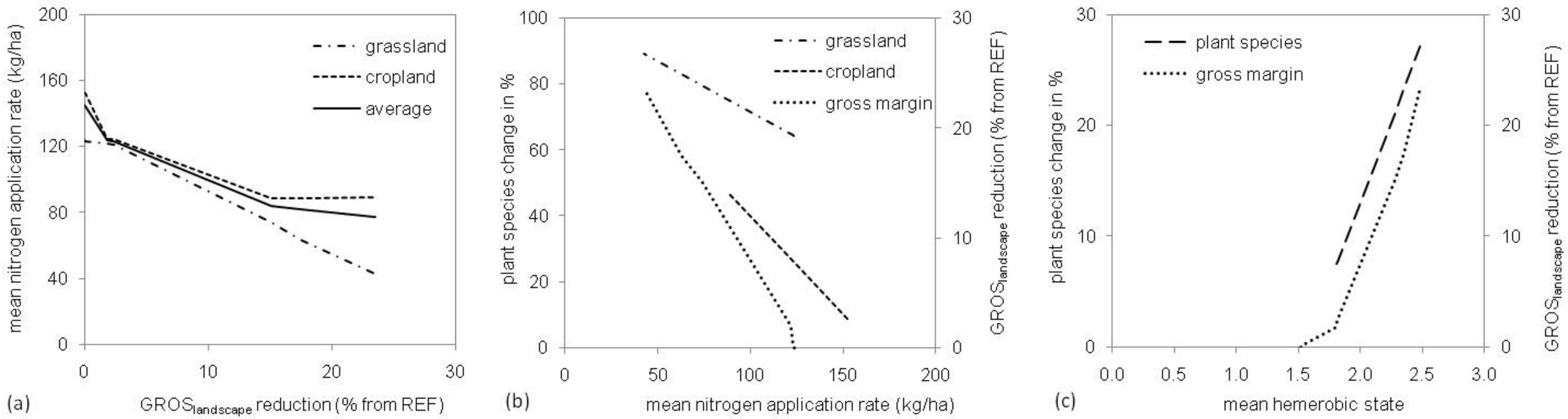
Sources: Own drawing with data from BMLFUW (2008)

Figure 3: The landscapes for the reference scenario (REF, left) and the agri-environmental policy scenario S6 (right)

Table 4: Average landscape indicator results and total farm gross margin ($GROS_{landscape}$) in % from the reference scenario (REF)

		indicator (average value for the landscape in % from REF)								
		habitat value	*nitrogen use intensity*	*mowing frequency*	*landscape diversity [SDI]*	*patch number [NP]*	*patch size [MPS]*	*edge length [TE]*	*habitat connectivity*	*$GROS_{landscape}$*
scenario	*S1*	0	-1	0	1	7	-7	-	-6	0
	S2	20	-14	0	14	15	-14	400	-25	-2
	S3	20	-15	0	16	21	-17	739	-37	-2
	S4	53	-42	0	27	20	-17	739	-37	-15
	S5	53	-43	-17	36	27	-22	737	-37	-18
	S6	60	-47	-50	36	26	-21	737	-37	-24

Note: For a description of the indicators see Table 1; reference value for TE is scenario S1.



Sources: Own figures, (b) based on Schmitzberger et al. (2005) and Zechmeister et al. (2003), (c) based on Zechmeister and Moser (2001)

Figure 4: (a) Trade-off curves between total farm gross margin ($GROS_{landscape}$) %-changes from the reference scenario (REF) and mean nitrogen application rates (kg/ha); (b) correlation between mean nitrogen application rates (kg/ha) and total farm gross margin ($GROS_{landscape}$) %-changes from the reference scenario (REF) as well as %-changes of species richness of vascular plants (cropland) and vascular and bryophyte plants (grassland); (c) correlation between mean hemerobic state and total farm gross margin ($GROS_{landscape}$) %-changes from the reference scenario (REF) as well as %-changes of species richness of relative bryophyte plants

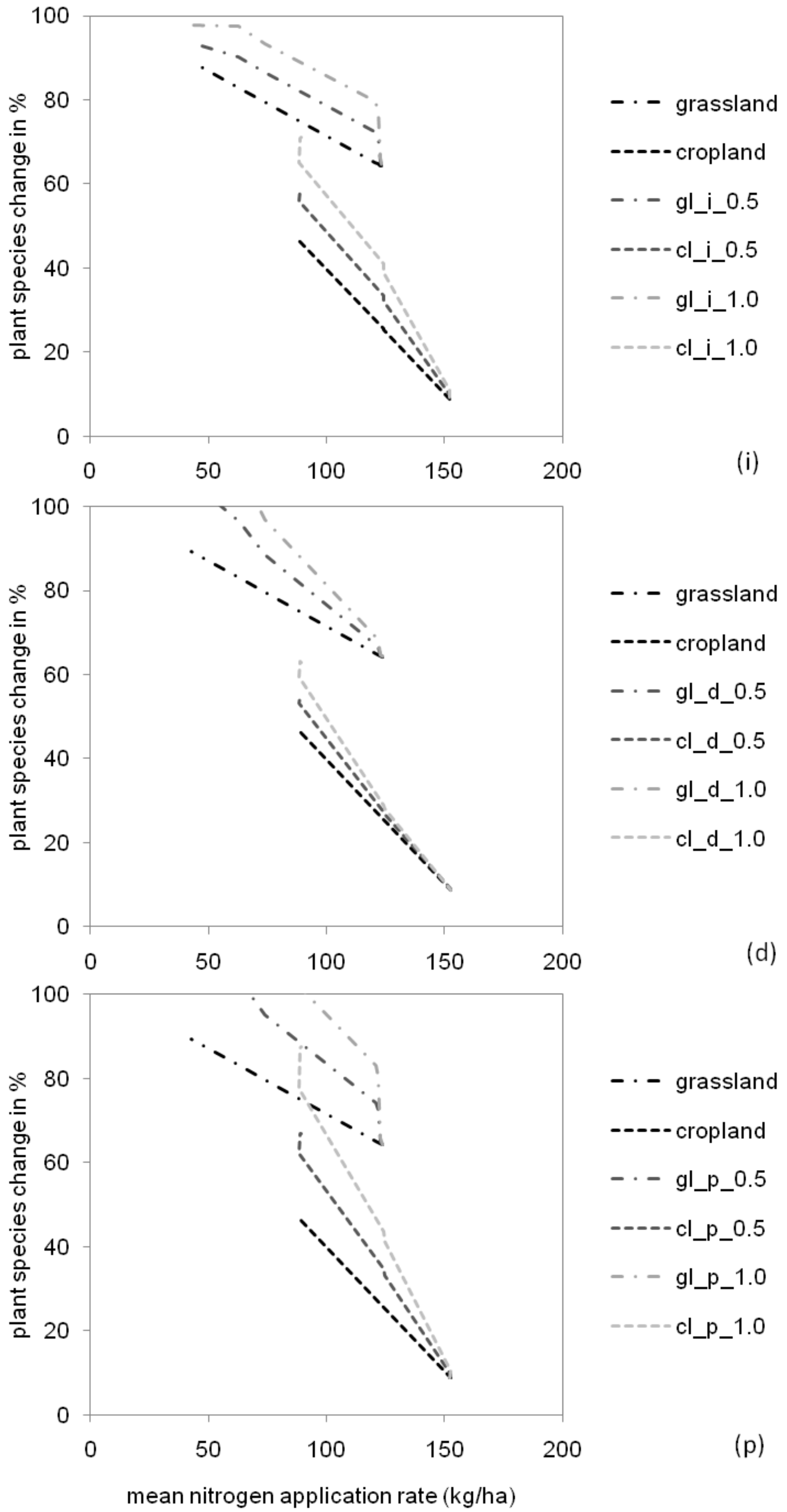


Figure 5: Sensitivity analysis results for hypothetical correlations between nitrogen application rates (kg/ha), landscape complexity (SDI) and relative plant species richness (%)

6 Discussion

6.1 Agri-environmental policy implications of the case study results

Farm economic and biodiversity effects of agri-environmental measures have been assessed in an integrated farm land use modeling framework (IMF). The implemented measures represent rather strong limitations on land use compared to the current situation. For example, scenario S6 forces farms to a production intensity comparable to organic farming, to the maintenance and establishment of landscape elements on all potentially available sites in the case study landscape (cf. Figure 3), and to extensification of 75 % of the permanent grassland to one-cut meadows. Birdlife (2009) proposes a 10 %-standard of farmland that should be mainly managed for biodiversity conservation. Such value is approximated in the scenarios S5 and S6 dedicating 7.3 % and 20.4 % of total farm land for nature conservation (landscape elements, extensive meadows), respectively. However, these shares seem rather high considering the already available forest patches and other natural vegetation in the case study landscape. Model results show declining total farm gross margins of up to 25 % on average with single farms facing even higher reductions. These results are based on historical land use and livestock choices. Therefore, if farmers have already applied agri-environmental measures in the past, FAMOS[space] may underestimate the full intensification potential and opportunity costs. In the model, farms can compensate forage yield losses by purchases or forage productions on cropland. We may also underestimate opportunity costs, because agri-environmental measures implemented on a larger scale likely reduce the regional supply of marketed forage and increase its price, which is currently assumed constant in the model. On the other side, products from extensive land use systems may gain higher market prices, which are also not considered in FAMOS[space].

A negative relationship between biodiversity and gross margins per ha has also been shown by other empirical studies (Zechmeister et al., 2003; Schmitzberger et al., 2005). The absence of agri-environmental measures likely leads to a loss of semi-natural landscape elements such as orchard meadows and hedges as well as to farmland intensification. Pascual and Perrings (2007) highlight the need to correct for market failures in order to reduce the disinvestments in farmland biodiversity. Empirical findings indicate that well structured agricultural landscapes of high ecological value are appreciated by the society (cf. Lindemann-Matthies et al., 2010) and agri-environmental measures have been implemented to reward farmers for maintaining heterogeneous landscapes and to reduce land abandonment and intensification. However, premiums seem insufficient to maintain HNV farm land at a European scale and even in Austria, where the support for HNV farmland is higher than in other European countries (EEA, 2009). For example, decreasing areas of hedges in grassland landscapes as well as extensive orchard meadows have been observed (Pötsch et al., 2009; Schönhart et al., 2010b) and empirical studies could not confirm a major influence of the Austrian agri-environmental program ÖPUL on the development of landscape elements in selected agricultural landscapes (Bartel, 2006).

6.2 A critical note on the interpretation of biodiversity results

Besides basic farm model assumptions such as constrained farm profit maximization, other assumptions have been made on the relationship between land use management and biodiversity. We followed a rather European perspective and see agriculture as potential supplier of biodiversity and pleasant landscapes subject to appropriate land management (cf. Tscharntke et al., 2005). However, there is a second perspective in landscape ecology that underlines the role of undisturbed land for nature protection. Its proponents argue that intensification in some regions may spare land in others for conservation purposes (Green et al., 2005; Polasky and Vossler, 2006). There seems to be no final answer on the superiority of one of these two strategies over the other so far (Pain and Pienkowski, 1997; Tucker, 1997), because it may depend on the detailed objectives of biodiversity and habitat protection as well as on local contexts and framework conditions such as the demand for agricultural products under population growth. Furthermore, it may also depend on the question whether or not it is possible to develop intensive agricultural systems without harming the environment (e.g. precision farming).

Context sensitivity also relates to the assumed relationships between land use management and biodiversity. In the case of species diversity and nitrogen application rates (Figures 4 (b)), we assume a linear relationship although there are empirical evidences for non-linear relationships as well (Kleijn *et al.*, 2009). Furthermore, one has to stress contradicting empirical studies about the effectiveness of agri-environmental measures on biodiversity maintenance (Kleijn *et al.*, 2001) and the importance of local or site-specific conditions as well as the species to be protected. Heterogeneous landscapes are not favorable to all species (Filippi-Codaccioni et al., 2010), which highlights the need for clear objectives prior to any policy implementation and evaluation.

The complex nature of biodiversity in agricultural landscapes calls for a rich indicator set instead of single indicators (Duelli and Obrist, 2003). We are aware of this complexity and therefore evaluate land use results from FAMOS[space] with a rich surrogate indicator set. The correlations are based on Austrian case studies and expressed in relative rather than absolute terms. Furthermore, we apply sensitivity analysis to show the effects of a changing landscape complexity (SDI) on the correlation of land use intensity and relative plant species richness. The sensitivity is drawn on hypothetical relationships from landscape ecology literature (Tscharntke et al., 2005; Concepción et al., 2008). Surrogate indicators are criticized for their limited explanatory power (Clergue et al., 2005) and further research is necessary to improve both, the validity of intra-patch as well as matrix indicators as proxies for biodiversity. This includes knowledge on the interactions between both levels (cf. Concepción et al., 2008), which may determine the effectiveness of agri-environmental measures especially in already heterogeneous landscapes such as the case study landscape. Such interactions have been assessed by the sensitivity analysis. It shows that the interference of landscape complexity on biodiversity is relevant for results interpretation and reveals the substantial uncertainties related to the effects of agri-environmental measures concerning biodiversity. Although hypothetical in its nature, the sensitivity analysis gives an impression on the magnitude of interaction and emphasizes the importance of further research. Functional relationships like the ones presented can be used to better target agri-environmental measures.

6.3 Methodological considerations on integrated farm land use modeling and biodiversity

There is increasing demand for collaborative research between different disciplines and between scientists and other stakeholders to better assess the relationships between farm decision making, agri-environmental measures, land use, and farm land biodiversity on the landscape level (Opdam and Wascher, 2004; Pascual and Perrings, 2007; Smith et al., 2010). Integrated land use models can play an important role in transdisciplinary research processes. Due to the possibilities of mapping and landscape visualizations, stakeholders are enabled to easily interact in scenario definition and results discussion. Furthermore, approaches as the one presented in this study provide land use costs and data for quantification and mapping of a range of land use effects, which are the basis for valuations within cost-benefit assessments.

Bio-economic farm land use models can also act as interdisciplinary tools for knowledge integration on biodiversity because they are able to provide the necessary interfaces to landscape ecology and estimate field and farm specific opportunity costs of alternative land use management choices. The latter is achieved in our IMF by the integration of field specific crop yields, which have been simulated with the bio-physical process model EPIC. Crop rotations are integral of sustainable agricultural systems, which have been generated by CropRota for each farm. The IMF allows to jointly consider important land use effects such as on biodiversity on a field and landscape level and to assess the cost-effectiveness of agri-environmental measures or landscape planning strategies such as the design of environmental networks for biodiversity enhancement (Dutton et al., 2008; Nassauer and Opdam, 2008). In contrast to some approaches presented in section 2.2, we evaluated biodiversity effects subsequent to the modeling process, which allowed us to apply a rich indicator set and correlations on biodiversity and land use including sensitivity analyses. We did not integrate the indicator sets directly in FAMOS[space], because this would create non-linearity and would require simultaneous optimizations at farm and landscape levels. Furthermore, any kind of biodiversity targets or objective function weight would be needed, which are usually difficult to obtain. To conclude, the integration of biodiversity in economic land use optimization models remains rather superficial concerning the assumptions on functional relationships between land use intensity, landscape complexity and biodiversity. However, joint optimization of land use and biotic effects seems desirable such as presented by Groot et al. (2007) and Parra-López et al. (2009). Consequently, further methodologies need to be developed that can jointly and endogenously consider the complexities of the socio-economic land use system and the surrounding natural processes at sufficient detail for biodiversity assessments.

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