Analysing agricultural landscape change in a marginal European landscape

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Author's contribution:

In paper 1, I performed the data preparation and had the main responsibility for data analysis and writing. The co-authors contributed to the planning of the study and gave valuable comments.

In paper 2, I developed the approach, prepared and analysed the data, and was responsible for the writing. The first co-author performed the aerial photograph interpretation. The second and third co-authors provided helpful ideas and criticism.

In paper 3, I had the main responsibility for data analysis and writing while the first co-author performed the land-use modelling. The other co-authors contributed constructive suggestions and helpful comments.

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1. General introduction

1.1 Background

1.1.1 Marginal agricultural landscapes

Society's demand for food, fibre, and other raw materials has heavily influenced the composition and structure of European landscapes. In about 7500 years of agriculture, anthropogenic activities have led to a variety of agriculturally managed landscapes with complex land-use mosaics (Meeus et al., 1990; Vos and Meekes, 1999). Today, about 184 million hectares, covering 44% of the total land area of the 27 European Union member states, are classified as 'utilised agricultural area' (UAA; Eurostat, 2008).

In the second half of the 20th century, agricultural landscapes have faced major transformations all over Europe (Bastian and Bernhardt, 1993; Meeus, 1995). Shortly after World War II, subsistence agriculture was being practised to reduce the need for food and thus the proportion of cultivated land was generally high. Since about 1955, progress in agricultural mechanisation, plant breeding, and pesticide formulation, easy access to (mineral) fertiliser and various additional factors related to the agricultural practice have resulted in agricultural land-use intensification (Matson et al., 1997). Economic revival, booming markets, and international trade of agricultural products (growing global demands for food, feed, fuel, and fibre) also contributed to this development. Meanwhile, traditional and diverse management systems are largely replaced by modern production systems. However, this process has mainly affected the most productive land and landscapes.

The 'opposite' trend in agricultural change, i.e. the process of marginalisation began simultaneously in landscapes where physical constraints for agricultural production (e.g. unproductive soils or steep sites) reduce competitiveness and place severe limits on technical and structural adaptation (Brouwer et al., 1997, MacDonald et al., 2000). In the so-called marginal agricultural landscapes, arable crop production was no longer profitable and was thus largely replaced by extensive grassland use, plantation forestry, or natural succession on abandoned land (Baldock et al., 1996). This process was additionally aggravated by an agrarian structure (e.g. small farms, small land parcels) inappropriate for modern agriculture and by income alternatives outside of the agricultural sector.

Resulting from these processes of intensification and marginalisation, arable farming today concentrates on the flattest and most productive land. However, substantial areas in marginal landscapes, mainly in (sub)mountainous regions, still feature agricultural land, despite adverse climatic, edaphic, and/ or topographic conditions for production (Brouwer, 2006; MacDonald et al., 2000). Marginal agricultural landscapes are widespread across Europe. About 56% of the European Union's UAA is officially classified as 'less-favoured areas' (Council of the European Union, 2005), i.e. as marginal agricultural landscapes.

In general, the retreat of agriculture from less favourable sites affected natural resources like water and soil as well as components of biodiversity. Positive effects like high quality surface and ground water (e.g. Brouwer, 2006; Mander et al., 1999) and negative effects like loss of farmland habitats and decrease in farmland plant species (e.g. Burel and Baudry, 1995; Henle et al., 2008; Korneck et al., 1998; Waldhardt et al., 2003) are evident. Moreover, land abandonment resulted in the impoverishment of our cultural heritage and loss of local identity (Vos and Meekes, 1999). Nevertheless, despite these general trends and consequences, even today quite few marginal agricultural landscapes offer a rich variety of farmland habitats for plant and animal species, mainly outside the lowlands and resulting from a high diversity of environmental conditions and low-intensity farming systems (cf. Baldock et al., 1996; MacDonald et al., 2000). Plant species richness may be specifically high for semi-natural grasslands (e.g. Eriksson et al., 2002; Simmering et al., 2006; Wellstein, 2007). But also certain faunistic groups such as arthropods or birds are likely to have profited from a mosaic of low-intensity farmland habitats (e.g. Dauber et al., 2003; Jeanneret et al., 2003; Jonsen and Fahrig, 1997; Robinson et al., 2001; Weibull et al., 2003; Woodhouse et al., 2005). Thus, many marginal agricultural landscapes were identified as areas of high biodiversity conservation value (cf. Brouwer et al., 1997; MacDonald et al., 2000). However, the biodiversity in these landscapes may be threatened by future agricultural landscape changes (cf. Jongman et al., 2002).

Due to economic and social pressures faced by marginal landscapes in the context of recent developments in EU Common Agricultural Policy (CAP) and economic globalisation (cf. Robinson and Sutherland, 2002; Strijker, 2005), marginalisation is likely to remain an ongoing trend in agriculture. The analysis of agricultural landscape change has therefore gained more and more attention in recent decades and will become even more important in the future as land-use changes will for better or worse continue to affect landscape functions and processes (Sala et al., 2000; Tilman et al., 2001).

1.1.2 Landscape change research

Analysis of landscape change has become widespread in recent years. Landscape-change studies typically focus on the analysis of causes, processes, and consequences of land-cover and land-use change trough time (cf. Wu and Hobbs, 2002). They primarily investigate the influence of socioeconomic and environmental factors on landscape change (e.g. Bürgi and Turner, 2002; Hietel, 2004; Poudevigne et al., 1997; Reid et al., 2000) or analyse the effects of land-cover changes on ecological functions and processes (e.g. Cousins and Eriksson, 2002; Verheyen et al., 1999).

Landscape change analyses highly rely on multiple spatio-temporal landscape information. Data on land use (i.e. management practices) and/ or land cover (i.e. land-use types or classes) may be derived from satellite data (e.g. Jobin et al., 2003; Munroe et al., 2004; Romero-Calcerrada and Perry 2004), aerial photographs (e.g. Mendoza and Etter, 2002; Sklenička, 2002; van Eetvelde and Antrop, 2004), historical maps (e.g. Bender et al., 2005; Cousins, 2001; Nikodemus et al., 2005), or modelled scenarios (e.g. Farrow and Winograd, 2001; Rounsevell et al., 2006; Verburg et al., 2006). Alternative sources may be published statistics and census data (e.g. Brown et al., 2005; Fjellstad and Dramstad, 1999). Data availability sets the restrictions to landscape change analyses with respect to the considered time periods (i.e. few years to several decades) and areas (i.e. from the individual patch of land to sizeable regions). Geographical Information Systems (GIS), which have been continually improved in recent decades, are normally used to integrate these data sets and to measure the complex spatial and temporal changes in the landscape pattern (Käyhkö and Skanes, 2006; Kienast, 1993). Despite of multiple studies dealing with landscape change analysis, there are still substantial deficiencies in our knowledge about the nature of agricultural landscape changes in marginal landscapes.

Results from landscape change analysis revealed ‘that landscapes have memory, in the sense that the characteristics we see today are often carried over from previous management regimes’ (Haines-Young, 2005). However, few studies can be found that make a classification of landscapes that considers both current land-cover patterns and their dynamics (Haines-Young, 2005). The classification of current land-cover patterns with respect to their past dynamics allows to systematically identify areas, which have been highly dynamic in the past. These areas may also be potentially sensitive to future land-use change. To incorporate the temporal dimension into landscape classification is particularly important in marginal agricultural landscapes, where concepts on how to manage these landscapes in the future are urgently needed (Baldock et al., 1996; Frede and Bach, 1999; MacDonald et al., 2000; Pinto-Correia et al., 2006). Since the quantification of long-term land-cover changes at the landscape scale seems to be hindered by adequate data sets and methods, landscape change analyses with an emphasis on the identification of land-cover patterns and dynamics still pose a challenge.

Research in marginal landscapes has shown that large portions of arable land have been consecutively abandoned in favour of grassland (Baldock et al., 1996; MacDonald et al., 2000). As a consequence of this successive land-use change, the current landscape pattern consists of a large number of grassland patches that differ in age, i.e. in the duration of grassland management after cessation of arable farming. Recent studies revealed that grassland age may have a strong impact on various ecological functions and processes like biodiversity (e.g. Austrheim and Olsson, 1999; Bruun et al., 2001; Cousins and Eriksson, 2002; Dauber and Wolters, 2005; Holzhauer et al. 2006; Waldhardt and Otte, 2003) and natural resources (e.g. Breuer et al., 2006; McLauchlan et al., 2006). Despite the obvious importance of grassland age as indicator for various ecological functions and processes, there have been, to our knowledge, no attempts to determine grassland age at larger spatial scales. A reason might be that quantifying grassland age in a large-scale context is considered to require area-wide, spatially explicit, high-resolution data on land-cover change for several decades. Thus, adequate methods in this field are needed.

In the scope of landscape change research, a multitude of scenario-based modelling approaches were developed to quantify and predict potential effects of changing socioeconomic factors on the landscape. The focus is often on agricultural policies since these are expected to be a major driving force of landscape change (Robinson and Sutherland, 2002; Strijker, 2005). Recent studies primarily addressed the impact of changing agricultural policies on land use (e.g. Höll and Andersen, 2002; Lehtonen et al., 2005; Rounsevell et al., 2005; van Meijl et al., 2006; Weinmann et al., 2006), but also on water quality (e.g. Bärlund et al., 2005; Schmid et al., 2007), or species diversity (e.g. Gottschalk et al., 2007; Sheridan and Waldhardt, 2006; Sheridan et al., 2007). Scenario-based studies relating agricultural policies and habitat diversity at the landscape scale are, in contrast, scarce (but see Bolliger et al., 2007 and Miettinen et al., 2004). Thus, further research on how agricultural policies may impact habitat diversity is urgently needed.

1.2 Objectives

Given this background, the two general aims of this thesis were (A) to gain insights into the pattern of landscape change in marginal agricultural landscapes, and (B) to develop methods that may support landscape change research at multiple spatio-temporal scales. To this end, we utilised a model region and addressed the following objectives as be described in three separate papers (presented here in Chapter 4, 5 and 6).

(1) Identifying patterns of land-cover change and their physical attributes:

In the first study (Chapter 4), we developed an approach to identify types of land-cover patterns and dynamics (TLPDs) at the rural district scale. The two specific objectives were (i) to classify the rural districts according to land cover and its change between 1955 and 1995, and (ii) to characterise the derived types of land-cover patterns and dynamics with respect to physical attributes that are known to be conditional variables of land-cover change. The underlying hypothesis for the second objective was that land-use change occurred primarily in districts with relatively unfavourable physical conditions for agriculture. We combined recent satellite data with historic information on land cover from 1955. These data were derived from agricultural statistics at the lowest administrative level, the rural district (Gemarkung). We applied a k-means cluster analysis to classify TLPDs.

(2) Assessing the spatial distribution of grassland age:

In the second study (Chapter 5), we focussed on grassland age, since grassland has been identified as the predominant land-use system, which increased in the process of marginalisation. Our objective was to develop a methodological approach to systematically assess the spatial distribution of grassland age in a marginal agricultural landscape. Our approach is based on a representative selection of a large number of grassland patches from regionally differentiated grassland types and a subsequent extrapolation of grassland age from patches to the landscape scale. The method is applicable at several spatial scales (i.e. from patches to districts and landscapes of several hundred square kilometres), and covers the last five decades.

(3) Potential effects of direct transfer payments on habitat diversity:

The third study (Chapter 6) aimed to assess potential effects of alternative direct transfer payment schemes (part of Pillar One of the CAP) on the farmland habitat diversity in a marginal agricultural landscape. Therefore, agri-environmental schemes as supported by Pillar

Two of the CAP were explicitly excluded. We defined (1) a scenario with direct transfer payments coupled to production, (2) a scenario with direct transfer payments decoupled from production, and (3) a scenario phasing out all direct transfer payments to illustrate the effects of varying CAP frameworks. We combined land-use patterns generated by an agro-economic land-use model with data on topography and soil to generate habitat patterns. Habitat diversity was characterised by three indices. Scenario analyses were complemented by an investigation of the farmland habitat pattern in 1995, which served as the basis for comparisons.

Our model region was the Lahn-Dill Highlands, a marginal agricultural landscape in Hesse (Germany), where the process of marginalisation could be observed since the 1950s (Hietel, 2004; Kohl, 1978; Schulze-von Hanxleden, 1972). This landscape also served as study area in the interdisciplinary research project 'Land Use Options for Peripheral Regions' (SFB 299; www.sfb299.de), which aims to develop future land-use systems that are sustainable in terms of economic, social, and ecological landscape functions at multiple scales (Breuer et al., 2007; Frede and Bach, 1999; Waldhardt, 2007). The current investigation is mainly embedded in the landscape ecology subgroup of the SFB, but also in close collaboration with SFB subgroups from other disciplines, e.g. agro-economy, zoology, and resource management.

2. Study area

The Lahn-Dill Highlands are located in the western part of Hesse (Germany) and cover about 1270 km² (Fig. 1). The study area extends from Giessen and Wetzlar in the south to Biedenkopf in the north, and from the river Lahn in the east to the borders of the states of North Rhine-Westphalia and Rhineland-Palatinate in the west. Research on identifying patterns of land-cover change and their physical attributes (Chapter 4) and on assessing the spatial distribution of grassland age (Chapter 5) addressed the entire area of the Lahn-Dill Highlands, while research on the potential effects of direct transfer payments on habitat diversity (Chapter 6) focussed on the Dill catchment covering about 644 km² in the western part of the Lahn-Dill Highlands.

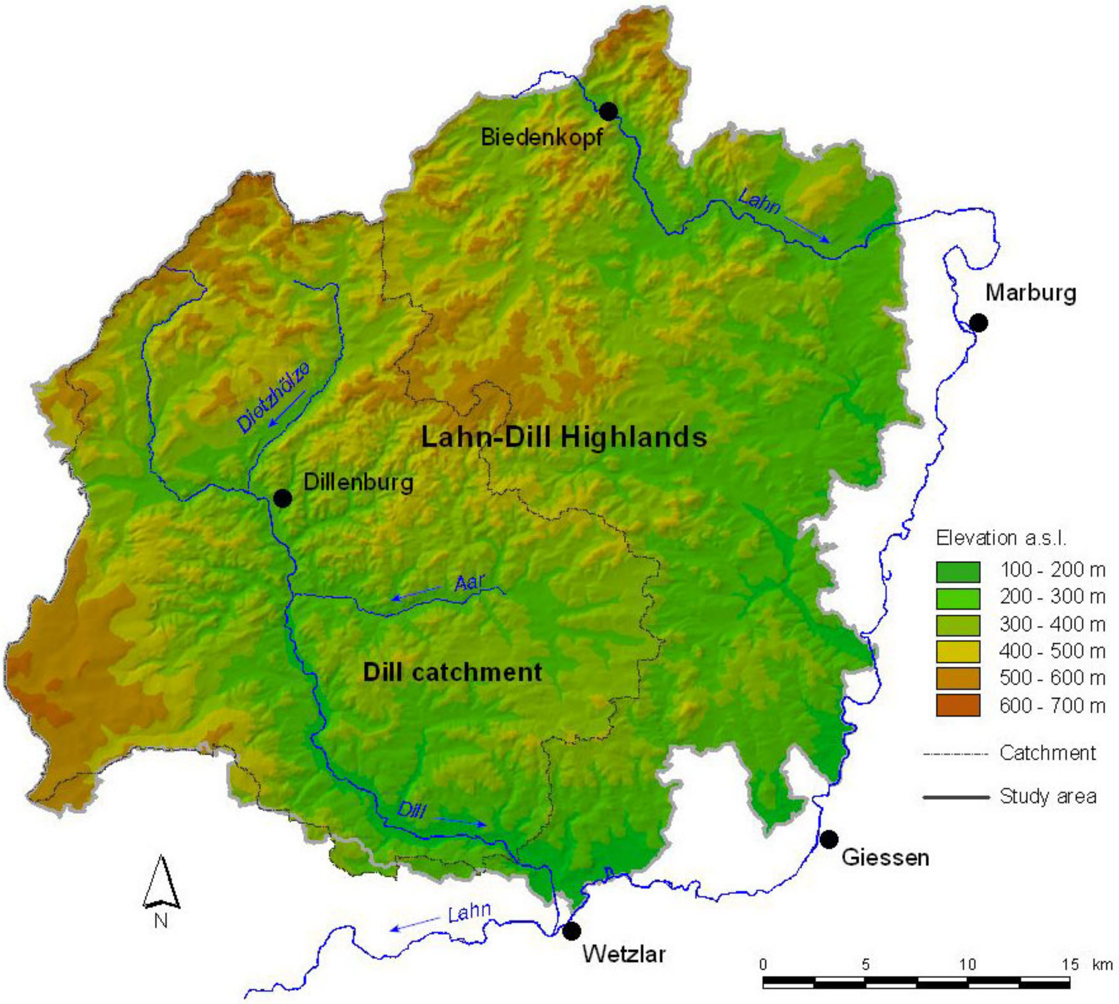

Fig. 1. Topographical map of the study area.

The Lahn-Dill Highlands are characterised by environmental conditions that are relatively unfavourable for cultivation (Frede and Bach, 1999). The low-mountainous landscape with altitudes between 150 and 670 m above sea level (a.s.l.) is characterised by rough and rather damp climatic conditions. The mean annual temperature is between 6° and 8° C, and the mean annual precipitation ranges between 650 and 1100 mm. The Lahn-Dill Highlands are part of the eastern ridge of the Rhenish Uplands and are mainly composed of clay schist, siliceous schist, diabase, and greywacke. Various parent materials (particularly loess and periglacial debris layers from the Pleistocene and Holocene periods) and processes of soil development formed a heterogeneous, small-scale mosaic of soil types (Harrach, 1998). The small-scale mosaic comprises acidic shallow ranker soils and regosols on hill tops and upper slopes, cambisols and luvisols on slopes, and planosols and gleysols in alluvial plains (classification according to FAO (1998)). Overall, the amount of poor soils is relatively high. The agrarian structure is dominated by part-time farming, small farm sizes (mean size of about 14 ha; Waldhardt and Otte, 2003), and a heterogeneous, small-parcelled mosaic of arable fields, grasslands, and fallow lands (mean field size around 0.4 ha; cf. Simmering et al., 2006; Waldhardt et al., 2004). The entire study area has been included in the EU less-favoured area support scheme since 1976 (EC Regulation No 75/268). For these reasons and considering the pronounced land-use changes in the past (see below), the Lahn-Dill Highlands may be considered as a marginal agricultural landscape.

In the Lahn-Dill Highlands, agriculture has always been a matter of small-scale farming providing only a sideline income. Mining and steel industry have had a substantial relevance as non-agricultural employment alternatives. Like in many other marginal European landscapes, the agricultural land-use pattern in the Lahn-Dill Highlands has encountered major changes since the 1950s. Non-agricultural job opportunities within the region and in the adjacent Rhine-Main area as well as the introduction of the EC Common Agricultural Policy have led to a substantial abandonment of farms and agricultural land by many part-time farmers and a general decrease in farming activities throughout the region. In some parts, formerly predominant arable crop production was largely replaced by extensive grassland use or by abandoned fields (Hietel et al., 2004, 2005; Kohl, 1978; Schulze-von Hanxleden, 1972). Today, about 32% of the Lahn-Dill Highlands are farmed as agricultural land with grassland as the predominant farming system. More than half of the agricultural land are mown or grazed, mainly with cattle or sheep (Wellstein et al., 2007). Only 11% of the study area are arable land and 5% fallow land (according to modified land-cover data of Nöhles (2000), see Section 3.1). On the latter, stands of Scotch Broom (*Cytisus scoparius*) have become typical ecosystems in the Lahn-Dill Highlands (Simmering et al., 2001). Since most farmers still operate at the margin of economic viability, present day cultivation and grassland management are characterised by a rather low input of nutrients and pesticides.

Due to a combination of heterogeneous environmental conditions and small-scale mosaic of low-intensity farming systems, the Lahn-Dill Highlands feature a high biological diversity and constitute one of the most species-rich marginal agricultural landscapes in Germany (Nowak, 1988). This is particularly true for plant species richness (Korsch, 1999; Nowak, 1992; Nowak and Wedra, 1988; Simmering, 2006; Waldhardt and Otte, 2003; Wellstein, 2007).

3. Materials and methods

This chapter summarises the materials and methods used in Chapter 4, 5 and 6 of this thesis. This includes a brief description of the data sets and the data pre-processing operations as well as the applied methods and techniques. More detailed information can be found in the materials and methods sections of the respective chapters.

3.1 Data sets and data pre-processing

The analysis of landscape change required a variety of landscape information and data pre-processing operations. The data sets comprised land-use data at multiple spatial and temporal scales, but also data on the physical conditions of the study area. Preliminary GIS-based data pre-processing operations like format conversions or filtering were performed to adjust the different data sets for subsequent analyses.

Land-use data

Agricultural statistics published by Hessisches Statistisches Landesamt (1956) were used to analyse historical land cover in 1955 at the scale of rural districts (Chapter 4). For every district, the data set provided the total area of arable land and of grassland (comprising meadows, litter meadows, pastures, and rough pastures). To reconstruct the land-cover history since 1953 at the patch scale (Chapter 5), a chronosequence of black and white aerial photographs (mainly at a scale of 1:12 000 covering an area of about 4 km², 1:24 000 for 1953) was available from Hessisches Landesamt für Bodenmanagement und Geoinformation. A satellite image interpretation (Landsat-TM, 25 m; Nöhles, 2000) from 1995 was used to obtain information on the current land-use pattern. The data set included spatially explicit information on arable land, grassland, deciduous forest, coniferous forest, waters, settlement, and fallow land. The latter corresponds to old fallows with shrub succession. The satellite-derived land-use data showed a relatively high proportion of isolated cells (≤ 0.125 ha) for fallow land. Using ArcGIS 9, we performed a 3 x 3 cells modal filter to remove these cells (Burrough and McDonnell, 2000). The satellite-derived land-use pattern in 1995 served as basis for the analysis of the current agricultural land-cover patterns (Chapter 4), for the selection of grassland patches (Chapter 5), and for scenario modelling and comparison (Chapter 6). Three modelled land-use patterns, generated by the agro-economic land-use model ProLand (Kuhlmann et al., 2003; Sheridan et al., 2007; Weinmann et al., 2006), were used as basis for the derivation of habitat types (Chapter 6). For the data analysis in raster format, the vector-based land-use maps were converted to raster maps with a cell size of 25 m × 25 m.

Physical landscape data

Elevation data (used in Chapter 4, 5 and 6) were obtained from a digital elevation model (DEM, 40 m; Hessische Verwaltung für Bodenmanagement und Geoinformation, undated). In order to correct errors in the DEM, the data set was smoothed by a low-pass filter, i.e. the value for the cell at the centre of a 3 × 3 cells window was computed as a simple arithmetic mean of the values of all other cells of the window (Burrough and McDonnell, 2000). To match cell resolution of the raster data with the land-use map, cell size of the DEM was altered to 25 m × 25 m. The value of the resized cell was estimated by bilinear interpolation resampling technique, which computes the new value by calculating the distance-weighted average from the four neighbouring cells (Burrough and McDonnell, 2000). Based on the modified DEM, we calculated altitude (metre a.s.l.) and classified two elevations, colline (≤ 400 m a.s.l.) and submontane (> 400 m a.s.l.). From the modified DEM, the slope was derived by use of the slope function in the ArcGIS 9.0 Spatial Analyst tool, which calculates the maximum rate of change of elevation between each cell and its eight neighbouring cells (Burrough and McDonnell, 2000). We grouped slopes into the three classes ≤ 5°, 6–10°, and > 10°.

Information on soil characteristics (used in Chapter 4, 5 and 6) were derived from the official digital soil map of Hesse (scale 1:50 000) obtained from Hessisches Landesamt für Umwelt und Geologie (2002). For the data analysis in raster format, the vector-based soil map was converted to raster data with 25 m × 25 m cell size. From the modified soil map we derived information on soil moisture and base-richness. Aggregating information on available water capacity (AWC) in the root zone, the degree of soil wetness, i.e. gleyic and stagnic soil properties, and slope, soil moisture was classified into the four classes dry, mesic, moist, and wet. Base-richness was grouped into the four classes base-poor, moderate, base-rich, and calcareous according to substrate properties of the soils.

Additionally, we compared the districts of the study area (Chapter 4) by applying the agricultural comparability index LVZ (Landwirtschaftliche Vergleichszahl). This index is used to classify German areas that are less favourable for agriculture. It aggregates natural characteristics of agricultural areas such as soil quality, climatic conditions, heterogeneity of soils, and water management problems and assigns an overall rate based on a points system ranging from 0 to 100 for the best value. The LVZ is published by Hessisches Ministerium für Umwelt, ländlichen Raum und Verbraucherschutz (2004) at the district scale.

3.2 Cluster analysis

The k-means cluster analysis was performed to classify six types of agricultural land-cover patterns and dynamics at the district scale (Chapter 4 and 5). The purpose of the non-hierarchical k-means clustering procedure is to classify objects with respect to optional quantitative traits into a user-specified number of clusters. The data sets for the classification were derived for the year 1955 from agricultural statistics for 192 districts of the study area and for 1995 from a satellite image interpretation (Landsat-TM, 25 m raster; Nöhles, 2000). The clustering was based on three variables assessed at the district scale: (i) the percentage of grassland in 1995, (ii) the percentage of fallow land in 1995, both with respect to total area of agricultural land, and (iii) the difference of arable land to grassland ratios of the two years 1955 (from agricultural statistics) and 1995 (from satellite data). The latter variable was included to obtain an estimate for land-cover change in each district. Prior to clustering the following data preparation steps were performed. To improve statistical normality, the percentages of grassland and fallow land in 1995 were arcsine square root-transformed, and the ratios of arable land and grassland in 1995 and 1955 were log-transformed. To meet the assumption of dimensionless variables, the data were standardised to z-scores. Five districts were excluded from the analysis as 'strong outliers' (standard deviation > 3; McCune and Grace, 2002). On the basis of these three derived input variables a k-means clustering (MacQueen, 1967) was performed using STATISTICA 6.0 software (StatSoft Inc., 2001), which allocated the districts into different clusters by minimising the variability within clusters and maximising the variability between clusters. In order to find well-contrasted and compact clusters, we ran the calculations for different user-defined k values ranging from 3 to 8. Preliminary classification revealed that a small number of clusters resulted in a large variability within each cluster. While a larger number of clusters produced a strongly skewed distribution of districts with many small cluster sizes. A value of $k = 6$ achieved the best classification with small within and large between distances as well as homogeneous cluster sizes. Hence, these six clusters were chosen to represent the TLPDs.

3.3 GIS-based data analysis

The analysis of spatially-referenced landscape information required various GIS techniques (e.g. stratification, random sampling, multitemporal aerial photograph interpretation, raster calculations). While almost all GIS-based operations were processed with the ESRI Inc. software packages ArcGIS 9.0, 9.1 and the Spatial Analyst extension, the habitat diversity indices (Chapter 6) were quantified by the software Fragstats 3.3 (McGarigal et al., 2002) and a spreadsheet programme.

Selection of grassland patches

The selection of grassland patches (Chapter 5) was performed by a two-stage stratified random sampling. In a first step (stratification I), we pre-stratified the study area according to the identified six TLPDs at the scale of districts. For the second stratification (stratification II), we classified grassland types within the TLPDs by combining data on soil moisture, base-richness, and elevation. From each of the 50 selected grassland types, we draw a random sample of 20 patches to ensure a balanced representation of each type, i.e. a total of 1000 scattered grassland patches were sampled.

Determination of grassland age

In order to assess the duration of grassland use for the sampled patches (Chapter 5), we reconstructed the land-cover history of each patch by visual multitemporal aerial photograph interpretation. Recent grassland use was differentiated from former land-cover types according to tonal contrast and texture. For the entire study area, a chronosequence of black and white aerial photographs since 1953 was available. However, in order to cover the entire study area a time interval of approximately 5 years was needed, i.e. photographs from the time period 1998-2001, 1989-1994, 1979-1983, 1967-1973, 1959-1962, and 1953 were examined. To determine the patch's grassland age, we moved back in time steps of about 10 years until the land cover changed. This permitted to assign to each sampled patch one of the three age classes young (<18 years), mid-aged (18-47 years), or old (>47 years).

Spatial extrapolation of grassland age

Spatial extrapolation was used to project identified age of the grassland patches to districts (Chapter 5). Based on the three age classes and 50 grassland types, we calculated grassland type-specific age probabilities. In order to determine the areal proportions of grassland age classes at the scale of districts, we used direct extrapolation. For each district, we first weighted the area of each grassland type by its associated age probability and summed over all grassland types. Dividing this sum by the total grassland area of the corresponding district, we obtained the district's age composition with respect to the classes young, mid-aged, and old.

Determination of farmland habitat patterns

We used GIS raster calculation functions to identify farmland habitat types for the study area (Chapter 6). We classified and combined soil moisture, base-richness, and elevation obtained from digital soil maps and DEM to derive physical attributes (Fig. 2).

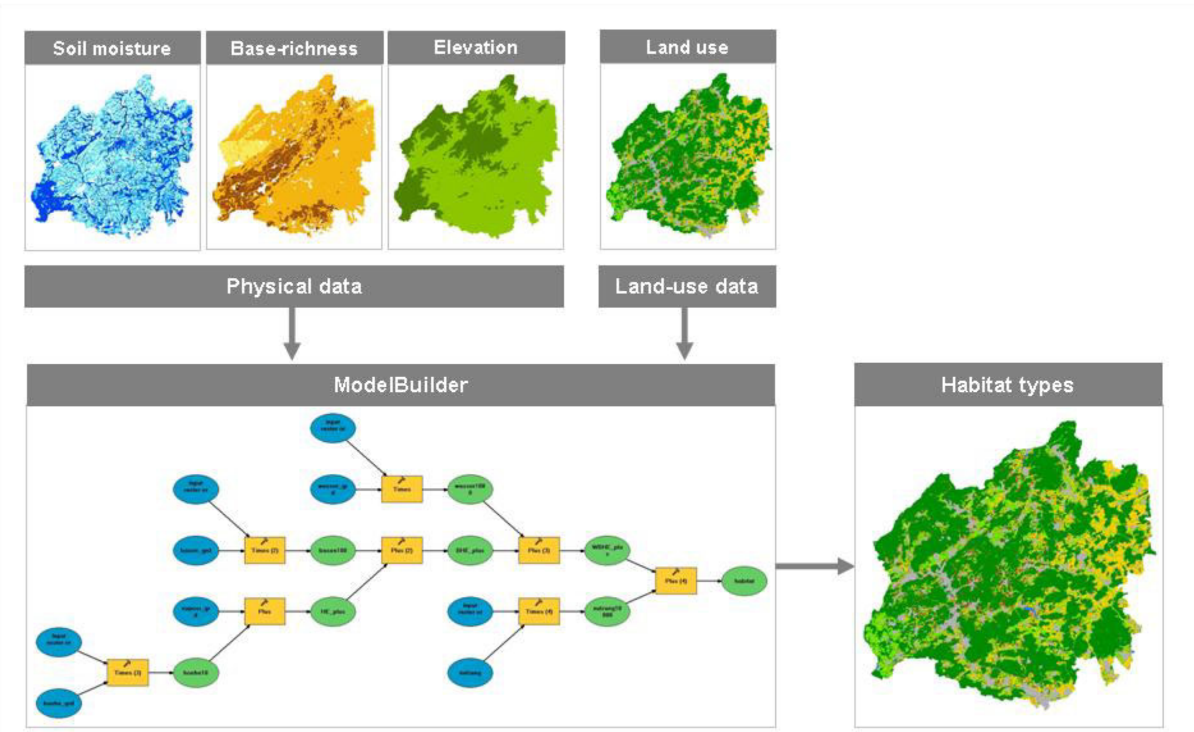

Fig. 2. GIS-based derivation of farmland habitat types.

These physical attributes were intersected with information on land use (i.e. recent land use or the three modelled land-use patterns) in order to generate the farmland habitat patterns for our study area. The intersection was processed within the ESRI Model builder ArcGIS 9.1 and resulted in four maps representing different habitat patterns for recent and modelled land use.

Calculation of habitat diversity indices

For the characterisation of four maps representing different habitat patterns for recent and modelled land use (Chapter 6), a set of three complementary habitat diversity indices, i.e. habitat richness, habitat evenness, and habitat rarity, were calculated. All indices were assessed for a network of 2676 landscape units with the standard size of 22.6 ha (475 m x 475 m). Landscape units containing no farmland habitats (i.e. arable land, grassland, or fallow land habitats) were excluded from analysis. We used the software Fragstats 3.3 (McGarigal et al., 2002) to quantify habitat richness and habitat evenness and a spreadsheet programme to calculate habitat rarity.

3.4 Statistical data analysis

Statistical analyses were primarily based on non-parametric methods due to a lack of normality for most variables of interest. Methods applied were Kruskal-Wallis rank ANOVA, Mann-Whitney U-test, Friedman test, Wilcoxon signed-ranks test and for frequency data G-tests. The calculations were processed with STATISTICA 6.0 software (StatSoft Inc., 2001) and for frequency data with PopTools version 2.6.4 (Hood, 2004), an add-in for the spreadsheet programme MS Excel 2003 (Microsoft Inc., 2003).

The Kruskal-Wallis rank ANOVA was used to test land-cover data as well as physical landscape data for significant differences between the TLPDs (Chapter 4). In case of significance, the analysis was followed by a Mann-Whitney U-test with Bonferroni correction for multiple testing ($p < 0.05$).

To detect significant differences of age class composition of patches (Chapter 5), G-tests were performed. We used such G-tests to determine differences among the TLPDs, among the soil moisture classes, among the base-richness classes, and among the elevation classes. The G-test is equivalent to the more commonly used chi-square test, but is computationally simpler and G appears to follow the chi-square distribution a bit more closely (Sokal and Rohlf, 2004).

The non-parametric Friedman test is a powerful statistical method for the two-way analysis of variance by ranks of several related data that are non-normally distributed (Legendre and Legendre, 1998). Friedman was applied to test the calculated habitat diversity indices for differences among the four habitat maps (Chapter 6). In case of significance, the analysis was followed by a Wilcoxon signed-ranks test with Bonferroni correction for multiple testing ($p < 0.05$).

4. Identifying patterns of land-cover change and their physical attributes in a marginal European landscape

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Abstract

Over the last six decades, land-cover patterns in Europe have dramatically changed, and major future changes are expected. Land-cover changes affect landscape functions. Therefore, methods are needed to include the temporal dimension into landscape classification. By combining recent satellite data with historic information on land cover from 1955, and the application of k-means cluster analysis, we developed an approach to identify types of land-cover patterns and dynamics (TLPDs) at the rural district scale. Our study area was the Lahn-Dill Highlands, a marginal German landscape with a total of 192 rural districts. We identified six TLPDs that showed a general trend of abandonment, but revealed remarkable differences in current land-cover patterns and the directions of land-cover change. The TLPDs showed notable differences in physical attributes: In the eastern part of the area, where elevation, the proportion of steep slopes, and dry soils are low, land cover remained relatively stable. Slight to dramatic changes occurred, in contrast, in the remaining districts with comparatively unfavourable conditions for cultivation. The spatially differentiated information on areas with contrasting land-cover dynamics within a region may be useful to develop effective concepts for future land management.

Key words

Agricultural landscape; Landscape structure; Landscape change; Abandonment; Agricultural statistics; Satellite image

4.1 Introduction

Over the last six decades, patterns of land cover have changed dramatically all over Europe (Bastian and Bernhardt, 1993; Meeus, 1995). Particularly the two major tendencies in agriculture, intensification and marginalisation, have shaped the pattern within this time period. Land-cover change directly affects ecological landscape functions and processes with far-reaching consequences for biodiversity and natural resources (Hansen et al., 2004; Stoate et al., 2001; Vitousek et al., 1997). Furthermore, the current European land cover is expected to undergo major changes in the future, particularly in the course of recent developments in EU agricultural policy. Hence, the analysis of current land-cover patterns with respect to historic changes in cultural landscapes is an essential field of landscape research (Burel and Baudry, 2003; Poudevigne and Baudry, 2003; Turner et al., 2001). Methods are needed to incorporate the temporal dimension into landscape classification approaches (cf. Bastian et al., 2006).

Many studies have focussed on land-cover patterns and dynamics. They primarily analysed the effects of land-cover change on ecological functions and processes (e.g. Cousins and Eriksson, 2002; Verheyen et al., 1999) or studied the driving forces of land-cover dynamics (e.g. Bürgi and Turner, 2002; Poudevigne et al., 1997; Reid et al., 2000; and many others). Resulting from these studies, landscape ecologists have increasingly realised 'that landscapes have memory, in the sense that the characteristics we see today are often carried over from previous management regimes' (Haines-Young, 2005). However, rarely has such work gone on to make a classification of landscapes that considers both current land-cover patterns and their dynamics (Haines-Young, 2005).

The need for such a classification is particularly important in marginal cultural landscapes, since these landscapes have undergone dramatic land-cover changes in the last few decades. In many marginal regions, arable land and mixed systems were largely replaced by extensive grassland, plantation forestry, or natural succession. This directed land-use change had major impacts on landscape functions, like e.g. biodiversity (e.g. Burel and Baudry, 1995; Power and Cooper, 1995; Purtauf et al., 2004). Concepts on how to manage these landscapes in the future are urgently needed (Baldock et al., 1996; Frede and Bach, 1999; MacDonald et al., 2000; Pinto-Correia et al., 2006). In this context, the classification of current land-cover patterns with respect to their past dynamics allows to systematically outline areas with a highly dynamic land use in the past. These areas may also be potentially sensitive to future land-use change. Therefore, spatially differentiated knowledge about past land-cover dynamics is important in landscape planning and resource management (Marcucci, 2000). As land-use dynamics appear to be closely related to the physical attributes of landscapes (e.g. Chen et al., 2001; Pan et al., 1999; Paquette and Domon, 1997), it is also important to consider the patterns of environmental conditions. This is particularly true for marginal regions, which are often characterised by a heterogeneous topography and a broad range of different soils.

To address these issues in a landscape context, remote sensing technologies provide valuable data sets (Dunn, 1990; Turner et al., 2001). The state-of-the-art technique to detect current land-cover patterns on broader spatial scales is the interpretation of satellite images (e.g. Jobin et al., 2003; Munroe et al., 2004; Romero-Calcerrada and Perry 2004). However, satellite data have been continuously available only for the last 25 years (Lillesand et al., 2004), and images dating back to the 1970s have a much lower resolution (e.g. Landsat MSS 80 m). Hence, satellite-derived data may be feasible for short-term analyses of spatially explicit land-cover changes, but to date they are not applicable to analyse changes that occurred prior to this period. Long-term spatially explicit land-cover changes may be reproduced by historical aerial photographs (e.g. Mendoza and Etter, 2002; Sklenička, 2002; van Eetvelde and Antrop, 2004) or historical maps (e.g. Bender et al., 2005; Cousins, 2001; Nikodemus et al., 2005), but area-wide interpretations of these sources require manual mapping techniques or the digitisation of maps, which are time-consuming and costly and thus not feasible for large areas. Consequently, few studies have quantified long-term land-cover change at the landscape scale. Alternative data sets that provide information on former land-cover pattern in a landscape context may be derived from published statistics and census data (e.g. Brown et al., 2005; Fjellstad and Dramstad, 1999). However, these sources may vary in their spatial resolution and are not spatially explicit at finer scales.

In this study, we aimed at finding a workable approach to analyse land-cover patterns and dynamics for large areas at the highest possible resolution. Our study area was the Lahn-Dill Highlands, a marginal cultural landscape in Hesse, Germany. Here, information on historic land cover has been documented in agricultural statistics at the lowest administrative level, the rural district (Gemarkung), since 1955. At this scale, we investigated land-cover change by relating historic data from 1955 to current land cover derived from satellite images. The specific objectives were (i) to classify the rural districts according to land cover and its change between 1955 and 1995, and (ii) to characterise the derived types of land-cover patterns and dynamics with respect to physical attributes that are known to be conditional variables of land-cover change. The underlying hypothesis for our second objective was that land-use change occurred primarily in districts with relatively unfavourable physical conditions for agriculture.

4.2 Study area

Our study area, the Lahn-Dill Highlands, lies in the western part of Hesse, Germany and covers 1270 km² (Fig. 3). The actual delimitation of the area followed the requirements of an ongoing larger research project, which this study is part of (Frede and Bach, 1999; Hietel et al., 2005; Sheridan and Waldhardt, 2006). Our area covers the biogeographical region Gladenbacher Bergland (a name, which is frequently used synonymously for the Lahn-Dill Highlands) and all areas outside that region that belong to the Dill catchment (Klausing, 1988; Meynen and Schmithüsen, 1957). For this study, the borders were aligned to the borders of those rural districts that lie within this area.

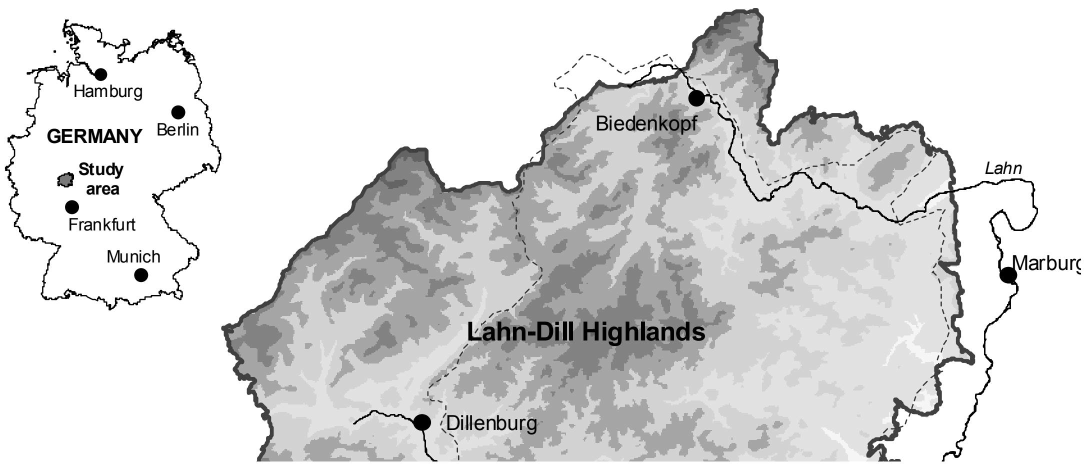


Fig. 3. Map of the study area showing its location in Germany and its topographical situation.

The Lahn-Dill Highlands represent the eastern ridge of the Rhenish Uplands (Rheinisches Schiefergebirge) and are mainly composed of clay schist, siliceous schist, and greywacke. The low-mountainous region, with altitudes of 150-670 m above sea level (a.s.l.), is typical of marginal cultural landscapes characterised by environmental conditions that are relatively unfavourable for cultivation (Frede and Bach, 1999). The mean annual temperature is between 6° and 8° C, and the mean annual precipitation ranges between 650 and 1100 mm. Soil types in the area form a characteristic small-scale mosaic of acidic shallow ranker soils and regosols on hill tops and upper slopes, cambisols and luvisols on slopes, and planosols and gleysols in alluvial plains (classification according to FAO (1998)).

Iron-ore mining, steel industry, and agriculture have shaped the cultural landscape of the Lahn-Dill Highlands for centuries (Kohl, 1978). As a result of the mining history and the high demand for charcoal in former times, large parts of the region are traditionally covered with forests. Only 32% of the Lahn-Dill Highlands are agricultural land, with 35% arable land, 51% grassland, and 14% fallow land (according to modified land-cover data of Nöhles (2000) as described in Section 4.3.2).

Agriculture has always been a matter of small-scale farming providing only a sideline income, whereas mining and steel industry have had an outstanding relevance as non-agricultural employment alternatives. Since the 1950s, the agricultural land-use pattern has changed considerably. Additional non-agricultural jobs within the region and in the adjoining Rhine-Main area as well as the introduction of the EC Common Agricultural Policy (CAP) have led to a substantial abandonment of business by many part-time farmers and a general decrease in farming activities throughout the region. In many districts of the study area, the formerly predominant arable crop production was largely replaced by extensive grassland use or abandoned fields (Kohl, 1978; Schulze-von Hanxleden, 1972).

However, the region's farmland still features a heterogeneous, small-parcelled land-use mosaic (field size ranging from less than 0.5 to 5 ha; Waldhardt et al., 2004). Present day cultivation and grassland management is carried out with a low input level of fertiliser and pesticides throughout the entire region. Large parts of the farmland are managed on the verge of profitability or even below it (Nowak, 1988). Due to the predominance of low-input management systems, the Lahn-Dill Highlands have a high biological diversity and thus are actually one of the most species-rich, low-mountainous regions in Germany (Nowak, 1988). This is specifically true for the plant species richness (Korsch, 1999; Nowak, 1992; Nowak and Wedra, 1988; Waldhardt and Otte, 2003).

4.3 Materials and methods

4.3.1 Historical land cover at the district scale

In order to derive information on historical land cover, we analysed agricultural statistics from the Hessian State Agency for Statistics (Hessisches Statistisches Landesamt, 1956). The data set provided the total area of arable land and grassland (comprising meadows, litter meadows (i.e. grasslands mown once a year for litter production), pastures, and rough pastures) in 1955 at the scale of rural districts. Rural districts, which are today part of larger municipalities, usually comprise one village and its agri- and silvicultural surroundings. They are the lowest administrative level and represent the basic spatial unit of the analysis. Due to the long common history of development within the villages, they reflect areas with homogenous economic and social conditions. In total, the study area comprises 192 rural districts with a mean size of about 6.6 km².

4.3.2 Current land cover

Information on the most pronounced aspect of land-use change, i.e. the proportion of old fields with woody plant succession, is not given in present and historical agricultural statistics. To obtain information on current land cover including these old fallows, we used an available satellite image interpretation (Landsat-TM, 25 m; Nöhles, 2000) from 1995. A total of seven land-cover classes were defined: arable land, grassland, deciduous forest, coniferous forest, waters, settlement, and fallow land. The latter corresponds to old fallows with shrub succession. The satellite-derived data set showed a relatively high proportion of isolated cells ($\leq$ 0.125 ha) for fallow land. Using ArcGIS 9, a 3 x 3 cells modal filter was performed to remove these cells (Burrough and McDonnell, 2000). The obtained result was validated by field data from 1997 within a test area (Fuhr-Bossdorf et al., 1999). To integrate satellite data into statistical analysis at the district scale, our basic spatial unit, we calculated the proportional cover of arable land, grassland, and fallow land for each district.

4.3.3 Physical landscape data

To characterise the districts with respect to the most important physical conditions for agriculture, we quantified elevation, slope, and soil moisture. These physical constraints were identified in several studies to be related to land-cover change (Bürgi and Turner, 2002; Chen et al., 2001; Hietel et al., 2004; Poudevigne et al., 1997).

Elevation data were obtained from a digital elevation model (DEM, 40 m; Hessische Verwaltung für Bodenmanagement und Geoinformation, undated). In order to correct errors, the data set was smoothed by a low-pass filter, i.e. the value for the cell at the centre of a 3 × 3 cells window was computed as a simple arithmetic average of the values of the other cells (Burrough and McDonnell, 2000). Furthermore, cell size was altered to 25 m × 25 m to adjust cell resolution. The value of the resized cell was estimated by bilinear interpolation resampling technique, which computes the new value by calculating the distance-weighted average from the four neighbouring cells (Burrough and McDonnell, 2000). From this information, we determined the median elevation expressed as metre a.s.l. within each district. Slope was calculated from the modified DEM by the use of the slope function in the ArcGIS 9 Spatial Analyst tool, which calculates the maximum rate of change of elevation between each cell and its eight neighbouring cells (Burrough and McDonnell, 2000). Within each district, we differentiated the proportion of slopes falling within the three classes ≤ 5°, 6-10°, and > 10°. Soil moisture was derived from the official digital soil map of Hesse (scale 1:50 000) combining information on available water capacity (AWC) in the root zone and the degree of soil wetness, i.e. gleyic and stagnic soil properties (Hessisches Landesamt für Umwelt und Geologie, 2002). In addition, slope was considered within the classification of soil moisture. Soil moisture was classified into four groups: dry (AWC < 50 mm or AWC 50-90 mm and > 5° slope), mesic (AWC > 90 mm or AWC 50-90 mm and ≤ 5° slope), moist (low to intermediate gleyic/ stagnic soil properties) and wet (high gleyic/ stagnic soil properties).

In addition to the physical landscape attributes, we applied the agricultural comparability index (Landwirtschaftliche Vergleichszahl, LVZ) to compare the districts. This index is used to classify German areas that are less favourable for agriculture. It aggregates natural characteristics of agricultural areas such as soil quality, climatic conditions, heterogeneity of soils, and water management problems and is based on a points system ranging from 0 to 100 for the best value. It is published by the Hessian State Ministry of the Environment, Rural Development and Consumer Protection (Hessisches Ministerium für Umwelt, ländlichen Raum und Verbraucherschutz, 2004) at the district scale.

4.3.4 Data analysis

In order to classify types of agricultural land-cover patterns and dynamics (TLPDs) at the district scale, we performed a k-means cluster analysis using STATISTICA 6.0 software (StatSoft, 2001). The purpose of the k-means clustering procedure is to classify objects with respect to optional quantitative traits into a user-specified number of clusters. To quantify recent land cover for each district, we calculated from the satellite data (i) the percentage of grassland in 1995 and (ii) the percentage of fallow land in 1995 with respect to total area of agricultural land. To obtain an estimate for land-cover change in each district, we assessed the arable land to grassland ratios for 1955 (from agricultural statistics) and 1995 (from satellite data) and calculated (iii) the difference between the two ratios. Since they have been continuously managed as grassland over the last century, alluvial plains were not considered for the calculation of the three variables. Their surface area was calculated from the intersection of DEM (slope $\leq 5°$), soil data (moisture wet), and satellite-derived land-cover data and was then subtracted as grassland from the total area of agricultural land for all calculations.

To improve statistical normality, the percentages of grassland and fallow land in 1995 were arcsine square root-transformed, and the ratios of arable land and grassland in 1995 and 1955 were log-transformed prior to analysis. According to the assumption that scaling of all variables must be similar, the data were standardised to z-scores. Five districts that were detected as ‘strong outliers’ (standard deviation > 3; McCune and Grace, 2002) were excluded from the analysis to reduce the undue influence on the outcome of the classification.

On the basis of the adjusted data set and with the help of k-means algorithm (implemented by MacQueen, 1967), the respective cluster means were calculated. The districts were allocated into different clusters by minimising the variability within clusters and maximising the variability between clusters. In order to find well-contrasted and compact clusters, the analysis was performed for different user-defined numbers of clusters ranging from 3 to 8. Preliminary classification results revealed that a small number of clusters resulted in a large variability within each cluster. A larger number of clusters, however, produced a strongly skewed distribution of districts with many small clusters consisting of very few districts. The best classification result was given by grouping the districts in six clusters, and thus these six clusters were chosen to represent the TLPDs.

Land-cover data and also physical landscape data were tested for differences between the defined TLPDs. Due to the non-normal distribution of some data sets, we chose the non-parametric Kruskal-Wallis one-way ANOVA by ranks as a powerful statistical method for the analysis of variance of such data (Legendre and Legendre, 1998). In case of significance, the analysis was followed by a Mann-Whitney U-test with Bonferroni correction for multiple testing ($p < 0.05$). The five detected outliers were not included in the tests.

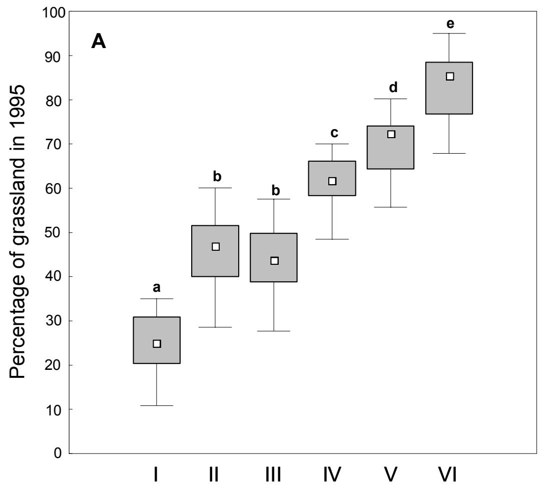
4.4 Results

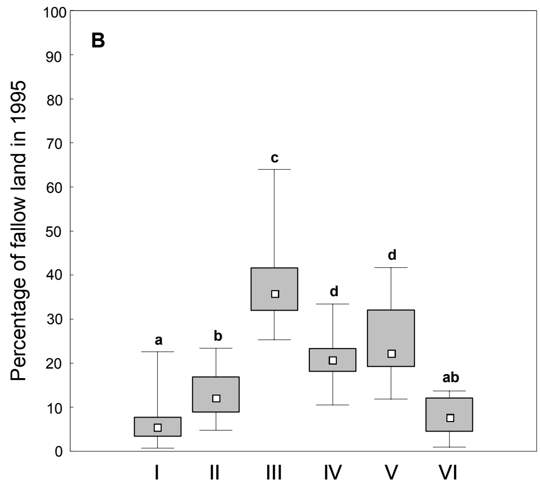
4.4.1 Types of land-cover patterns and dynamics (TLPDs)

The analysis of land-cover change revealed a dramatic decrease of arable land in favour of grassland and fallow land at the landscape scale, i.e. the entire study area. From 1955 to 1995, the median ratio of arable land to grassland changed from 66.6 : 33.4 to 38.5 : 61.5, respectively. In 1995, 14.2% of the total area of agricultural land outside the alluvial plains was fallow land.

Despite this general trend at the landscape scale, we detected pronounced differences at the district scale. With the aid of k-means cluster analysis, each district was assigned to one of six types of land-cover patterns and dynamics (TLPD I-VI). The statistical analysis of the land-cover variables used for classification showed significant differences between the TLPDs (Fig. 4). These types represent patterns of land cover and their dynamics between 1955 and 1995 (Table 1) that are to be found in different subregions of the study area (Fig. 5).

The districts of the easternmost part of the Lahn-Dill Highlands are grouped in TLPD I. They are characterised by a significantly low proportion of grassland (24.9%) and fallow land (5.5%) in 1995. The ratio of arable land to grassland remained almost unchanged from 1955 to 1995, i.e. the proportion of arable land is traditionally high. In contrast, the agricultural pattern of TLPD II-VI, covering 77.2% of the entire region, has been dynamic since 1955. The TLPD II and TLPD III showed a slight progressive change of the arable land to grassland ratio in favour of grassland. The districts of the predominant TLPD II, covering 26.6% of the landscape, still exhibit the traditional small-scale mosaic of arable land and grassland. TLPD III, with districts located in the central part of the Lahn-Dill Highlands, is in contrast characterised by a higher proportion of abandoned arable land (35.7%). The districts in the western part of the study area (TLPD IV-VI) are dominated by grassland. The proportion of grassland is above 50% in all three TLPDs, but it differs significantly between these types. Also, they show differences in land-cover change. The districts of TLPD IV in the mid-western part of the Lahn-Dill Highlands (22.8% of the entire region) experienced only a slight loss of arable land in favour of grassland (61.8%) and fallow land (20.7%). TLPD V, in comparison, significantly differs by a much stronger progressive change of land-cover patterns. These districts were formerly dominated by arable land, but have drastically changed over the last few decades to large proportions of grassland (72.4%) and fallow land (22.3%). The highest proportion of grassland, however, with a median of 85.4% is typical of TLPD VI in the westernmost part of the Lahn-Dill Highlands. Here, the proportion of grassland was already high in 1955, and land-cover change occurred only to a very limited extent.





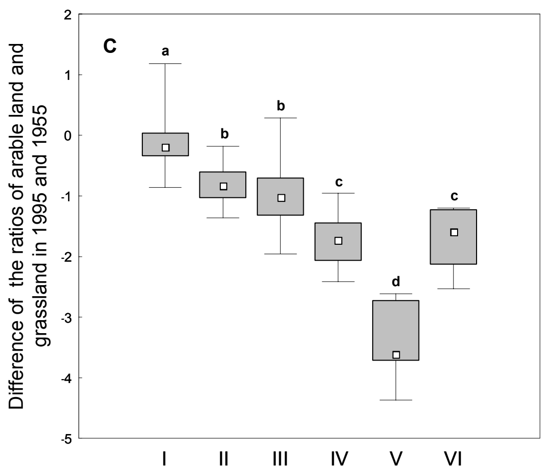


Fig. 4. Statistical analysis of land-cover data in relation to the types of land-cover patterns and dynamics (TLPD I-VI): A) Percentage of grassland in 1995, B) percentage of fallow land in 1995, and C) the difference of the ratios of arable land to grassland in 1995 and 1955. Values below zero indicate a decrease of arable land in favour of grassland. Significant differences between the TLPDs are indicated by different letters. Box plot: Median value (□), 25-75 % percentile (▭), minimum and maximum value (⊥).

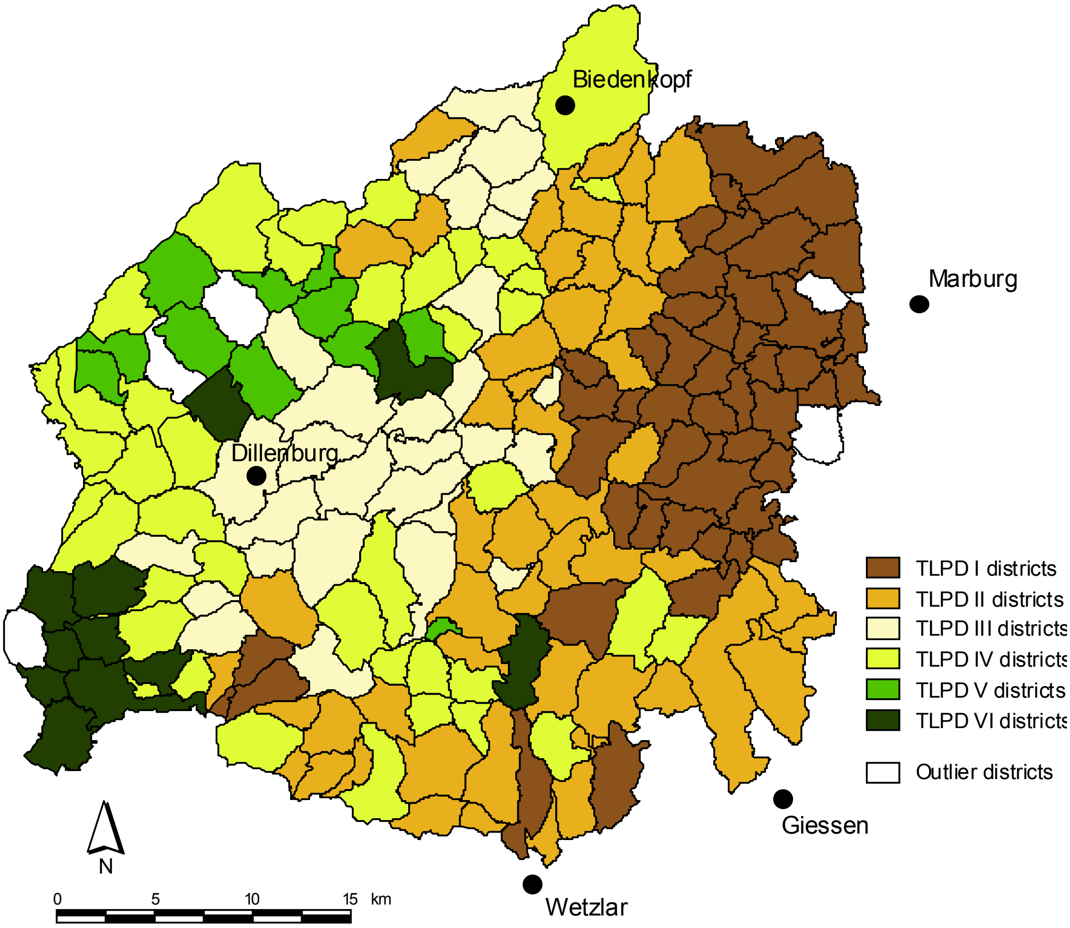


Fig. 5. Spatial distribution of the types of agricultural land-cover patterns and dynamics (TLPD I-VI) within the study area (see Table 1 for description of TLPDs).

Table 1. Description of the types of agricultural land-cover patterns and dynamics (TLPD I-VI) as derived from k-means cluster analysis at the district scale.

TLPD	Description	*n*	Area [km²]
I	Districts with low proportion of grassland and fallow land, stable	46	259
II	Districts with intermediate proportion of grassland and fallow land, slight progressive in favour of grassland	50	338
III	Districts with intermediate proportion of grassland and very high proportion of fallow land, slight progressive in favour of grassland	29	210
IV	Districts with high proportion of grassland and fallow land, progressive in favour of grassland	40	289
V	Districts with very high proportion of grassland and high proportion of fallow land, strong progressive in favour of grassland	11	65
VI	Districts with extremely high proportion of grassland and low to intermediate proportion of fallow land, progressive in favour of grassland	11	78
Outlier	Districts excluded from the analysis	5	30

n, number of districts.

4.4.2 Characterisation of TLPDs by physical landscape attributes

The TLPDs are characterised by physical landscape attributes as shown in Table 2. Comparing the 187 districts with respect to the classified TLPDs, the statistical analysis of the physical landscape attributes revealed significant differences.

Table 2. Statistical analysis of physical landscape attributes of the types of land-cover patterns and dynamics (TLPD I-VI).

		Elevation a.s.l. [m]		Slope > 10° [%]		Soil moisture 'dry' [%]		LVZ	
TLPD	*n*	Median	25-75% percentile	Median	25-75% percentile	Median	25-75% percentile	Median	25-75% percentile
I	46	252[a]	236-285	2.3[ae]	0.6-4.7	21.1[a]	13.1-27.7	27.5[a]	23.9-31.4
II	50	292[a]	230-336	4.3[ad]	1.9-10.3	29.8[b]	21.6-43.2	21.7[b]	16.0-25.1
III	29	354[b]	310-378	24.6[b]	16.7-33.6	58.4[c]	42.5-66.8	11.7[c]	8.2-14.8
IV	40	360[b]	281-413	11.4[c]	5.3-17.9	50.4[c]	37.9-57.3	13.7[c]	10.5-16.7
V	11	363[bc]	352-392	18.4[bcd]	4.3-22.4	41.9[abc]	34.4-58.7	12.8[c]	10.0-15.8
VI	11	522[c]	469-561	0.7[e]	0-2.8	2.4[d]	0-10.8	15.6[bc]	10.7-20.4

n, number of districts within groups; LVZ, agricultural comparability index (Landwirtschaftliche Vergleichszahl); significant differences are indicated by different letters.

The TLPD I districts, dominated by arable land, show favourable physical settings for cultivation. Generally, we found low elevations (median value 252 m), a low proportion of steep slopes (2.3%), and an intermediate proportion of dry soil conditions (21.1%). The districts are comparatively highly rated with a median LVZ of 27.5. TLPD II, characterised by an intermediate proportion of grassland and fallow land, significantly differs from TLPD I due to a lower LVZ (21.7) and a higher proportion of dry soils (29.8%). Similar to TLPD I, the median elevation (292 m) and proportion of steep slopes (4.3%) is low. The districts of TLPD III are characterised by very high proportions of fallow land and steep slopes. In 75% of these districts, at least 16% of the agricultural land is on steep slopes. Accordingly, rather unfavourable conditions for agricultural land use are given, also due to the corresponding high proportion of dry soils (58.4%). Hence, TLPD III was ranked with the lowest LVZ (11.7) of all types. TLPD IV is, similar to TLPD III, characterised by an intermediate elevation (360 m), a very high proportion of dry soil conditions (50.4%), and a low LVZ (13.7). However, in contrast to TLPD III, the proportion of steep slopes is significantly lower (11.4%). The analysis of physical attributes of TLPD V, which has experienced the most pronounced increase in grassland since 1955, revealed a large range of the proportion of steep slopes (4.3-22.4%) and dry soil conditions (34.4-58.7%). The grassland dominated districts of TLPD VI have low proportions of steep slopes (0.7%) and dry soils (2.4%) and thus seem to offer even better physical conditions for cultivation than TLPD I. However, due to their higher elevation (522 m) they are exposed to climatic constraints for crop production, which is expressed in a comparatively low LVZ (15.6).

4.5 Discussion

4.5.1 Types of land-cover patterns and dynamics

The results of our analysis confirmed a general trend of decreasing arable land and increasing grassland and fallow land for the entire study area. This development has also been reported from other marginal European cultural landscapes within the last few decades (Baldock et al., 1996; MacDonald et al., 2000). Despite this general trend of abandonment of cultivation at the landscape scale, the land-cover pattern reflects a high temporal and spatial variation at the local scale as has been described also from, e.g. France (Poudevigne and Alard, 1997), Sweden (Skanes and Bunce, 1997), Norway (Fjellstad and Dramstad, 1999), the Czech Republic (Lipsky, 1995), and Austria (Krausmann et al., 2003).

In the years immediately following World War II, the lack of food led to relatively high proportions of arable land within the entire study area since even poor soils on steep slopes were cultivated (Schulze-von Hanxleden, 1972). Thus, there were only slight differences in the composition of the land-cover patterns between the districts in 1955. Between 1955 and 1995, economic prosperity and increasing mechanisation, intensification and specialisation of agriculture led to dramatic changes. Cultivation on unfavourable sites has ceased, and former fields were either turned into grassland or were completely abandoned (Fuhr-Bossdorf et al., 1999; Schulze-von Hanxleden, 1972). Accordingly, we found a low proportion of arable land and a higher proportion of fallow land in districts with steep slopes and dry soils in 1995. This result is in line with a study by Hietel et al. (2004) who found in an investigation at the patch scale that fallow land in our study area is situated mainly on elevated, steep sites with sandy soils and low available water capacity. Further, the study area exhibits a distinct gradient of grassland distribution, which is well in accordance with the pattern of elevation at the district scale. The proportion of grassland is low in the eastern basin areas and very high in the more elevated south-western parts. The spatial patterns of environmental variables considered in this study, i.e. slope, soil moisture, and elevation, are thus congruent with land-cover patterns and the dynamics of agricultural activities at the district scale. Our underlying hypothesis for this study was thus confirmed. The LVZ incorporates the most important environmental variables and gives a consistent ranking order from favourable (high proportion of arable land) to moderate (high proportion of grassland) and unfavourable conditions (high proportion of fallow land) for cultivation.

However, spatial variations in the magnitude and direction of land-cover change at the district scale may not be explained by physical constraints alone. TLPD V in our classification is characterised by the largest increase in grassland and a large loss of arable land. The overall conditions for agriculture are rather unfavourable (low LVZ), but the districts of this type show a broad range and no distinct pattern of important physical attributes (slope, soil moisture). As socioeconomic factors also have an essential influence on land cover (e.g. Baudry, 1993; de Koning et al., 1998), this result suggests that in TLPD V socioeconomic variables are more important than physical constraints. Remarkably, the districts in TLPD III, with a similar low LVZ as in TLPD V, showed the highest proportion of fallow land in 1995, which also points to a massive loss of arable land. The striking difference in the direction of land-cover change from arable land to grassland in TLPD V and arable land to fallow land in TLPD III is also very likely owed to complex socioeconomic reasons. For the study area, Hietel et al. (2005) analysed relations between numerous socioeconomic variables and land-cover patterns and dynamics. Their study was based on land-cover data of seven districts in the Lahn-Dill Highlands between 1945 and 1999. High proportions of arable land were related to inheritance traditions, high leasehold rents, a large number of farms, and a high agricultural employment rate. High proportions of grassland were, in contrast, indicated by variables characterising urbanisation (e.g. high employment in industry and population density, favourable traffic infrastructure). Fallow land was indicated by variables indicating a minor importance of agriculture (e.g. high proportion of commuters and part-time occupation of farms). Although Hietel et al. (2005) included comprehensive socioeconomic aspects, the analysed variables were not causal drivers of land-cover change, but they do interact with these changes and often result from them. Due to the multiple interactions and intercorrelations between land cover and environmental and socioeconomic variables (Van der Veen and Otter, 2001; Hietel et al. 2005), true effects of socioeconomic 'drivers' of land-cover change are difficult to distinguish. Another serious problem associated with the analysis of socioeconomic variables in entire regions is the deficiency of available data with an appropriate spatial resolution.

4.5.2 Database and methodology

The quality and outcome of the identified and characterised types of land-cover patterns and dynamics depend on the thematic and spatial resolution of the databases and the applied classification method.

When analysing land-cover data from different sources, a prior equalisation of their thematic content and spatial entity is an essential requirement that must be taken with some precautions. Data equalisation always involves a loss of information due to spatial and thematic aggregation of the sources to the smallest common denominator (Petit and Lambin, 2002). Since satellite-derived data and agricultural statistics are collected in a very different way (i.e. computer-assisted techniques versus census data), land-cover is categorised differently. In our study, the detailed information on grassland given in the agricultural statistics was aggregated into the main category ‘grassland’, which accords to the lowest common class of the satellite-derived data.

Furthermore, simplifications had to be made concerning the spatial resolution of the used data sets. Whereas satellite-derived data are available at a fine resolution (i.e. raster cell), land-cover statistics are published at the comparatively coarse resolution of administrative levels (i.e. the rural district). In order to allow for a spatially differentiated comparison of both data sets at the highest possible resolution, we reduced the satellite data information to its mere compositional components at the district scale, i.e. we calculated proportions of land-cover types. Thus, the aim of an area-wide landscape classification was achieved at the expense of spatial resolution. High-resolution spatially explicit information on the configuration of different land cover within the districts was lost.

Naturally, the outcome of such classifications is influenced by the applied statistical methods. There are many possible ways to classify a landscape depending on the nature of input data and scales. The use of k-means cluster analysis has proven to be a simple and workable approach to classify patterns of recent and historic land cover for large areas at the district scale. Multivariate analyses such as clustering are common practice in landscape ecology (Bernert et al., 1997; Bunce et al., 1996). The use of the k-means cluster procedure is gaining interest in many scientific fields because of its simplicity and rapidity (Estivill-Castro and Yang, 2004) and has been successfully applied in landscape ecology for the definition of homogeneous land units and temporal dynamics (Hietel et al., 2004; Simmering et al., 2006).

4.5.3 Conclusions

We conclude that the combination of remote sensing data with agricultural statistics is suitable to identify patterns of current land cover and land-cover dynamics at the landscape scale. We found a general trend of abandonment of cultivation at the landscape scale, which is governed by significant differences between the districts. We classified six types of land-cover patterns and dynamics at the district scale and characterised their physical settings.

Our simple landscape-oriented approach provides a database that may be useful in a variety of contexts. As proposed by Marcucci (2000), the consideration of landscape history might be a useful tool in landscape planning and resource management to develop effective concepts for future landscape management. For studies in landscape research, derived types of land-cover patterns and dynamics may be used as strata in sampling designs that aim to include land-cover dynamics as a factor of interest. The identification and characterisation of spatially and temporally heterogeneous land-cover patterns in marginal cultural landscapes will further support research that aims to outline areas potentially sensitive to future land-cover changes.

Acknowledgements

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5. Assessing the spatial distribution of grassland age in a marginal European landscape

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Abstract

Grassland age is increasingly recognised to be an indicator for present-day biodiversity, e.g. plant species richness, and is also important for other landscape functions. We developed a methodological approach to systematically assess the spatial distribution of grassland age in marginal European landscapes. This approach - applied to the Lahn-Dill Highlands (1270 km²), a marginal landscape in Hesse, Germany - comprises three steps: (1) In a two-stage stratification process, we pre-stratified the study area according to recent land-cover patterns and their changes between 1955 and 1995 (stratification I) and classified grassland types by combining data on soil moisture, base-richness, and elevation (stratification II). From 50 grassland types, we randomly selected 1000 representative grassland patches. (2) We determined the age of these patches by means of aerial photograph interpretation of a chronosequence dating back to 1953 and classified each patch with respect to the age classes young (<18 years), mid-aged (18-47 years), and old (>47 years). (3) Based on this information, we calculated grassland type-specific probabilities for grassland patches to belong to the respective age classes. These probabilities were projected to districts by direct extrapolation. An exemplary validation of extrapolation results for two test areas was performed. The results revealed that 49% of the investigated patches were old grassland. The remaining patches were mid-aged (36%) or young grassland (15%). The extrapolation results indicated accordingly a predominance of old grassland at the district scale. Occurrences of mid-aged grassland were concentrated in districts with a pronounced land-cover change, whereas young grassland is apparently evenly distributed across the study area. Validation results suggest that our approach is suitable for a realistic estimation of grassland age in marginal European landscapes. The method may be applied in landscape models of various disciplines that rely on large-scale information on grassland age.

Keywords

Land-cover change; Landscape change; Marginal agricultural landscape; Stratified random sampling; Aerial photographs; GIS; Spatial extrapolation; Germany

5.1 Introduction

Since the end of World War II, many marginal European landscapes have experienced severe land-use changes. Several socioeconomic factors like unfavourable agricultural structures, changing labour markets, relative prices for agricultural products, agricultural policies, migration, and infrastructure developments were identified as important driving forces of land-use change in these landscapes (Baldock et al., 1996; MacDonald et al., 2000; Strijker, 2005). However, most marginal agricultural landscapes are characterised by abiotic constraints, such as less-favoured topographic, edaphic, and climatic conditions for cultivation, which are the main obstacle for modern agricultural development (cf. Frede et al., 1999; MacDonald et al., 2000).

In many marginal landscapes, large portions of arable land have been consecutively abandoned in favour of grassland (Baldock et al., 1996; MacDonald et al., 2000). As a consequence of this successive land-use change, the current landscape pattern consists of a large number of grassland patches that differ in age, i.e. in the duration of grassland management after cessation of arable farming. The increase in grassland, its driving forces, and the potentially large impacts on ecological functions and processes are thus important topics in integrative landscape research (cf. Turner et al., 2001; Wu and Hobbs, 2002).

Recent studies focussed on the influence of grassland age on soil properties like soil carbon and nitrogen content or pH (e.g. Breuer et al., 2006; McLauchlan et al., 2006). Further, grassland age may also have an impact on faunal species richness (e.g. Balmer and Erhardt, 2000; Dauber and Wolters, 2005; Purtauf et al., 2004) or the genetic structure of arthropod (Holzhauer et al. 2006) and plant populations (Prentice et al., 2006). In particular, plant species richness and species composition depend on grassland age (e.g. Austrheim and Olsson, 1999; Bruun et al., 2001; Cousins and Eriksson, 2002; Ejrnæs and Bruun, 1995; Pärtel and Zobel, 1999; Waldhardt and Otte, 2003). From Austrheim and Olsson (1999) and Waldhardt and Otte (2003) it may be concluded that land-use changes in the last five decades are highly relevant for plant species diversity in grasslands. They found that plant species diversity of grassland patches increased with grassland age and several species occurred exclusively in certain successional grassland stages. Moreover, the spatial distribution of these successional stages, i.e. the spatial distribution of grassland age, was found to enhance species diversity at the landscape scale. Hence, vegetation and landscape ecologists have increasingly recognised that grassland age is an important indicator for plant species diversity (e.g. Luoto et al., 2002; Norderhaug et al., 2000; Waldhardt et al., 2003).

Despite the obvious importance of grassland age for various ecological functions and processes, there have been, to our knowledge, no attempts to reproduce and assess grassland age at larger spatial scales. A reason might be that quantifying grassland age in a large-scale context is considered to require area-wide, spatially explicit, high-resolution data on land-cover[1] change for at least several decades. Several studies indicate that aerial photographs are a powerful tool to detect land-cover changes in the landscape (e.g. de Blois et al., 2001; Ihse, 1995; Pan et al., 1999; Ruuska and Helenius, 1996). An area-wide interpretation of aerial photographs is yet time-consuming and costly and thus not feasible for large regions.

Given this background, our objective was to develop a methodological approach to systematically assess the spatial distribution of grassland age in a marginal European landscape. Our approach is based on a representative selection of a large number of grassland patches from regionally differentiated grassland types, applicable at several spatial scales (i.e. from patches to districts and landscapes of several hundred square kilometres), and covers the last five decades.

5.2 Materials and methods

5.2.1 Study area

The Lahn-Dill Highlands cover a total area of about 1270 km² in the western part of the state of Hesse, Germany (Fig. 6) and are a typical marginal agricultural landscape characterised by unfavourable conditions for cultivation (Frede and Bach, 1999). The low-mountainous landscape features altitudes between 150 and 670 m above sea level (a.s.l.) and slopes of up to about 20°. The small-scale mosaic of soil types comprises a relatively high amount of poor soils such as acidic shallow ranker soils and regosols. The climatic conditions are rough and rather damp (mean annual temperature: 6-8° C, mean annual precipitation: 650-1100 mm).

[1] Following the definitions of Turner and Meyer (1994), we use the term land cover to refer to the physical state of the land, whereas land use denotes the human employment of the land.

The agrarian structure of the Lahn-Dill Highlands is dominated by small farm sizes (mean farm size 14 ha; Waldhardt and Otte, 2003) and a heterogeneous, small-parcelled mosaic of arable fields, grasslands, and fallow lands (mean field size of about 0.4 ha; cf. Simmering et al., 2006; Waldhardt et al., 2004). The entire study area is included in the less-favoured area support scheme since 1976 (EC Regulation No 75/268).

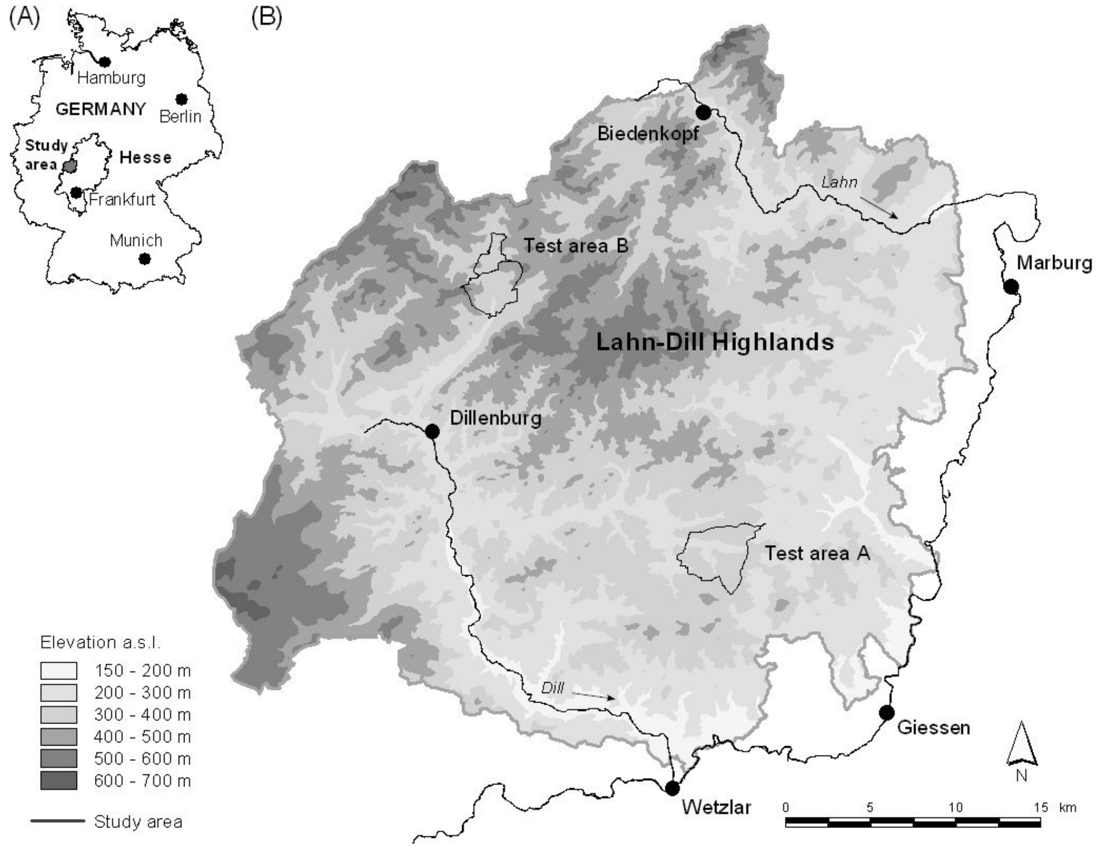

Fig. 6. Map of the study area showing (A) its location in Hesse, Germany and (B) its topographical situation. Test area A: Erda; test area B: Steinbrücken and Eibelshausen.

Agriculture in the Lahn-Dill Highlands always was a matter of small-scale farming providing only a sideline income, while mining and steel industry had an outstanding relevance as non-agricultural employment alternatives. Since the 1950s, the agricultural land-use pattern of the Lahn-Dill Highlands has changed considerably. Additional non-agricultural jobs within the region and in the adjoining Rhine-Main area as well as the introduction of the EC Common Agricultural Policy (CAP) led to a substantial abandonment of business by many part-time farmers and a general decrease in farming activities throughout the region. In large parts of the study area, extensive grassland use has replaced the formerly predominant crop production (Hietel et al., 2004, 2005; Kohl, 1978; Schulze-von Hanxleden, 1972). Today, about 51% of the agricultural land (about 405 km²; according to land-cover data by Nöhles (2000)) are grasslands managed at low intensity ranging from grazing without fertiliser application to mowing three times a year for fodder production (Wellstein et al., 2007). Owing to the predominance of extensive farming systems, the Lahn-Dill Highlands feature a high biological diversity and thus are one of the most species-rich, low-mountainous landscapes in Germany (Nowak, 1988). This is specifically true for the plant species diversity of grasslands (Nowak, 1992; Simmering et al., 2006; Wellstein et al., 2007).

5.2.2 Methodological approach

Our GIS-based methodology permits to systematically assess the spatial distribution of grassland age in a marginal agricultural landscape and involves three major steps: (1) A two-stage stratified random selection of grassland patches, (2) a multitemporal aerial photograph interpretation of the selected patches, and (3) the spatial extrapolation of grassland age (Fig. 7). Supplementary to our approach, we performed a validation procedure using reference data for two test areas (Fuhr-Bossdorf et al., 1999). All spatial data were processed with the ESRI Inc. software package ArcGIS 9.0 and the Spatial Analyst extension.

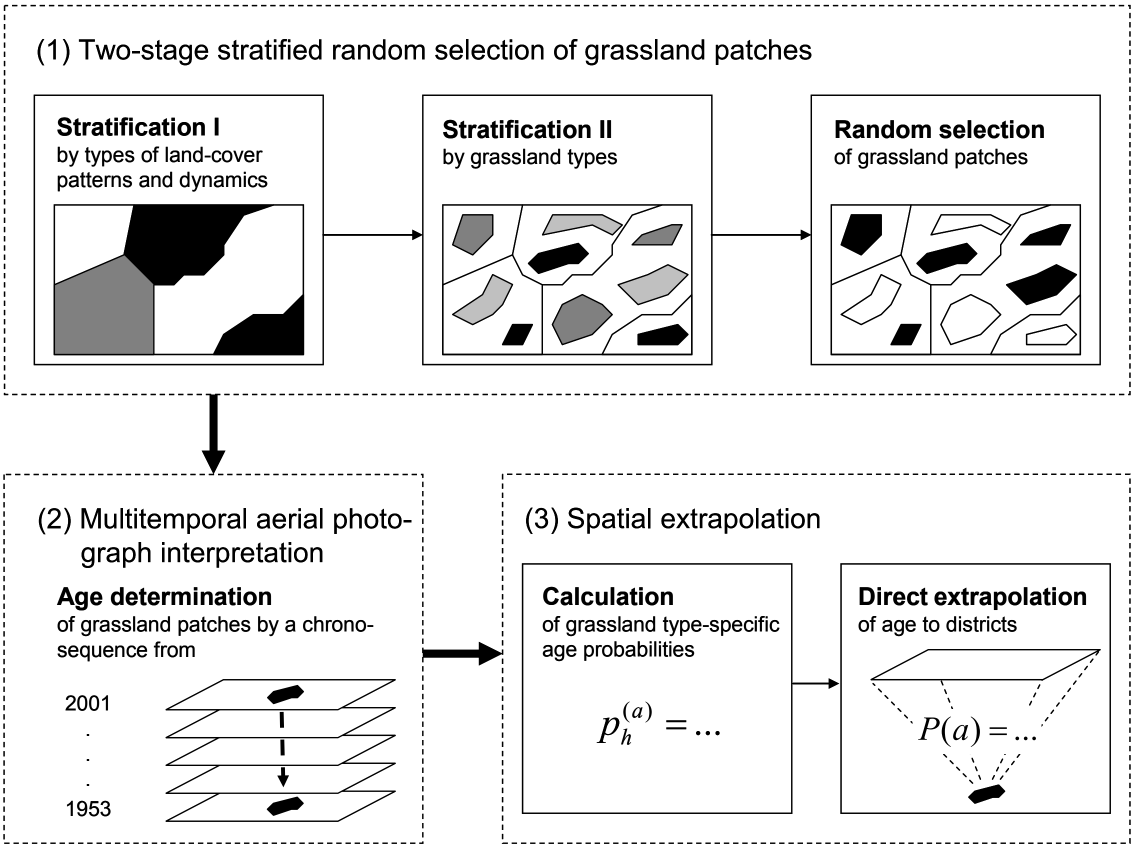

Fig. 7. Schematic workflow of the methodological approach to assess the spatial distribution of grassland age in a marginal landscape.

Two-stage stratified random selection of grassland patches

The two-stage stratification process performed in our analysis (Fig. 7) is based on the assumption that the probability of a grassland patch to have a certain age is affected by socioeconomic variables and physical attributes (cf. Section 5.1) at different spatial scales. On an intermediate spatial scale (stratification I), the administrative unit ‘Gemarkung’ (district), which traditionally represents the smallest political and socioeconomic entity in Germany, we often find homogeneous socioeconomic characteristics depending on, for example, local traditions, the predominance of independent or ‘follow-the-leader’ mentalities, and the pace of spread of innovation. These strongly affect the probability of a patch to be recently managed as either arable field, grassland, or fallow (Hietel et al., 2005, 2007). In order to systematically consider the close connections between the socioeconomic environment and land use, we pre-stratified the entire study area according to recent land-cover patterns and their changes between 1955 and 1995. The data sets were derived from agricultural statistics from 1955 (Hessisches Statistisches Landesamt, 1956) for 187 districts of the study area (five more districts were identified as outliers in previous analysis (cf. Reger et al., 2007) and not considered) and a 1995 satellite image interpretation (Landsat-TM, 25 m raster; Nöhles, 2000). At the scale of the districts, we calculated (1) the percentage of grassland in 1995 and (2) the percentage of fallow land in 1995 with respect to the total area of agricultural land. We considered the percentage of grassland and fallow land since cessation of arable farming favoured these land-cover types in the study area within the investigated time period. To obtain an integrated estimate for land-cover changes, we further assessed (3) the arable land to grassland ratios for 1955 and 1995 and calculated the difference between the two ratios for each district. By means of a k-means cluster analysis based on the three input variables, we identified and localised six types

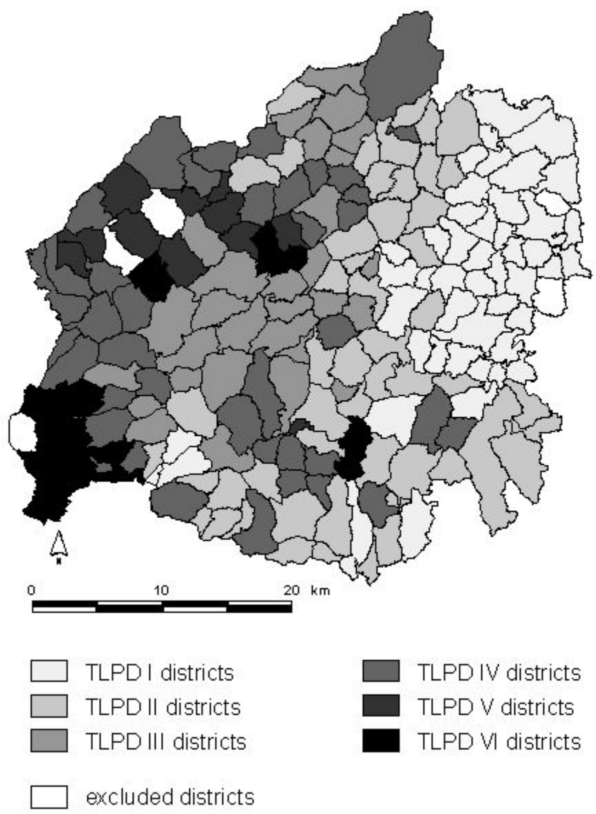

Fig. 8. Spatial distribution of the six TLPDs within the study area (after Reger et al., 2007; see Table 3 for description of TLPDs).

of land-cover patterns and dynamics (TLPDs; Reger et al., 2007) that represent patterns of present-day agricultural land cover and past land-cover dynamics between 1955 and 1995 at the district scale (Fig. 8). A description of the TLPDs is given in Table 3. For more details see Reger et al. (2007).

Table 3. TLPDs and land-cover variables used for the classification of the TLPDs (see Reger et al., 2007). Grassland and fallow land are calculated as the median proportions of agricultural land in the districts. n = number of districts.

TLPD	n	Total area	Grassland 1995		Fallow land 1995		Change in arable land to grassland ratio 1955 – 1995*	
		[km^2]	Median [%]	25 – 75% percentile	Median [%]	25 – 75% percentile	Median	25 – 75% percentile
I	46	259	24.9	20.4 – 30.9	5.5	3.4 – 7.7	-0.20	-0.34 – 0.03
II	50	338	46.8	40.0 – 51.6	12.2	8.9 – 16.9	-0.83	-1.03 – -0.61
III	29	210	43.7	38.8 – 49.8	35.7	32.0 – 41.6	-1.02	-1.32 – -0.71
IV	40	289	61.8	58.4 – 66.1	20.7	18.2 – 23.4	-1.74	-2.06 – -1.45
V	11	65	72.4	64.4 – 74.1	22.3	19.3 – 32.1	-3.61	-3.71 – -2.73
VI	11	78	85.4	76.8 – 88.5	7.6	4.5 – 12.1	-1.59	-2.13 – -1.23

* Values below zero indicate a decrease of arable land in favour of grassland.

On a small spatial scale (stratification II), i.e. at the scale of patches, physical attributes like soil moisture, base-richness, and elevation are the most relevant variables that influence land-use decisions by farmers (e.g. Bürgi and Turner, 2002; Hietel et al., 2004; Pan et al., 1999). Particularly in marginal landscapes, which are often characterised by a heterogeneous topography and a broad range of different soils, these physical attributes are important variables for grassland age as they strongly affect the potential for agricultural land use (Hietel et al., 2004). For the second stratification, we classified grassland types within the identified TLPDs. The localisation of recent grassland was obtained from the satellite-derived land-cover map from 1995 (Landsat-TM, 25 m raster; Nöhles, 2000). Users' accuracy for grassland was 86.7% (Nöhles, 2000). We classified and combined soil moisture, base-richness, and elevation obtained from digital soil maps and DEM to derive physical attributes (Table 4) according to their suitability and constraints for different types of agricultural land use.

The intersection of physical attributes with TLPDs led to a total of 118 grassland types. Grassland types of alluvial plains (soil moisture: wet, n = 13 grassland types; cf. Table 4) were excluded from further analysis, since they were continuously managed as grasslands over the last half century and not subject to land-cover change (Hietel et al., 2004). Additionally, grassland types (n = 55), covering less than 5% of the grassland area within each TLPD, were omitted from the analysis. Thus, the final grid data set consisted of 50 grassland types, which were included in the stratified random selection of grassland patches.

Table 4. Variables and classes used for stratification II.

Variable	Class	Description
Soil moisture[a, b]	1 dry	AWC[c] <50 mm or AWC4 50-90 mm and >5° slope
	2 mesic	AWC[c] >90 mm or AWC4 50-90 mm and ≤5° slope
	3 moist	Low to mean gleyic/ stagnic soil properties
	4 wet	High gleyic/ stagnic soil properties
Base-richness[b]	1 base-poor	Soil types with acidic substrate (e.g. sandstone to claystone, quartzite, schist)
	2 moderate	Soil types with neutral substrate (e.g. siliciclastic sedimentary rocks, metamorphic rocks)
	3 base-rich	Soil types with alkaline substrate (e.g. basaltic extrusive rocks)
	4 calcareous	Soil types with calcareous substrate (e.g. limestone, dolomite)
Elevation[b]	1 colline	≤400 m a.s.l.
	2 submontane	>400 m a.s.l.

[a] Information derived from official digital soil map of Hesse (scale 1:50 000), HLUG (Hessisches Landesamt für Umwelt und Geologie); [b] Information derived from digital elevation model (40 m raster), HVBG (Hessische Verwaltung für Bodenmanagement und Geoinformation); [c] Available Water Capacity in the root zone

Subsequently, grassland types were assigned to polygons derived from digital cadastral maps from 2004 (scale 1:5000). The shape and size of the cadastral polygons represent land parcels, which are the smallest spatial unit of uniform land-cover development since 1945 (Herzog et al., 2001; Hietel et al., 2004). Using GIS, we randomly selected an equal sample size of 20 polygons per grassland type to ensure balanced representation of each grassland type. Hence, a total of 1000 scattered grassland patches with a mean size of about 0.3 ha were sampled for the survey of land-cover change.

Multitemporal aerial photograph interpretation

In order to assess the duration of grassland use for the sampled patches, we reconstructed the land-cover history of each patch by visual multitemporal aerial photograph interpretation (Fig. 7). For the entire study area, a chronosequence of black and white aerial photographs since 1953 was available (mainly at a scale of 1:12 000 covering an area of about 4 km², 1:24 000 for 1953). Recent grassland use was differentiated from former land-cover types (i.e. arable land or fallow land with woody plant succession) according to tonal contrast and texture. However, with respect to the entire study area the aerial surveys cover different time intervals. Therefore, we had to consider periods of several years in our investigation. We started with photographs from the period 1998-2001, since these were the most recent photographs available and continued with photos from the periods 1989-1994, 1979-1983, 1967-1973, 1959-1962, and 1953. The time span between the photographs was approximately 10 years. Moving back in time until the land cover changed, the interpretation of the time series permitted to assign each sampled grassland patch to either the age class young (<18 years), mid-aged (18-47 years), or old (>47 years), which are expected to be ecological relevant stages for grasslands (cf. Austrheim and Olsson, 1999; Waldhardt and Otte, 2003). The resulting age classes of the grassland patches were tested for significant differences between the TLPDs, soil moisture, base-richness, and elevation classes by performing G-tests. The G-test is equivalent to the more commonly used chi-square test, but is computationally simpler and G appears to follow the chi-square distribution a bit more closely (Sokal and Rohlf, 2004).

Spatial extrapolation

The identified age of the grassland patches was used to perform a spatial extrapolation (Fig. 7). Our approach is based on the assumption that grassland patches of the same grassland type - patches with comparable physical attributes located in areas with similar land-cover patterns and dynamics - have the same probability to belong to a certain age class since we expect that farmers' decisions are driven by similar physical attributes and socioeconomic conditions. Based on the three age classes and the 50 grassland types, we calculated grassland type-specific age probabilities $p_h^{(a)}$, defined as

$$p_h^{(a)} = \frac{c_h^{(a)}}{n_h} \tag{1}$$

where $c_h^{(a)}$ is the number of investigated patches found to belong to age class *a* of grassland type *h*, and n_h is the number of investigated patches of grassland type *h*, in our case n_h = 20.

In order to determine the areal proportions of grassland age classes at the scale of districts we used direct extrapolation. For each district, we first weighted the area A_h covered by grassland type h by the probability $p_h^{(a)}$ (Equation 1) and summed over all grassland types $h = 1, 2, \ldots, 50$. Dividing this sum by the total grassland area of the corresponding district, we obtained the proportions of the age classes young, mid-aged, and old in the district. The areal percentage $P(a)$ of age class a for each district is thus calculated by the formula

$$P(a) = \frac{\sum p_h^{(a)} \cdot A_h}{\sum A_h} \cdot 100 \qquad [\%] \qquad (2)$$

Validation procedure based on reference data

In order to judge if our results permit a realistic estimation of the grassland age for single districts, we performed an exemplary validation using reference data. The estimated proportions of grassland belonging to the respective age classes were validated for two test areas with contrasting land-cover dynamics (Fig. 6B). Test area A, the district Erda (11.6 km²), represents TLPD I (Table 3) with little land-cover dynamics, whereas test area B, the neighbouring districts Steinbrücken and Eibelshausen (9.3 km²), belongs to TLPD V (Table 3), which is characterised by a strong land-cover change in favour of grassland. Results of an interpretation of black and white aerial photographs from the years 1953, 1961, 1972, 1979, and 1989 and field data from 1996 covering the entire surface of agricultural land (Fuhr-Bossdorf et al., 1999) provided independent information on the age of the entire grassland in the two areas. We grouped the data from this study into the age classes young (<18 years), mid-aged (18-47 years), and old (>47 years), calculated their areal proportions, and compared these with the estimated proportions from the study. To test for significant differences between the extrapolation results and the reference data, a G-test was performed.

5.3 Results

5.3.1 Age structure in TLPDs and physical attributes of the investigated patches

In total, 49% of the 1000 investigated grassland patches were old or permanent grassland stands. The remaining patches were young (15%) and mid-aged grasslands (36%), almost all of which were formerly used as arable land. Pronounced differences of the grassland age structure, i.e. the proportions of grassland patches belonging to the age classes, were observed between the six TLPDs and confirmed by statistical analysis (G-test, $G_{adj} = 40.09$, $df = 10$, $p < 0.001$; Fig. 9A). For TLPD I, which is characterised by a low proportion of grassland and an almost unchanged ratio of arable land to grassland from 1955 to 1995, our analysis indicated that 61% of the investigated grassland patches were old. Also in TLPD II-IV, old grassland patches predominated with a comparatively high proportion (53-57%). The grassland age structure of TLPD V differed significantly from TLPD I-III and was characterised by a proportion of old grassland patches well below the average. With a severe decrease of arable land in favour of grassland since 1955 in TLPD V, only 30% of the investigated patches in TLPD V were old. Mid-aged grassland patches showed a relatively high proportion of about 56%. The highest proportion of young grassland of all types was found with 22% in TLPD II and VI.

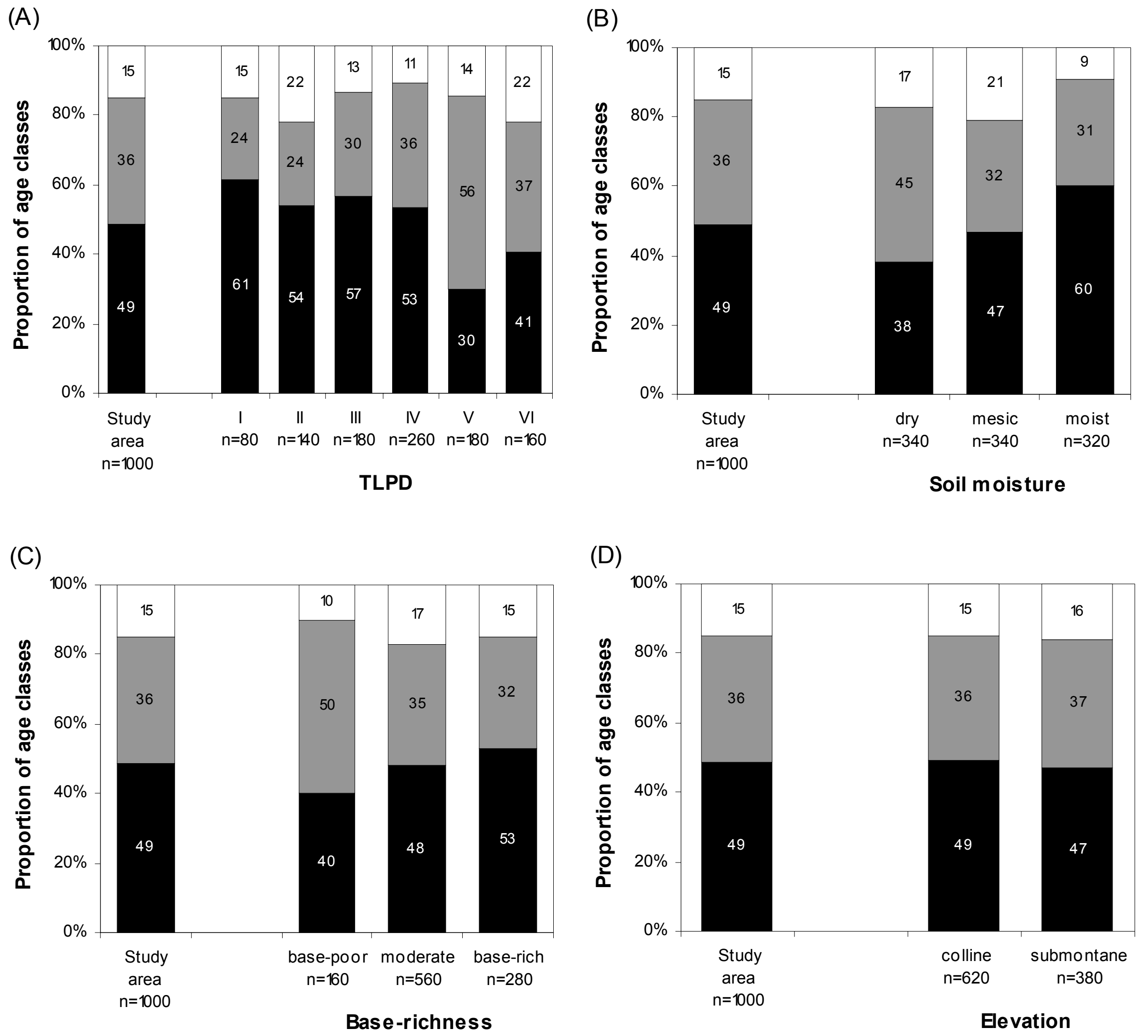

Fig. 9. Proportion of young (▭), mid-aged (▩), and old (■) grassland patches calculated for (A) the six TLPDs (G-test, $G_{adj} = 40.09$, $df = 10$, $p < 0.001$) and the classes of (B) soil moisture (G-test, $G_{adj} = 13.26$, $df = 4$, $p < 0.05$), (C) base-richness, and (D) elevation. n = number of grassland patches.

Statistical analysis of the frequency of age classes among classes of physical attributes revealed that grassland age structure is independent from base-richness and elevation, but not from soil moisture (G-test, $G_{adj} = 13.26$, $df = 4$, $p < 0.05$). Moist grassland stands were predominantly permanent grassland (60%), since moist sites are less suitable for arable farming (Fig. 9B). Some of the moist grassland types comprised almost exclusively old grassland patches. In contrast, dry sites featured only 38% old grassland. Hence, more than 60% of dry grassland were subject to land-cover changes in the respective time period. Base-poor grassland patches also tended to be predominantly mid-aged (Fig. 9C). However, moderate and base-rich sites accounted for 48% and 53% of old grassland patches. The proportions of young, mid-aged, and old grassland patches per elevation class were nearly identical (Fig. 9D).

5.3.2 Extrapolation of grassland age at the district scale

Based on the age of the investigated grassland patches, we calculated a total of 150 grassland type-specific age probabilities for grassland patches (Appendix 1). Considering all 187 districts, these probabilities were extrapolated to the districts (Equation 2). The extrapolation results were mapped separately for the grassland age classes old, mid-aged, and young (Fig. 10). Very high proportions of old grassland stands (> 75%) were calculated for 31 districts mainly located in the eastern part of the Lahn-Dill Highlands (Fig. 10A). In large parts, mostly in the centre of the study area (n = 90 districts), old grassland stands were less dominant (50-75%). Except for one single district, all remaining districts (n = 65 districts) in the north-western and south-western part had an estimated proportion of old grassland stands of only 25-50%. High proportions of mid-aged grassland (> 50%) were calculated for only 9 districts in the western and southern part of the study area, while low proportions (< 12.5%) were estimated for 15 districts in the eastern part (Fig. 10B).

Most districts, however, featured 12.5-50% mid-aged grassland. For districts in the western and north-western parts of the Lahn-Dill Highlands we estimated 25-50% mid-aged grassland compared to 12.5-25% in districts in the east and south-east. Most notable for all districts, however, was the relatively low proportion of young grassland, which varied between 2% and 31% (Fig. 10C). Eighty-five districts had even less than 12.5% young grassland. These districts were found in the east and the west of the study area. Only districts in the southern part of the Lahn-Dill Highlands had an estimated proportion of 25-50% young grassland.

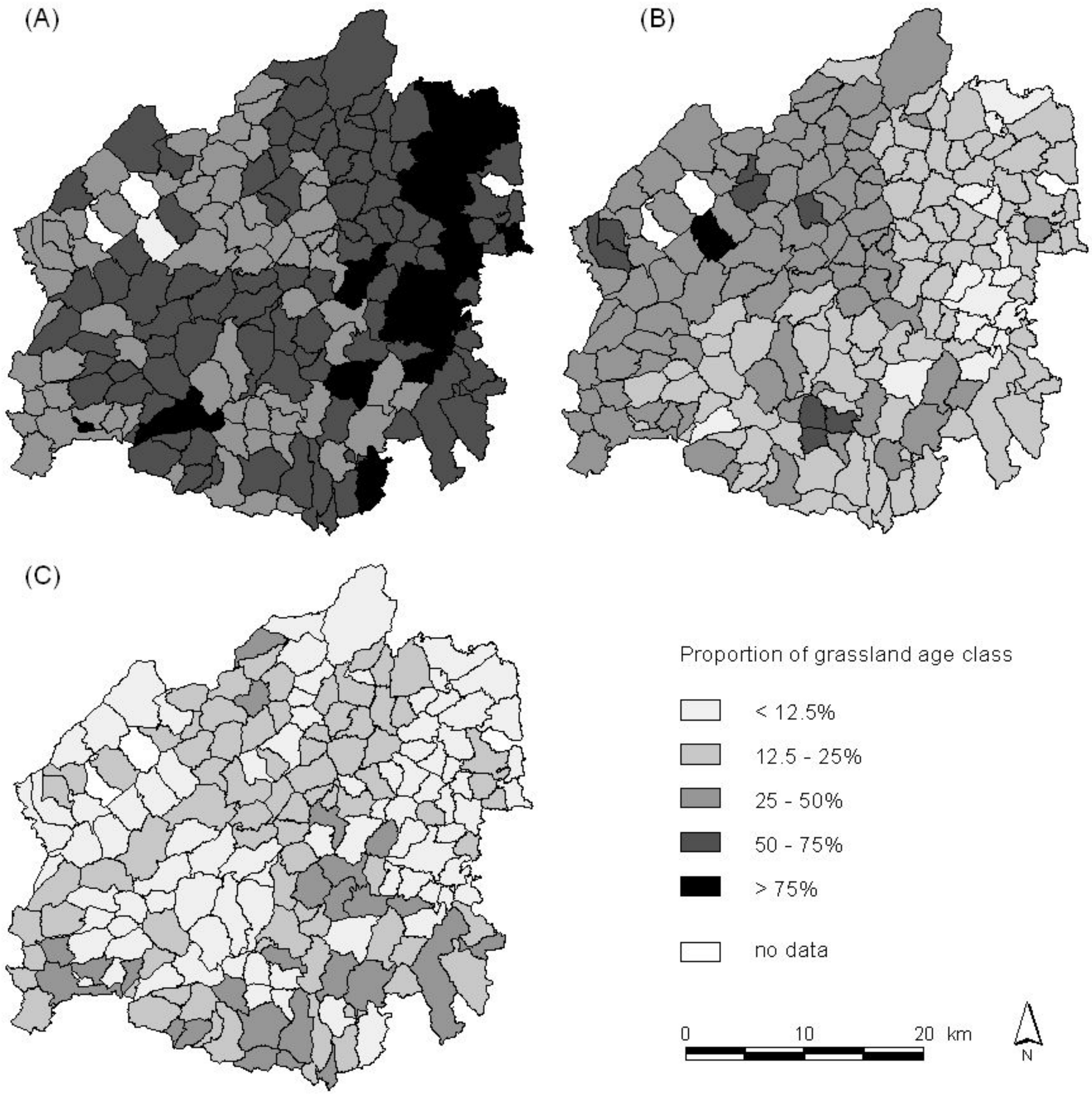

Fig. 10. Extrapolation results mapped for the grassland age classes (A) old (>47 years), (B) mid-aged (18-47 years), and (C) young (<18 years) in 187 districts.

5.3.3 Validation of extrapolation results

The validation results indicated that the estimated proportions of grassland belonging to the respective age classes were well in accordance with the reference data (Fuhr-Bossdorf et al., 1999) of test area A, Erda (Fig. 11A). The largest difference with 13% was found for young grassland stands under dry soil conditions (grassland type 121), while the other results differed less than 9%. In comparison to the district Erda, the validation results of test area B, Steinbrücken and Eibelshausen, revealed a lower conformity of the estimated proportions with the reference data (Fig. 11B). The differences varied between 2% and 21%. The calculated proportions obtained from spatial extrapolation tended to underestimate old grassland stands (3% to 13%), while the proportion of mid-aged grassland stands were overestimated (8% to 21%). However, statistical analysis of the grassland age classes revealed a significant difference between the extrapolation results and the reference data only for grassland type 121 (G-test, $G_{adj} = 9.89$, df = 2, $p < 0.01$).

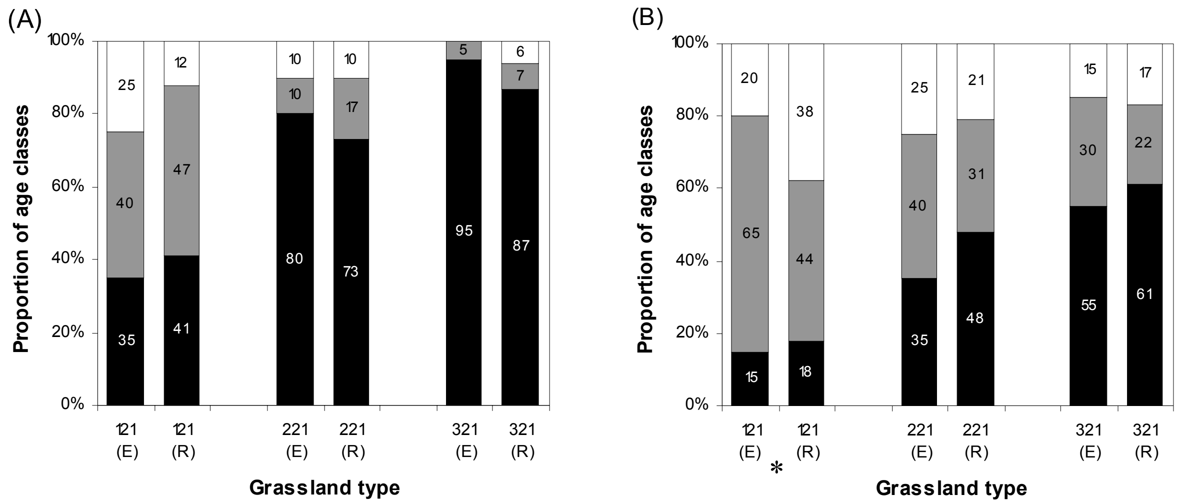

Fig. 11. Proportion of young ( ), mid-aged ( ), and old ( ) grassland of the test areas Erda (A) and Steinbrücken and Eibelshausen (B). Within each grassland type, left bars give the age structure of the extrapolation results (E) and right bars the age structure of the reference data (R). Coding of the grassland types (e.g. 121) refers to Table 4 in the order soil moisture, base-richness, and elevation. * Asterisk indicates significant differences (G-test, $G_{adj} = 9.89$, df = 2, $p < 0.01$).

5.4 Discussion

5.4.1 Methodological approach

In this paper we proposed a methodological approach for the assessment of the spatial distribution of grassland age in marginal European landscapes. The results showed that our approach is suitable for a realistic estimation of grassland age in a marginal landscape. We combined data sets and techniques of geographical information systems (GIS), remote sensing, and spatial extrapolation that are well established in landscape research. In the following, we discuss the steps of our methodological approach in detail.

Stratified random sampling

Stratified random sampling is considered to maximise the efficiency of landscape ecological assessment while focussing on maximum variation and representativeness of sampling (Goedickemeier et al., 1997; Knollová et al., 2005). Stratification enables the organisation of environmental variability of large areas into strata that are relatively homogeneous according to the characteristics of interest (Jongman et al., 2006). Thus, modern surveys of ecologically relevant data in large areas increasingly use environmentally stratified sampling designs (e.g. Bunce et al., 1996; Cooper and Loftus, 1998; Grabherr et al., 2003; Smart et al., 2003; Stohlgren et al., 1997).

The feasibility of our stratified random survey primarily depends on appropriate stratification data, their quality, and error propagation. Problems encountered at any of these levels can result in lower prediction success. The selection of data used for stratification in our study was primarily limited by availability. We used land-cover statistics, a DEM, and soil data for the two-stage stratification process to derive variables shown to be relevant for land-cover change in previous research (Hietel et al., 2004; Reger et al., 2007). Stratification permitted to systematically outline the spatial heterogeneity within the study area, regarding TLPDs at the district scale and physical attributes at the patch scale. A subsequently performed G-test confirmed that the grassland age of the investigated patches significantly differed according to the TLPDs and soil moisture, not, however, according to the physical attributes base-richness and elevation.

Generally, the assessment of accuracy and errors in spatial data like land-cover data (e.g. Bach et al., 2006; Foody, 2002; Wickham et al., 2004) or DEM (e.g. Bolstad and Stowe, 1994; Holmes et al., 2000; Wechsler and Kroll, 2006) has received considerable attention in recent research. Unfortunately, it has not yet become a standard to state positional and thematic accuracy of a data set (Bach et al., 2006). Hence, we do not know the accuracy of the DEM and soil data used. Visual screening for extreme values, however, gave us confidence of sufficient data quality. Errors in spatial data may be further propagated by GIS based operations like the conversion of different formats (i.e. vector to raster) or the intersection of different thematic layers (Heuvelink, 1998). Error propagation may thus influence the quality of the stratification outcome. However, general, integrated practical tools for statistical error propagation in GIS are still missing (Burrough and McDonnell, 2000).

Multitemporal aerial photograph interpretation

Aerial photographs provide only arbitrary snapshots in time. Our interpretation is dependent on six snapshots from the period 1953 to 2001, since further aerial photographs were not available. Therefore, temporary alterations of cropland and grassland use between two snapshots cannot be detected by using exclusively aerial photographs. This uncertainty could be diminished by a more integrative approach that also considers time-dependent patch characteristics like soil pH (Breuer et al., 2006; Waldhardt and Otte, 2003) or local farmers' knowledge on past management (Calvo-Iglesias et al., 2006a; Robertson and McGee, 2003). Nevertheless, several studies show that multitemporal aerial photograph interpretation is an effective way to receive valid information on past land-cover changes in agricultural landscapes, even when land-cover history is assessed by using aerial photographs of longer time intervals (e.g. Alard et al., 2005; Chen et al., 2001; Olsson et al., 2000; Poudevigne et al., 1997). Additional data sets for the analysis of land-cover change within larger regions, e.g. satellite images, historical maps, or statistics, are usually limited by a lack of appropriate time periods, infrequent observations, or a coarse resolution.

It seems to be of greater importance to consider visual misinterpretations as a source of under- or overestimated grassland age. Considering our sample size of 20 patches per grassland type, a misinterpretation of one patch has an effect of 5%. However, due to the variation in tonal contrast and its specific texture, grasslands in Central Europe can be readily identified in black and white aerial photographs at a scale of about 1:12 000 (Albertz, 1991; Schneider, 1974), so that potential misinterpretations may be minimal in terms of both, the sampled grassland patches and the reference data (Fuhr-Bossdorf et al., 1999).

Spatial extrapolation

Our multitemporal interpretation of aerial photographs facilitated the calculation of grassland type-specific age probabilities, which we then generalised via direct extrapolation to the district scale. Direct extrapolation is a prominent technique in landscape ecology in cases when measurements were made at relatively fine scale (Miller et al., 2004; Turner et al., 2001). The challenges of direct extrapolation lie in a correct definition of the spatial and temporal heterogeneity of the fine-scale information (i.e. grassland age) and both an accurate integration and aggregation of this heterogeneity to the broader scale (King, 1991). By aggregating the investigated grassland age into three classes, we considered successional stages of grasslands (cf. Austrheim and Olsson, 1999; Waldhardt and Otte, 2003), which may be expected to be of ecological relevance after abandonment of arable land. However, aiming for other target values such as soil properties, it may be necessary to adapt the classification of the basic stratification units as well as the age classes we used in our methodological approach.

Validity

The validation of our extrapolation results focussed on two test areas with contrasting land-cover dynamics since area-wide reference data for further test areas were not available. Nevertheless, our results indicate the validity of our approach. The comparison of the estimated proportional area of grassland age classes with the reference data of the two test areas (Fuhr-Bossdorf et al., 1999) showed a satisfying conformity for the test area Erda, which underwent rather moderate land-cover changes in the past. However, over- and underestimation up to 21% were detected for the test area Steinbrücken and Eibelshausen, being significant only for one grassland type. These uncertainties may be viewed as an effect of the extensive land-cover changes within these districts and the rather low number of investigated patches. A better assessment within such areas might be attainable by increasing the number of patches. However, a larger sample size and thus a higher certainty would also increase the amount of work and costs and may thus be only practicable for single grassland types.

5.4.2 Grassland age structure

The grassland age structure of the entire study area reflects a high spatial heterogeneity, which can be ascribed to successive land-cover changes reported in the Lahn-Dill Highlands (Hietel et al., 2004, 2005; Kohl, 1978; Schulze-von Hanxleden, 1972). The majority (49%) of investigated grassland patches were old (>47 years). But one could expect that in a landscape with such unfavourable conditions for arable farming (see Section 5.2.1), the proportion of permanent and old grasslands should be even higher. However, after World War II the lack of food led to relatively high proportions of arable land within the entire study area, so that even poor soils on steep slopes were cultivated (Schulze-von Hanxleden, 1972). Only sites with conditions completely unsuitable for arable farming (e.g. wet soil conditions) were used as permanent grasslands.

The high amount of mid-aged grassland patches (36%) may be closely related to major land-cover changes, which took place since the early 1960s. Since that time, Germany and other European countries strove to increase production and efficiency in agriculture (cf. Meeus et al., 1990). Economic prosperity and increasing mechanisation, intensification and specialisation of agriculture led to intensive cropland farming on more fertile sites whereas cultivation on less favourable sites ceased. Former fields were either turned into grassland or were completely abandoned (e.g. Bender et al., 2005; Fjellstad and Dramstadt, 1999; Krausmann et al., 2003; Mottet et al., 2006). Thus, particularly districts with poorest conditions for cultivation showed an increase of grassland (Hietel et al., 2004). Consequently, we found higher proportions of mid-aged grassland in these districts.

In the 1980s, Common Agricultural Policy (CAP) was fundamentally reoriented with the aim to reduce overproduction and environmental pressures caused by intensive production. With the implementation of the MacSharry reforms of the CAP in 1992, market price support was partially replaced by a system of direct payments to farmers. Accompanying these reforms, regional agri-environmental schemes were established that financially supported the extensification of grassland (de Putter, 1995; Primdahl et al., 2003). In their study, Hietel et al. (2007) identified that the agri-environmental schemes offered by the state of Hesse favoured the conversion to grassland use. These changing economic conditions for agricultural land use led to a further phase of abandonment in areas with inferior conditions for cropland farming, representing today's young grassland patches. These young patches are almost uniformly distributed across the entire study area, which indicates that economic developments in the last two decades affected the grassland age structure in all districts of the marginal landscape.

5.5 Conclusions

In this study we proposed a 3-step methodological approach to systematically assess the spatial distribution of grassland age in a marginal European landscape. Based on the combination of an a-priori two-stage landscape stratification with conventional aerial photograph interpretation of selected patches, and the subsequent spatial extrapolation of the determined grassland age, our approach sidesteps the shortcomings caused by the lack of feasible data on spatially explicit land-cover change. Results proved that our approach provides a realistic estimation of grassland age at the scale of districts and over a time period of five decades. We found that the derived probabilities of grassland age classes are specific for grassland types in areas with a homogenous pattern of land-cover change. Furthermore, the results confirmed a predominance of old grassland patches and a high amount of mid-aged grasslands that can be ascribed to major land-cover changes in this time period.

Due to comparatively simple data sets and techniques, our approach may be applied to other marginal agricultural landscapes under study - given the existence of feasible data. Further, our approach is suited for application in landscape models of various disciplines, which rely on large-scale information on grassland age. For instance, grassland age can be used as indicator for the prediction of vascular plant species richness in mosaic landscapes (Waldhardt et al., 2004). Moreover, the approach may be easily adapted to other land-cover types such as fallow land whose phytodiversity is also dependent on age (Simmering et al., 2001).

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Appendix 1. Grassland type-specific age probabilities ($p_h^{(a)}$; Equation 1) for young (<18 years), mid-aged (18-47 years), and old (>47 years) grassland patches within the six TLPDs (cf. Table 3). Coding of the grassland types (e.g. 111) refers to Table 4 in the order soil moisture, base-richness, and elevation.

Grassland type	Age class	TLPD					
		I	II	III	IV	V	VI
111	young				0.10	0.15	
	mid-aged				0.55	0.55	
	old				0.35	0.30	
121	young	0.25	0.35	0.15	0.05	0.20	
	mid-aged	0.40	0.35	0.35	0.55	0.65	
	old	0.35	0.30	0.50	0.40	0.15	
122	young		0.20	0.10	0.25	0.10	
	mid-aged		0.45	0.55	0.35	0.45	
	old		0.35	0.35	0.40	0.45	
131	young		0.20	0.05	0.05		
	mid-aged		0.45	0.40	0.45		
	old		0.35	0.55	0.50		
132	young			0.20	0.10		0.30
	mid-aged			0.30	0.45		0.45
	old			0.50	0.45		0.25
211	young	0.25			0.00	0.15	
	mid-aged	0.40			0.35	0.45	
	old	0.35			0.65	0.40	
221	young	0.10	0.30	0.25	0.25	0.25	0.20
	mid-aged	0.10	0.15	0.20	0.50	0.40	0.45
	old	0.80	0.55	0.55	0.25	0.35	0.35
222	young				0.30	0.20	0.35
	mid-aged				0.15	0.75	0.25
	old				0.55	0.05	0.40
231	young		0.30	0.15	0.10		
	mid-aged		0.05	0.15	0.25		
	old		0.65	0.70	0.65		
232	young			0.15			0.20
	mid-aged			0.35			0.50
	old			0.50			0.30
311	young				0.00	0.05	0.10
	mid-aged				0.40	0.95	0.25
	old				0.60	0.00	0.65
321	young	0.00	0.15	0.00	0.10	0.15	0.25
	mid-aged	0.05	0.10	0.05	0.30	0.30	0.45
	old	0.95	0.75	0.95	0.60	0.55	0.30
322	young			0.15	0.05	0.05	0.05
	mid-aged			0.30	0.25	0.50	0.30
	old			0.55	0.70	0.45	0.65
332	young		0.00		0.05		0.30
	mid-aged		0.15		0.10		0.35
	old		0.85		0.85		0.35

6. Potential effects of direct transfer payments on farmland habitat diversity in a marginal European landscape

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Abstract

Farmland habitat diversity in marginal European landscapes changed significantly in the past decades. Further changes towards homogenisation are expected, particularly in the course of European agricultural policy. Based on three alternative transfer payment schemes, we modelled spatially explicit potential effects on the farmland habitat diversity in a marginal European landscape. We defined (1) a scenario with direct transfer payments coupled to production, (2) a scenario with direct transfer payments decoupled from production, and (3) a scenario phasing out all direct transfer payments. We characterised habitat diversity with three indices: habitat richness, evenness, and rarity. The habitat pattern in 1995 served as reference for comparison. All scenarios predicted a general trend of homogenisation of the farmland habitat pattern, yet to a differing extent. Transfer payments coupled to production (Scenario 1) favoured the abandonment of agricultural production particularly in low-productive areas and arable land use in more productive areas. Habitat richness and habitat evenness had intermediate values in this scenario. Decoupling transfer payments from production (Scenario 2) supported grassland as most profitable farming system. This led to a grassland-dominated landscape with low values of all habitat diversity indices. Phasing out transfer payments (Scenario 3) resulted in complete abandonment or afforestation of agricultural land and extremely low values in all habitat diversity indices. Scenario results indicate that transfer payments may prevent cessation of agricultural production, but may not counteract homogenisation in marginal landscapes. Conserving high farmland habitat diversity in such landscapes may require support schemes, e.g. Pillar Two of EU Common Agricultural Policy.

Keywords

Common Agricultural Policy; Scenarios; Land-use modelling; Agricultural landscape; Landscape structure; Germany

6.1 Introduction

Habitat diversity, i.e. the composition and spatial structure of habitats in an area, is specified as a factor most adequate to evaluate biodiversity in agricultural landscapes (Duelli 1997). Several studies have pointed out close positive relationships between farmland habitat diversity and the diversity of certain taxonomic groups like plants (e.g. Gabriel et al., 2005; Simmering et al., 2006; Waldhardt et al., 2004), birds (e.g. Devictor and Jiguet, 2007; Robinson et al., 2001; Woodhouse et al., 2005), or arthropods (e.g. Hendrickx et al., 2007; Jeanneret et al., 2003; Purtauf et al., 2005; Weibull et al., 2003). Further, habitat diversity has also been shown to be positively related with the genetic diversity of arthropod (Holzhauer et al., 2006; Sander et al., 2006) and plant populations (Prentice et al., 2006). Agricultural landscapes of high habitat diversity are thus often areas of high biodiversity conservation value (cf. Bennett et al., 2006; Benton et al., 2003).

In Europe, high habitat diversity may be expected in marginal agricultural landscapes, which are characterised by poor environmental conditions for modern agriculture. Particularly in mountainous areas, these landscapes typically feature a heterogeneous topography and a small-scale pattern of different soils (MacDonald et al., 2000). Although these landscapes have already undergone significant land-use changes in the past decades, the traditional small-parcelled mosaic of low-intensity farming systems has been preserved in some areas (Meeus et al., 1990). As a result of the high diversity of environmental conditions and low-intensity farming systems, these marginal agricultural landscapes offer a rich variety of farmland habitats (Baldock et al., 1996; MacDonald et al., 2000). However, the diversity of farmland habitats in these landscapes is threatened by future land-use change that may lead to a homogenisation of the landscape (cf. Jongman, 2002).

The European Union’s (EU) Common Agricultural Policy (CAP) has largely affected changes in Europe’s agricultural land use since the 1960s. The CAP is a strong policy framework supporting the EU’s agricultural sector mainly through transfer payments, which have large impacts on farm management decisions (Gay et al., 2005). Recent and expected CAP-changes to cut costs and to comply with the world trade agreements are likely to be a major driver of agricultural change in the future (Robinson and Sutherland, 2002; Strijker, 2005). This requires detailed research on how agricultural policies may impact the environment at the landscape scale.

In recent studies, a multitude of scenario-based modelling approaches were developed to quantify and predict these potential effects of agricultural policies on the environment. They primarily addressed land use (e.g. Höll and Andersen, 2002; Lehtonen et al., 2005; Rounsevell et al., 2005; van Meijl et al., 2006; Weinmann et al., 2006), but also on water quality (e.g. Bärlund et al., 2005; Schmid et al., 2007), or species diversity (e.g. Gottschalk et al., 2007; Sheridan and Waldhardt, 2006; Sheridan et al., 2007). Scenario-based studies relating agricultural policies and habitat diversity at the landscape scale are, in contrast, scarce (but see Bolliger et al. (2007) and Miettinen et al. (2004)).

This study aims to assess potential effects of alternative direct transfer payment schemes (part of Pillar One of the CAP) on the farmland habitat diversity in marginal European landscapes. Therefore, agri-environmental schemes (as supported by Pillar Two of the CAP) were explicitly excluded. We applied our analyses to the Dill catchment, a low-mountainous landscape in Hesse, Germany, with a high variety of farming systems and physical conditions. We focussed on the effects of three scenarios representing alternative transfer payment schemes to illustrate the effects of varying CAP frameworks. Scenario analyses were complemented by an investigation of the farmland habitat pattern in 1995, which served as the basis for comparisons.

6.2 Study area

The Dill catchment (644 km²) is a traditional agricultural landscape in the western part of Hesse (Germany). It has been included in the EU less-favoured area support scheme since 1976 (EC Regulation No 75/268). With altitudes between 150 and 670 m above sea level (a.s.l.), the landscape is characterised by a rough and rather damp climate (mean annual temperature: 7° C, mean annual precipitation: 900 mm). It is part of the eastern Rhenish Uplands and mainly composed of clay schist, siliceous schist, diabase, and greywacke. Soil types in the area form a heterogeneous, small-scale mosaic of cambisols, luvisols, planosols, gleyosols, and acidic shallow ranker soils. The agrarian structure is dominated by small farm sizes (mean farm size of about 14 ha; Waldhardt and Otte, 2003) and a small-scale mosaic of arable fields, grasslands, and fallow (mean field size around 0.4 ha; cf. Simmering et al., 2006; Waldhardt et al., 2004). Like in many other marginal European landscapes, the land-use pattern in the Dill catchment has changed substantially during the last six decades (Reger et al., 2007). Arable crop production was largely replaced by extensive grassland use or abandoned fields in some parts of the area (Hietel et al., 2004, 2005, 2007; Kohl, 1978;

Schulze-von Hanxleden, 1972). Prevailing farming practices are characterised by a low-input of nutrients and pesticides. Due to the combination of heterogeneous environmental conditions and the small-scale mosaic of land-use types, the Dill catchment features a high diversity of farmland habitats and is thus one of the most species-rich low-mountainous landscapes in Germany (Nowak, 1988).

6.3 Materials and methods

6.3.1 Land-use pattern in 1995

Information on the land-use pattern of the Dill catchment was derived from a satellite image interpretation from 1995 (Landsat-TM, 25 m; Nöhles, 2000). The data set included spatially explicit information on forest, waters, settlement, and agricultural land. The latter was classified into arable land, grassland, and fallow land. Fallow land was defined as former agricultural land with shrub succession abandoned several decades ago. Users' accuracy for arable land was 95.8%, for grassland 86.7%, and for fallow land 63.3%. Overall classification accuracy for the data set was 86% (Nöhles, 2000). The satellite-derived land-use pattern in 1995 served as basis for scenario modelling and comparison.

6.3.2 Transfer payment schemes

Transfer payments play an important role in the composition of farmers' incomes and, therefore, influence farmers' production decisions (Gay et al., 2005). The EU's CAP in the period from the 1960s to the early 1990s used market price support to improve farmers' incomes. In 1992, the CAP was fundamentally reoriented. Support prices were reduced and a system of transfer payments coupled to production was introduced. Both support schemes influenced farmers' land allocation decisions as certain production systems became economically more attractive than others. The CAP reform in 2003 replaced coupled payments with decoupled single farm payments (EC, 2004). Such decoupled payments do not influence the farmers' land allocation to certain production systems but only increase the revenues per ha of farm land. In Germany, single farm payments will be homogenised within specific regions to fully decoupled payments. All farmers within a certain region will then receive identical transfer payments per ha of farm land, i.e. an area payment.

In addition to income oriented transfers (referred to as Pillar One of the CAP), agri-environmental and infrastructural measures (referred to as Pillar Two of the CAP) were introduced with the Agenda 2000 and strengthened with the CAP reform of 2003. Such payments can contribute considerably to farmers' incomes in less-favoured areas. However, the study's objective is assessing the effects on habitat diversity of Pillar One direct transfer payments. Therefore, Pillar Two support schemes were explicitly excluded.

We defined three plausible scenarios to analyse the effects of coupled and decoupled transfer payments as well as complete abandonment of direct transfers. Transfer schemes were varied while prices for inputs and outputs were kept constant.

Scenario 1 'coupled transfers' reflects the transfer payment scheme of the Agenda 2000. This CAP-agreement from 1999 was originally planned to be implemented in several steps from 2000 to 2006 (EC, 1999). In the scenario, transfer payments are coupled to crop production (347 €/ha for cereals and oilseeds, 399 €/ha for protein crops) and heads of livestock (i.e. suckler cows, bulls, steers, dairy cows). Land set-aside is supported with 347 €/ha. Permanent grassland receives no payments.

Scenario 2 'decoupled transfers' refers to the 2003 CAP reform with a decoupled area payment for agricultural land use. Official estimates for the year 2013 predict a uniform area payment of 302 €/ha in the state of Hesse (BMVEL, 2005). Receiving decoupled transfer payments will not require any agricultural commodity production. However, farmland has to be kept in good agricultural and environmental condition.

In Scenario 3 'no transfers' all Pillar One direct transfer payments are phased out. However unrealistic such a scenario may seem, the reduction of agricultural support has long been demanded by other countries in WTO negotiations and emerges repeatedly in principal discussions on the future of the CAP (Lamy, 2001; WTO, 2004).

6.3.3 Spatial implementation of the transfer payment schemes

Land-use patterns for the three scenarios were generated by the agro-economic land-use model ProLand. As a comparative-static model, it simulates agricultural and silvicultural land-use patterns as endpoints of adaptation processes (see Kuhlmann et al. (2003), Sheridan et al. (2007), and Weinmann et al. (2006) for detailed model descriptions). The prediction is based on small-scale data of an area's natural, technological, political, and socio-economic characteristics and price and quantity structures of agricultural and silvicultural land-use systems. Land-use systems are sets of crop rotations and corresponding field operations. Thus, varying land-use intensities of the same land-use type, i.e. arable land, grassland or forest, are explicitly modelled. First, crop yields are estimated endogenously assuming linear response and plateau functions employing available water capacity, and monthly precipitation and temperature sums as variable function parameters. Revenues are then calculated with exogenous farm-gate product prices and transfer payments. Production costs are adjusted to plot specific conditions considering plot size, slope, tilling resistance, and yields. ProLand thus determines the land rent maximising land-use system for each land-use parcel. Assuming land rent maximising land users (Alonso, 1964; Brinkmann, 1922; Dunn, 1967), each parcel is then classified as arable land, grassland, or forest. Parcels with negative land rent are classified as fallow land. A more detailed land-use classification according to land-use systems is possible but not sensible given the study's objective.

6.3.4 Farmland habitat patterns

We used topographical and edaphical data as well as information on land use to classify habitat conditions that are relevant for plant species composition (cf. Waldhardt et al., 2004), and specified farmland habitats as follows: We combined the four land-use maps, i.e. recent land use and the three modelled land-use patterns, with data on elevation, soil moisture, and base-richness to generate the farmland habitat patterns for our study area.

Information on elevation was obtained from a digital elevation model (DEM, 40 m; Hessian Administration for Soil Management and Geoinformation). To match the raster data with the land-use map in 1995, we altered the cell size of the DEM to 25 m × 25 m by bilinear interpolation. Based on the modified DEM, we calculated altitude (metre a.s.l.) and classified two elevations, colline and submontane (Table 5).

Soil moisture and base-richness were derived from official digital soil maps (scale 1:50 000) obtained from the Hessian State Agency for Environment and Geology. Soil moisture was classified into four classes combining information on available water capacity (AWC) in the root zone, the degree of soil wetness, i.e. gleyic and stagnic soil properties, and slope (Table 5). Slope was derived from the DEM by the use of the slope function in the ArcGIS 9.1 Spatial Analyst tool, which calculates the maximum rate of change of elevation between each cell and its eight neighbouring cells (Burrough and McDonnell, 2000). Base-richness was classified into four classes according to substrate properties of the soils (Table 5). Using ArcGIS 9.1, the vector datasets were converted to raster data with 25 m × 25 m cell size.

Table 5. Physical attributes and classes used for the determination of farmland habitat types.

Variable	Class	Description
Elevation	1 colline	≤400 m a.s.l.
	2 submontane	>400 m a.s.l.
Soil moisture	1 dry	AWC <50 mm or AWC4 50-90 mm and >5° slope
	2 mesic	AWC >90 mm or AWC4 50-90 mm and ≤5° slope
	3 moist	Low to mean gleyic/ stagnic soil properties
	4 wet	High gleyic/ stagnic soil properties
Base-richness	1 base-poor	Soil types with acidic substrate (e.g. sandstone to claystone, quartzite, schist)
	2 moderate	Soil types with neutral substrate (e.g. siliciclastic sedimentary rocks, metamorphic rocks)
	3 base-rich	Soil types with alkaline substrate (e.g. basaltic extrusive rocks)
	4 calcareous	Soil types with calcareous substrate (e.g. limestone, dolomite)

These physical attributes were intersected with the information on land use. The intersection was processed with the help of ESRI Model builder ArcGIS 9.1 and resulted in four maps representing different habitat patterns for recent and modelled land use.

6.3.5 Habitat diversity indices

Differences in the farmland habitat patterns were assessed by a set of three complementary habitat diversity indices, habitat richness, habitat evenness, and habitat rarity.

Habitat richness simply refers to the number of different farmland habitat types, i.e. arable land, grassland, and fallow habitats in a landscape unit of a standard size. Habitat richness thus measures the habitat composition of that unit.

The spatial distribution of habitat types in landscape units was assessed by calculating habitat evenness (HE_k) as a measure of structural diversity. The calculation followed Pielou (1969):

$$HE_k = \frac{-\sum_{i=1}^{n}(P_i * \ln P_i)}{\ln n} \tag{1}$$

where n is the number of habitat types in landscape unit k, P_i measures the proportion of area covered by habitat type i and ln denotes the natural logarithm. Habitat evenness is one if the distribution of area among all habitat types is equal, and approximates zero as one habitat type becomes dominant. Evenness is by definition zero if there is only one habitat type in the landscape unit (McGarigal et al., 2002).

Besides the compositional and structural components of habitat diversity, the fate of rare habitats is of specific interest for the assessment of transfer payment effects. To express the occurrence and amount of rare habitats in landscape units, we calculated the habitat rarity index (HR_k), which was proposed by Simmering et al. (2006). The index accounts for rare habitat types inside landscape units with respect to the entire landscape. Thus, it identifies landscape units that contain habitat types that are rare at the landscape scale. The index was calculated with the equation (modified from Simmering et al. (2006)):

$$HR_k = \sum_{i=1}^{n} \frac{a_{ik}}{A_k} * \frac{a_{ik}}{A_{it}} \qquad (2)$$

where n is the number of habitat types in landscape unit k, a_{ik} the area of habitat type i in landscape unit k, A_k the total area of landscape unit k, and A_{it} the overall area of habitat type i in the study area. High values indicate the occurrence of more and/or larger proportions of rare habitat types in a landscape unit.

All indices were calculated for a network of 2676 landscape units with the standard size of 22.6 ha (475 m x 475 m). Although arbitrary in the exact dimension, this size corresponds to units used in several landscape pattern studies (e.g. Poudevigne and Alard, 1997; Sheridan and Waldhardt, 2006; Simmering et al., 2006). The size of the landscape units was further conditioned by technical and structural requirements: In much smaller landscape units, the indices could not be effectively calculated since most landscape units of the highly fragmented landscape would have had a uniform habitat composition. Much larger landscape units, instead, would have provided less detailed information on the spatial pattern of habitat diversity. Landscape units containing no farmland habitats (i.e. arable land, grassland, or fallow land habitats) were excluded from analysis. We used the software Fragstats 3.3 (McGarigal et al., 2002) to quantify habitat richness and habitat evenness and a spreadsheet programme to calculate habitat rarity. The calculation of habitat rarity partly resulted in extremely low values. For ease of presentation, all habitat rarity values were multiplied by 1000.

The calculated indices were tested for differences between the four habitat maps using STATISTICA 6.0 software (StatSoft Inc., 2001). Due to the non-normal distribution of the datasets, we chose the nonparametric Friedman test as a powerful statistical method for the two-way analysis of variance by ranks of several related data (Legendre and Legendre, 1998). In case of significance, the analysis was followed by a Wilcoxon signed-ranks test with Bonferroni correction for multiple testing ($p < 0.05$).

6.4 Results

6.4.1 Habitat pattern

In 1995, the landscape was dominated by a high proportion of afforested areas (59.7%; Table 6). Only 27.8% were agricultural land. The farmland habitat pattern predominately consisted of grassland (62.3%). Arable land and fallow land amounted to 18.9% and 18.8%, respectively. In total, we classified 72 farmland habitat types with 24 arable land habitats, 24 grassland habitats, and 24 fallow land habitats. Grassland habitats predominated in the northern and western part of the Dill catchment, while in the south-eastern part the farmland habitat pattern consisted of a small-scale mosaic of arable land and grassland habitats (Fig. 12). In the central part of the study area, the pattern was dominated by fallow land habitats.

Table 6. Area (ha) and percentage (%) of land use in 1995 and in the three policy scenarios.

	Status 1995		(1) Coupled transfers		(2) Decoupled transfers		(3) No transfers	
	ha	%	ha	%	ha	%	ha	%
Arable land	3400	5.3	3343	5.2	1359	2.1	0	0.0
Grassland	11177	17.3	8889	13.8	15453	24.0	0	0.0
Fallow land	3370	5.2	5028	7.8	769	1.2	10987	17.0
Forest	38503	59.7	39056	60.6	38735	60.1	45329	70.3
Waters	219	0.3	219	0.3	219	0.3	219	0.3
Settlement and others	7776	12.1	7910	12.3	7910	12.3	7910	12.3
Total	64445	100.0	64445	100.0	64445	100.0	64445	100.0

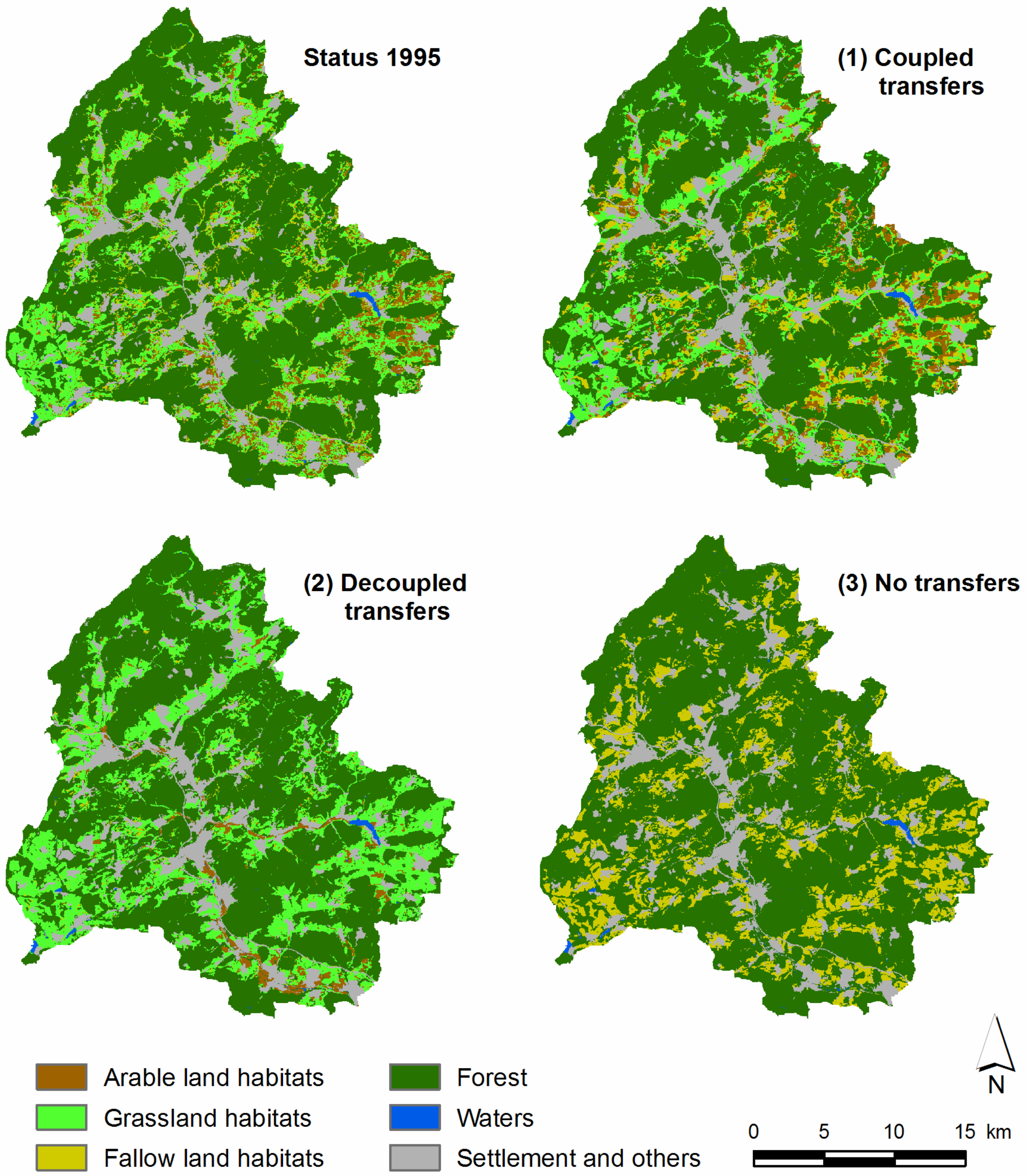

Fig. 12. Habitat pattern in 1995 and the three policy scenarios in the Dill catchment (for ease of illustration, farmland habitat types were subsumed to the level of similar land use).

In Scenario 1 with coupled transfer payments, the predicted proportion of forest amounted to 60.6% (Table 6). The farmland habitat pattern (26.8%) was characterised by 51.5% grassland and a comparably high proportion of fallow land (29.1%). Subsidies coupled to crops favoured arable land on the more productive sites in the south-eastern part of the study area and abandonment on less productive soils in the central part of the study area (Fig. 12). In this scenario, the landscape contained a total of 70 habitat types with 22 arable land habitats, 24 grassland habitats, and 24 fallow land habitats.

Decoupling transfer payments from production (Scenario 2) resulted in an extremely high proportion of grassland (87.9% of the agricultural land), whereas arable land and fallow land totalled only 7.7% and 4.4% (Table 6). Grassland with an annual cutting regime (i.e. mulching) was considered as the most profitable management practice. Therefore, most parts of the study area were dominated by grassland habitats (Fig. 12). Only some areas in the southern and central part of the study area had higher proportions of arable land or fallow land. With 16 arable land habitats, 24 grassland habitats, and 23 fallow land habitats, we assessed a total number of 63 habitat types for the study area.

Phasing out transfer payments (Scenario 3) resulted in complete abandonment of arable land and grassland, which were both not profitable without the support of subsidies. Correspondingly, 70.3% of the study area were afforested. The remaining 17% of agricultural land were not even profitable for silvicultural land-use systems and were thus completely abandoned (Table 6, Fig. 12). The farmland habitat pattern consisted of a total number of 24 remaining types of fallow land habitats.

6.4.2 Habitat diversity

The statistical analysis of the farmland habitat pattern in 1995 and the three scenarios revealed significant differences in habitat richness, habitat evenness, and habitat rarity at the scale of landscape units (Table 7, Fig. 13).

Table 7. Comparison of the farmland habitat pattern in 1995 and the three scenarios at the scale of landscape units (size: 22.6 ha) according to habitat richness, habitat evenness, and habitat rarity (see Section 6.3.5 for description of the indices). Different letters indicate significant difference ($p < 0.05$) determined by Wilcoxon signed-ranks test with Bonferroni correction. n = number of investigated landscape units.

		Habitat richness		Habitat evenness		Habitat rarity	
	n	Median	25-75% percentile	Median	25-75% percentile	Median	25-75% percentile
Status 1995	2106	6.0[a]	3-9	0.77[a]	0.66-0.85	1.13[a]	0.18-3.73
(1) Coupled transfers	2092	4.5[b]	3-7	0.73[b]	0.56-0.83	1.25[b]	0.18-4.16
(2) Decoupled transfers	2098	4.0[c]	2-5	0.70[c]	0.52-0.81	1.04[c]	0.15-3.50
(3) No transfers	1824	3.0[d]	2-4	0.66[d]	0.35-0.82	0.51[d]	0.05-2.10

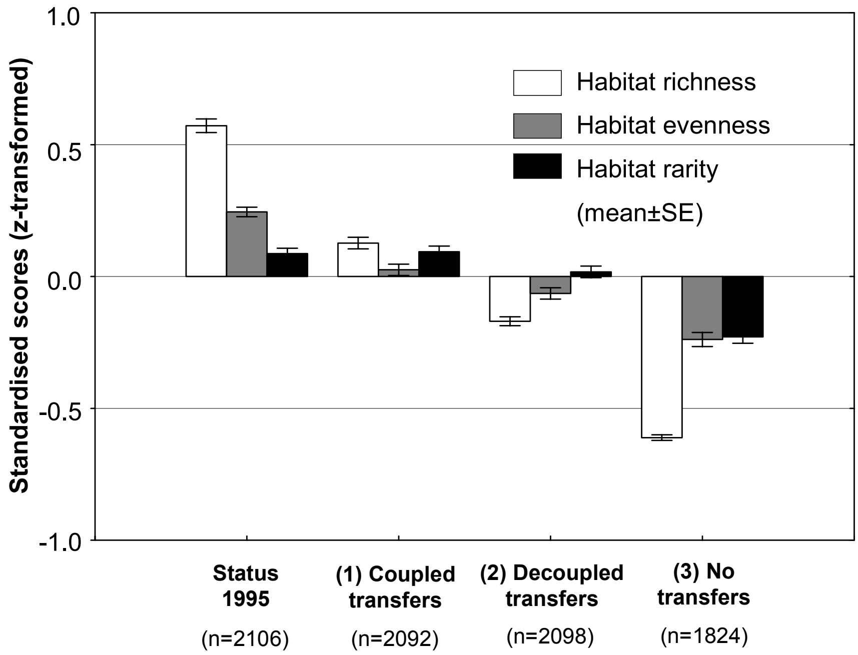

Fig. 13. Standardised mean values and standard error (SE) of habitat richness, habitat evenness, and habitat rarity calculated for the farmland habitat pattern in 1995 and the three scenarios at the scale of landscape units. Zero represents the overall mean of the respective index, one its standard deviation. n = number of investigated landscape units.

Habitat richness was highest for the farmland habitat pattern in 1995. We found a median of 6 habitat types per landscape unit (Table 7). Significantly lower values of habitat richness were predicted for the farmland habitat pattern of Scenario 1 with coupled transfer payments (4.5 habitat types), and Scenario 2 with decoupled transfer payments (4 habitat types). Scenario 3 with no transfer payments featured a median of 3 habitat types per landscape unit. The analysis of the landscape units revealed considerable variations of habitat richness primarily for the farmland habitat pattern in 1995 (Fig. 14). Landscape units with 10 or less different habitat types were predominant. However, 11.3% of the 2106 landscape units had 11-15 habitat types, while 0.9% contained 16-20 habitat types, and 0.1% had even more than 20 habitat types. These habitat rich landscape units were evenly distributed except for the far western part of the study area. In Scenario 1 and Scenario 2, we found only few landscape units with high numbers of habitat types. With 95.8% (of 2092 landscape units) and 98.7% (of 2098 landscape units) almost all landscape units had 10 or less habitat types. In Scenario 3 with no transfer payments, 3.2% of 1824 landscape units included 6-10 habitat types. The remaining landscape units contained less than 6 habitat types.

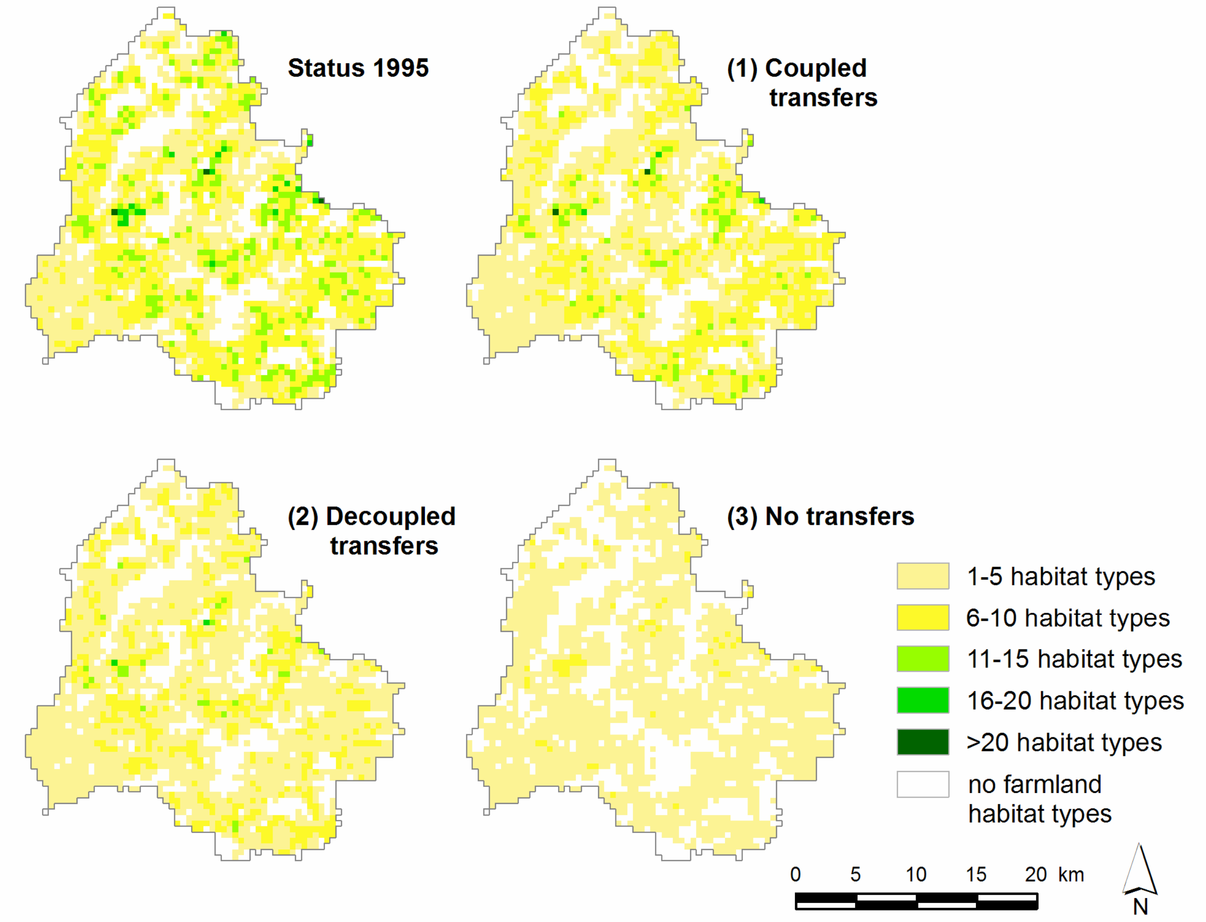

Fig. 14. Spatial distribution of habitat richness calculated for the farmland habitat pattern in 1995 and the three scenarios at the scale of landscape units (size: 22.6 ha).

The ranking of the four habitat patterns by habitat evenness was similar to that by habitat richness (Table 7, Fig. 13). At the scale of the landscape units, we found that the habitat types in 1995 were slightly more evenly distributed (median evenness 0.77) than the habitat types of Scenario 1 'coupled transfers' (0.73), Scenario 2 'decoupled transfers' (0.70), or Scenario 3 'no transfers' (0.66). In 1995, about 41.8% of 2106 landscape units had evenness values between 0.8 and 1.0 (Fig. 15). The habitat types within these landscape units were thus almost evenly distributed. Coupling transfer payments (Scenario 1) and decoupling transfer payments (Scenario 2) showed lower proportions of landscape units with an almost even distribution of habitat types. Only 34.2% and 28.3% landscape units belonged to category 0.8 to 1.0. The landscape units of Scenario 3 were characterised by a comparatively high proportion of dominating habitat types. 72.6% of 1824 landscapes units had evenness values below 0.4. These landscape units were predominantly located in the far western part of the study area.

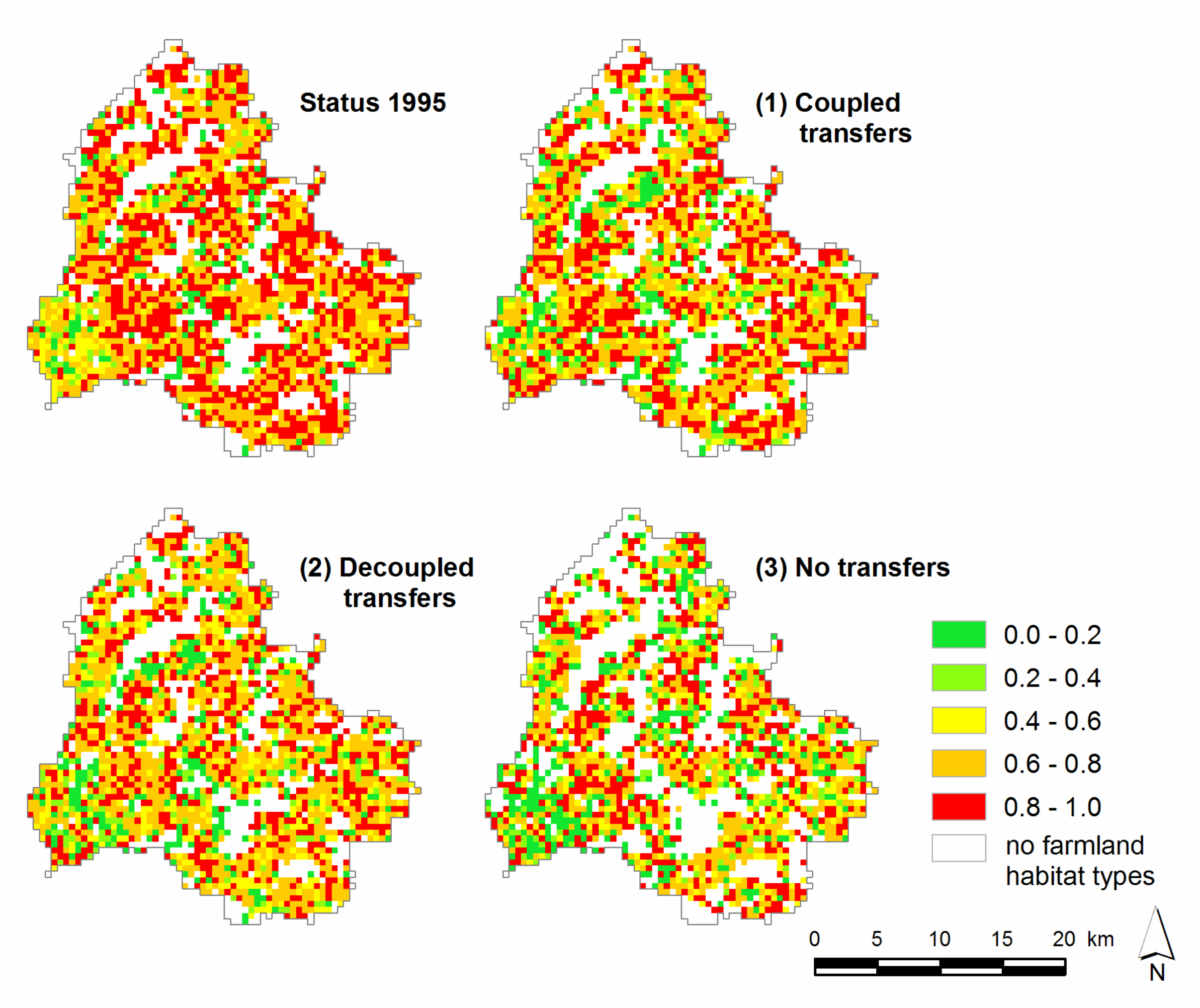

Fig. 15. Spatial distribution of habitat evenness calculated for the farmland habitat pattern in 1995 and the three scenarios at the scale of landscape units (size: 22.6 ha).

The comparison of the habitat patterns according to rare habitats revealed comparatively high habitat rarity values per landscape unit in 1995 (median of 1.13; Table 7). The values were even higher in Scenario 1 with ‘coupled transfers’(1.25) and slightly lower in Scenario 2 ‘decoupled transfers’ (1.04), which indicates that the scenarios still contain habitats rare at the landscape scale. Scenario 3, in contrast, with ‘no transfers’ and characterised by a complete abandonment of arable land and grassland, had the lowest median values for habitat rarity (0.51). In all scenarios, landscape units with higher values were primarily concentrated in the northern and western part of the study area with a rare combination of physical conditions (Fig. 16).

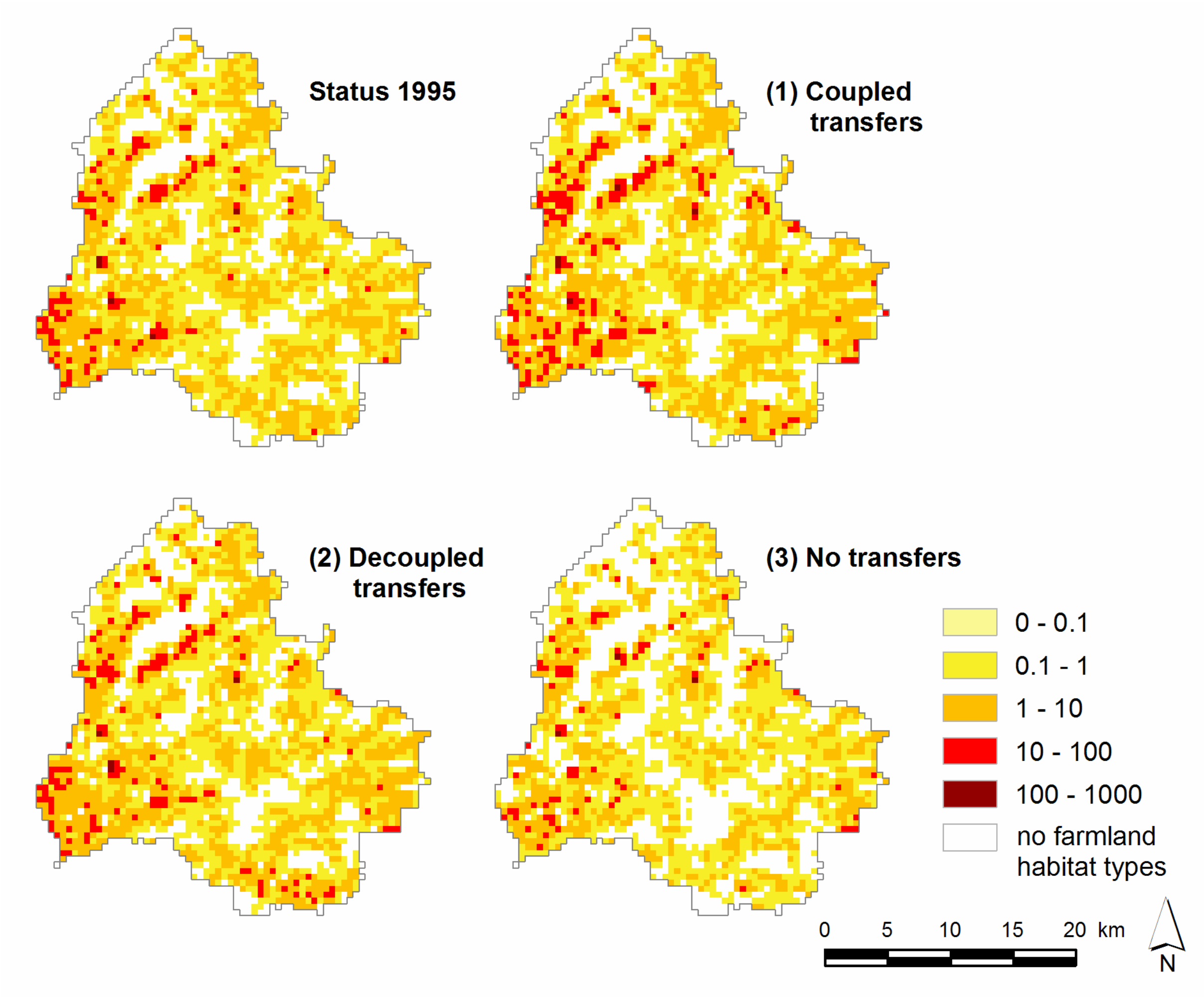

Fig. 16. Spatial distribution of habitat rarity calculated for the farmland habitat pattern in 1995 and the three scenarios at the scale of landscape units (size: 22.6 ha).

6.5 Discussion

The farmland habitat pattern of the study area in 1995 confirmed a high diversity, which is typical of marginal, low-mountainous European landscapes (Baldock et al., 1996; MacDonald et al., 2000). The highly diverse site conditions in combination with a small-scale land-use pattern led to a small-scale mosaic of arable land, grassland, and fallow land habitats. The comparatively high amount of fallow land on old fields indicates, in this respect, the adaptation of agricultural land use to the unfavourable environmental and socioeconomic conditions. In the past decades, these have led to land abandonment in the Dill catchment (Hietel et al., 2005) and in other marginal European landscapes (e.g. Calvo-Iglesias et al., 2006b; Gellrich et al., 2007; Krausmann et al., 2003; van Doorn and Bakker, 2007). As shown in these studies, large proportions of arable land on less productive sites have been consecutively abandoned in favour of grassland, fallow land, or afforestation.

Transfer payments coupled to crop production (Scenario 1) led to a comparatively high amount of arable land habitats and fallow land habitats. Financial support of crops specifically favours arable land use on more productive soils and enhances the risk of land abandonment in low-productive areas. Coupled transfer payments thus slightly favoured the spatial segregation of land use, which occurred in marginal European landscapes within past decades. Consequently, differences in the land-use patterns between Scenario 1 and 1995 were rather low. This was also shown by Bolliger et al. (2007) for Switzerland and by Miettinen et al. (2004) for Finland.

The amount of area covered by arable land habitats and fallow land habitats was predicted to be extremely low with decoupled transfer payments (Scenario 2). Decoupling transfer payments makes mulching more profitable than all other management systems on low-productive sites as farmers still receive a uniform area payment. Hence, the decoupling of transfer payments led to a landscape dominated by grassland. Similar trends of grassland expansion were also predicted by Schmid et al. (2007) for Austria.

Phasing out transfer payments (Scenario 3) finally led to a complete abandonment or afforestation of agricultural land. At first sight, this scenario result appears extreme and unrealistic. However, several studies confirmed that liberalisation would increase the risk of large-scale land abandonment in mountainous landscapes (e.g. Bolliger et al., 2007; Lundström et al., 2007; Verburg et al., 2006). According to these studies, large-scale abandonment of agricultural land would become an important issue for land use and farmland habitat diversity in Europe.

The analysis of the farmland habitat pattern in 1995 and the three scenarios revealed differences in the spatial distribution of habitat richness, habitat evenness, and habitat rarity at the scale of the landscape units. Habitat richness and habitat evenness were strongly affected by the predicted land-use changes within the respective landscape units. Higher habitat rarity values, in contrast, were found predominately in areas with distinct physical site conditions. However, the predicted land use in Scenario 3 led also to a severe reduction of rare habitats.

The three indices showed a similar ranking at the landscape scale. Lower farmland habitat diversity was found in all scenarios. These results are in line with the ongoing general trend of simplification and homogenisation of landscapes. This development has been observed during the past decades of modern agriculture, and is predicted by recent studies to continue for marginal as well as for intensively managed landscapes (Hietala-Koivu, 2002; Jongman, 2002; Roura-Pascual et al., 2005).

Our results indicate the importance of subsidies for the preservation of agricultural land use in landscapes with mostly unfavourable conditions for agriculture. Without transfer payments, the diversity of habitats for species confined to open, agricultural landscapes would be lost. The loss of low-input grassland and arable land habitats is considered the major threat to species diversity, particularly for rare and declining species (Korneck et al., 1998). Large-scale abandonment would, however, favour species, which profit from succession processes in woody habitats (Frelechoux et al., 2007; Simmering et al., 2001). Transfer payments alone may not prevent the ongoing homogenisation in marginal landscapes. Agri-environmental schemes offered in Pillar Two of the CAP may provide additional financial incentives to maintain small-scale mosaics of arable land, grassland, and fallow land habitats. Currently, however, these schemes do not explicitly focus on the creation and management of farmland habitat diversity (Benton et al., 2003; Concepción et al., 2008).

It is important to recognise that scenario predictions remain unlikely future landscape realisations. All models undoubtedly involve some uncertainties and limitations (Rounsevell et al., 2006). In general, scenarios are a product of their time and consider factors deemed influential at the time (Audsley et al., 2006). Exogenous variables like the rate of future technological innovations or price developments can differ more notably in the future than expected. Estimates of their respective rates may thus be afflicted with uncertainties. Re-calculating the scenarios with e.g. current prices for agricultural commodities would most likely lead to different results. Within these caveats however, scenario-based modelling is a useful tool that permits the study of potential effects on the environment, thereby improving our understanding of the underlying phenomena (e.g. Busch, 2006; Santelmann et al., 2006; Sheate et al., 2008; Verburg et al., 2002).

Our scenario approach was complemented by an analysis of the farmland habitat pattern in 1995. This pattern reflects one observed realisation of the farmland habitat pattern in the study area. As such, the analysed habitat diversity served as reference basis for the evaluation of our scenario results. However, due to the uncertainties and limitations of the scenario approach and the type of land-use model used, we did not compare our scenarios with the status of 1995 in the temporal dimension. Therefore, we did not analyse rates and directions of habitat change. Instead, we concentrated on the analysis and comparison of four states of farmland habitat patterns.

6.6 Conclusion

The study shows that alternative transfer payment schemes may have pronounced effects on farmland habitat diversity in a marginal European landscape. All scenarios predicted a general trend of simplification and homogenisation of the farmland habitat pattern, yet to a differing extent. According to our scenario results, we suggest that the payment of subsidies prevents cessation of agricultural production in landscapes with mostly unfavourable conditions for agriculture. In the current specification, they will not counteract the general trend of homogenisation in marginal landscapes. Thus, if society wants to maintain landscapes with high habitat diversity, agricultural policies specifically favouring a mosaic of arable land, grassland, and fallow land habitats are needed. Therefore, sufficiently endowed agri-environmental schemes offered in Pillar Two of the CAP are highly important.

Acknowledgements

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7. General discussion

This chapter summarises the main results as well as the underlying data sets and implemented methods of the three studies (Chapter 4, 5 and 6) and discusses their relevance in the context of agricultural landscape change and related areas of research. Subsequently, conclusions, potential applications, and some suggestions for further studies are briefly outlined in the synthesis and perspectives section.

7.1 Agricultural landscape change

The results of our study on patterns and dynamics of land cover (Chapter 4) confirmed a general trend of decreasing arable land and increasing grassland and fallow land for the entire study area. Similar developments have been observed in other marginal European landscapes for the last decades (e.g. Calvo-Iglesias et al. 2006b; Gellrich et al., 2007; Mottet et al., 2006; van Doorn and Bakker, 2007). This general trend of agricultural marginalisation at the landscape scale, however, exhibits remarkable differences in the magnitude and the direction at the district scale. In the investigated model region, six distinct types of land-cover patterns and dynamics could be identified. Comparable spatial variations at the local scale have been described for other regions too, e.g. in France (Poudevigne and Alard, 1997), Sweden (Skanes and Bunce, 1997), Norway (Fjellstad and Dramstad, 1999), the Czech Republic (Lipsky, 1995), and Austria (Krausmann et al., 2003). In the years immediately following World War II, the lack of food led to relatively high proportions of arable land within the entire study area since even poor soils on steep slopes were cultivated (Schulze-von Hanxleden, 1972). Between 1955 and 1995 (and also thereafter), the process of marginalisation led to a widespread abandonment of cultivation on unfavourable sites. Former fields were either turned into grassland or were completely abandoned. Accordingly, an increased proportion of fallow land occurs in districts with steep slopes and dry soils. This result is in line with a study by Hietel et al. (2004) who - at the patch scale - found that fallow land in our study area is situated mainly on elevated, steep sites with sandy soils and a low available water capacity. Further, the grassland distribution in the study area exhibits a distinct gradient with a low proportion of grassland in the eastern basin areas and a very high proportion in the more elevated south-western parts. This gradient is well in accordance with the pattern of elevation at the district scale. A similar accordance can be observed for the LVZ values, which show a ranking order with favourable (high proportion of arable land), moderate (high proportion of grassland), and unfavourable conditions (high proportion of fallow land) for cultivation. Thus, the spatial variation of the agricultural land cover may be primarily explained by environmental variables such as elevation, slope, soil moisture, or LVZ values. But socioeconomic factors like changing labour markets, agricultural policies, migration, and infrastructure developments may also be related to the land-cover patterns and dynamics (e.g. Hietel et al., 2007; Strijker, 2005).

The grassland age structure of the Lahn-Dill Highlands (Chapter 5) exhibits a high spatial heterogeneity, which corresponds to the identified high spatial variation in the land-cover patterns and dynamics. After World War II, only sites with conditions completely unsuitable for arable farming (e.g. wet soil) were used as permanent grasslands. About half (49%) of the investigated patches were such old grasslands. The remaining grassland patches can be ascribed to successive land-use changes since the early 1960s, when Germany and other European countries began to strive for higher production and efficiency in agriculture (cf. Meeus et al., 1990). Economic prosperity and increasing mechanisation, intensification and specialisation of agriculture led to intensive cropland farming on more fertile sites whereas cultivation on less favourable sites ceased. In the course of this agricultural marginalisation, particularly in districts with poorest conditions for cultivation, a large amount of arable land has been largely replaced by grassland use. Thus, today a substantial proportion of grassland (36%) has been predicted as mid-aged, which indicates a pronounced land-cover change in this time period. With the reorientation of the Common Agricultural Policy (CAP) in the 1980s, regional agri-environmental schemes were established that financially supported the extensification of grassland (de Putter, 1995; Primdahl et al., 2003). These changing economic conditions for agricultural land use led to a further phase of abandonment in areas with inferior conditions for cropland farming, representing today's young grassland patches. These young patches are apparently evenly distributed across the entire study area, which indicates that economic developments in the last two decades affected the grassland age structure in all districts of the marginal landscape.

The third study (Chapter 6) specifically investigated the potential effects of transfer payments on the farmland habitat diversity in a marginal agricultural landscape. Results confirmed a high diversity for the farmland habitat pattern of the study area in 1995, which is typical of marginal, low-mountainous European landscapes (Baldock et al., 1996; MacDonald et al., 2000). The highly diverse site conditions in combination with a small-scale land-use pattern led to a small-scale mosaic of arable land, grassland, and fallow land habitats. The analysis of the farmland habitat pattern of the three scenarios revealed pronounced effects on habitat diversity. Lower farmland habitat diversity was predicted in all scenarios. These findings might be associated with the ongoing trend of simplification and homogenisation of landscapes, which has been observed during the past decades of modern agriculture, and is predicted to continue for marginal as well as for intensively managed landscapes (Hietala-Koivu, 2002; Jongman, 2002; Roura-Pascual et al., 2005). Transfer payments coupled to crop production (Scenario 1) specifically favoured arable land use on more productive soils and enhanced the risk of land abandonment in low-productive areas. This spatial segregation of agricultural land use was also shown by Bolliger et al. (2007) for Switzerland and by Miettinen et al. (2004) for Finland. Consequently, the scenario predicted intermediate values for habitat richness and habitat evenness. Decoupling transfer payments from production (Scenario 2) makes mulching more profitable than all other management systems on low-productive sites, as farmers still receive a uniform area payment. This resulted in a landscape dominated by grassland with low values of all habitat diversity indices. Similar trends of grassland expansion were also predicted by Schmid et al. (2007) for Austria. Phasing out transfer payments (Scenario 3) finally led to a complete abandonment or afforestation of agricultural land and extremely low values in all habitat diversity indices. Although this Scenario 3 result appears extreme and unrealistic, several studies confirmed that liberalisation would increase the risk of large-scale land abandonment in mountainous landscapes (e.g. Bolliger et al., 2007; Lundström et al., 2007; Verburg et al., 2006). Thus, our scenario results indicate the importance of transfer payments to prevent cessation of agricultural production in landscapes with mostly unfavourable conditions for agriculture. However, transfer payments alone may not prevent the ongoing homogenisation in these landscapes. Agri-environmental schemes offered in Pillar Two of the CAP may provide additional financial incentives to maintain small-scale mosaics of arable land, grassland, and fallow land habitats. Currently, however, these schemes do not explicitly focus on the creation and management of farmland habitat diversity (Benton et al., 2003; Concepción et al., 2008).

7.2 Database and methodology

The quality and outcome of landscape change analyses is highly influenced by the underlying data sets and the implemented methods. In the following, some strengths and limitations that may be related to the used data sets and methods are discussed.

The feasibility of our analyses primarily depends on appropriate data sets, their quality, and error propagation. Problems encountered at any of these levels can result in lower prediction success. In our analyses on agricultural landscape change (Chapter 4, 5 and 6), we used land-cover data from different sources, a DEM, and soil data. Although the assessment of accuracy and errors in spatial data like land-cover data (e.g. Bach et al., 2006; Foody, 2002; Wickham et al., 2004) or DEM (e.g. Bolstad and Stowe, 1994; Holmes et al., 2000; Wechsler and Kroll, 2006) has received considerable attention in recent research, it has not yet become a standard to state positional and thematic accuracy of a data set (Bach et al., 2006). Hence, we do not know the accuracy of the DEM and soil data used. Visual screening for extreme values, however, gave us confidence of sufficient data quality. GIS-based operations like the conversion of different formats (i.e. vector to raster) or the intersection of different thematic layers may further propagate errors in spatial data (Heuvelink, 1998) and may thus influence the quality of the outcome. Unfortunately, however, general, integrated practical tools for statistical error propagation are still missing in GIS (Burrough and McDonnell, 2000).

Agricultural statistics and satellite images enabled us to identify types of land-cover patterns and dynamics at the scale of districts (Chapter 4). The combined use of land-cover data from census and remote sensing required a prior equalisation of their thematic content and spatial entity. This must be taken with some precautions since data equalisation always involves a loss of information due to spatial and thematic aggregation of the sources to the smallest common denominator (Petit and Lambin, 2002). In our study, the detailed information on grassland given in the agricultural statistics was aggregated into the main category 'grassland', which accords to the lowest common class of the satellite-derived data. Furthermore, satellite-derived data available at a fine resolution of raster cells were reduced to its mere compositional components at the district scale, which corresponds to the comparatively coarse resolution of the agricultural statistics. Thus, the aim of an area-wide landscape classification was achieved at the expense of thematic and spatial resolution. Landscape classification according to recent and historic land-cover data was based on a k-means cluster analysis, which has proven in previous research to be a simple and workable procedure for the definition of homogeneous land units and temporal dynamics (e.g. Bernert et al., 1997; Bunce et al., 1996; Hietel et al., 2004; Simmering et al., 2006).

The analysis of grassland age (Chapter 5) was based on stratified random sampling, multitemporal aerial photograph interpretations, and a spatial extrapolation. Stratified random sampling has been increasingly used in modern surveys to maximise the efficiency of landscape ecological assessment while focussing on maximum variation and representativeness of sampling (Goedickemeier et al., 1997; Knollová et al., 2005). Thus, stratified random sampling permitted to systematically outline the spatial heterogeneity within the study area, regarding TLPDs at the district scale and physical attributes at the patch scale, and to randomly select grassland patches for subsequent multitemporal aerial photograph interpretation. Aerial photographs were often used to detect land-cover changes at high spatial resolutions (e.g. de Blois et al., 2001; Ihse, 1995; Pan et al., 1999; Ruuska and Helenius, 1996). However, aerial photographs provide only arbitrary snapshots in time. In our study, the interpretation depends on six snapshots from the period 1953 to 2001, since further aerial photographs were not available. Temporary alterations of cropland and grassland use between two snapshots can thus not be detected by using exclusively aerial photographs. Additional time-dependent patch characteristics like soil pH (Breuer et al., 2006; Waldhardt and Otte, 2003) or local farmers' knowledge on past management (Calvo-Iglesias et al., 2006a; Robertson and McGee, 2003) may diminish this uncertainty. Further, visual misinterpretations might be a source of under- or overestimated grassland age. However, due to the variation in tonal contrast and its specific texture, grasslands in Central Europe can be readily identified in black and white aerial photographs (Albertz, 1991; Schneider, 1974). Thus, potential misinterpretations may be minimal. The interpretation of multitemporal aerial photographs facilitated the calculation of grassland type-specific age probabilities, which we then generalised via direct extrapolation to the district scale. The challenges of direct extrapolation lie in a correct definition of the spatial and temporal heterogeneity of the fine-scale information (i.e. grassland age) and in an accurate integration and aggregation of this heterogeneity to the broader scale (King, 1991). By aggregating the investigated grassland age into three classes, we considered successional stages of grasslands (cf. Austrheim and Olsson, 1999; Waldhardt and Otte, 2003), which may be expected to be of ecological relevance after abandonment of arable land. A subsequent comparison of our extrapolation results with reference data showed a satisfying conformity for the test area Erda, which underwent rather moderate land-cover changes in the past. However, over- and underestimation up to 21% were detected for the test area Steinbrücken and Eibelshausen characterised by extensive land-cover changes in the past. A better assessment within these areas might be attainable by increasing the rather low number of investigated patches. However, a larger sample size and thus a higher certainty would also increase the amount of work and costs and may thus be only practicable for single grassland types.

The analysis of the effects of transfer payment schemes on habitat diversity (Chapter 6) was based on three scenario predictions generated by an agro-economic land-use model. In this context, it is important to note that scenario predictions remain unlikely future landscape realisations. All models undoubtedly involve some uncertainties and limitations (Rounsevell et al., 2006). Since scenarios are a product of their time and thus consider factors deemed influential at that time (Audsley et al., 2006), exogenous variables like the rate of future technological innovations or price developments can differ more notably in the future than expected. Hence, estimates of their respective rates may be afflicted with uncertainties. Within these caveats however, scenario-based modelling is considered as a useful tool for the study of potential effects on the environment (e.g. Verburg et al., 2002; Busch, 2006; Santelmann et al., 2006; Sheate et al., 2008). Due to potential uncertainties and limitations of the scenario approach and the type of land-use model, the comparison of the three modelled farmland patterns with the observed realisation of the farmland pattern in 1995 was not done in the temporal dimension, i.e. we did not analyse rates and directions of habitat change. Instead, we concentrated on the analysis and comparison of four states of farmland habitat patterns.

7.3 Synthesis and perspectives

The results of our studies confirmed a general trend of abandonment of cultivation for the entire study area. However, this trend is characterised by remarkable differences in the magnitude and the direction of past and modelled land-cover change, which are related to the local spatial pattern of the underlying environmental conditions. Also, the spatial distribution of grassland age reflects this high spatial heterogeneity. The payment of subsidies may prevent the cessation of agricultural production. However, transfer payments alone will not fully counteract the general trend of homogenisation in marginal landscapes. Further incentives in agricultural policy are needed that specifically favour a mosaic of arable land, grassland, and fallow land. Overall, it can be concluded that marginalisation is a highly evident process in landscapes with mostly unfavourable conditions for agriculture. However, against the background of the current increase in global food and energy demands and the subsequent rise in prices for crops, a new trend of agricultural intensification is likely to affect also such landscapes. Crop production may become profitable even on less favourable areas. The portion of arable land may increase and thus reverse the current trend of marginalisation.

In any case, landscape change analysis in marginal landscapes will continue to be a challenging research area, especially with respect to future developments in agriculture.

This thesis provided methods, which may be useful in a variety of contexts. The combination of remote sensing data with agricultural statistics enables the identification of patterns of current land cover and land-cover dynamics. These may be used as strata in sampling designs that aim to include land-cover dynamics as a factor of interest. The identification and characterisation of spatially and temporally heterogeneous land-cover patterns in marginal cultural landscapes will further support research that aims to outline areas potentially sensitive to future land-cover changes. The combination of an a-priori two-stage landscape stratification with conventional aerial photograph interpretation of selected patches, and the subsequent spatial extrapolation of the determined grassland age sidesteps the shortcomings caused by the lack of feasible data on spatially explicit land-cover change. By utilising inexpensive, commonly available data combined with well established techniques, our 3-step approach may be applied to other marginal agricultural landscapes under study. Further, our approach is suited for application in landscape models of various disciplines, which rely on large-scale information on grassland age. For instance, grassland age can be used as indicator for the prediction of vascular plant species richness in mosaic landscapes (Waldhardt and Otte, 2003; Waldhardt et al., 2004). Moreover, the approach may be easily adapted to other land-cover types such as fallow land, whose phytodiversity is also dependent on age (Simmering et al., 2001). Further work in the field of agricultural landscape change analysis may profit from an ongoing technical progress in GIS and an improved availability of feasible and consistent land-use and land-cover data in a high spatial and temporal resolution. Recent developments in this field seem quite promising, as several new approaches for the processing of landscape information (e.g. Gardner et al., 2008) or newly established information systems like the GIS-based Land Parcel Identification Systems (LPIS) within the Integrated Administration and Control Systems (IACS), which include spatial explicit and high-resolution information on agricultural parcels all over Europe (Perez, 2005).

Summary

Since the end of World War II, marginal European landscapes with unfavourable environmental conditions for cultivation have experienced severe land-use changes. In many cases, large portions of arable land have been successively abandoned in favour of grassland or fallow land. This general trend of marginalisation in turn affected ecological landscape functions and processes with far-reaching consequences for biodiversity and natural resources. Furthermore, the land-use pattern in marginal European landscapes is expected to undergo further major changes in the future, particularly in the course of EU agricultural policy (**Chapter 1**). Given this background, this multiple-paper thesis (A) analysed agricultural landscape change in a marginal agricultural landscape and (B) developed methods that may support landscape change research at multiple spatio-temporal scales. Both aims were addressed in three studies (**Chapter 4, 5 and 6**) and separately discussed in the general discussion section (**Chapter 7**). Our study area was the Lahn-Dill Highlands (1270 km²), a marginal agricultural landscape in Hesse (Germany) with a pronounced land-use change in the past decades (**Chapter 2**). The methods used in the studies were summarised in **Chapter 3**.

In the first study (**Chapter 4**), we developed an approach to identify types of land-cover patterns and dynamics (TLPDs) at the rural district scale. By the combination of recent satellite data with historic agricultural statistics, and the application of k-means cluster analysis, we identified six TLPDs and characterised their physical settings. We found a general trend of abandonment of cultivation at the landscape scale, which is governed by significant differences in current land-cover patterns and the directions of land-cover change at the district scale: In the eastern part of the area, where elevation is low, and the proportions of steep slopes and dry soils are small, land cover remained relatively stable. Slight to dramatic changes occurred, in contrast, in the remaining districts with comparatively unfavourable conditions for cultivation.

In the second study (**Chapter 5**), we developed a 3-step methodological approach to systematically assess the spatial distribution of grassland age in a marginal agricultural landscape. The approach is based on the combination of an a-priori two-stage landscape stratification with conventional aerial photograph interpretation of selected patches, and the subsequent spatial extrapolation of the determined grassland age. Results proved that our approach provides a realistic estimation of grassland age at the scale of districts and over a

time period of five decades. We found that the derived probabilities of grassland age classes are specific for grassland types in areas with a homogenous pattern of land-cover change. Furthermore, the results indicated a predominance of old grassland patches (>47 years). Occurrences of mid-aged grassland (18-47 years) were concentrated in districts with a pronounced land-cover change in this time period, whereas young grassland (<18 years) is apparently evenly distributed across the study area.

In the third study (**Chapter 6**), we analysed the potential effects of three alternative transfer payment schemes on the farmland habitat diversity in a marginal agricultural landscape. We defined (1) a scenario with direct transfer payments coupled to production, (2) a scenario with direct transfer payments decoupled from production, and (3) a scenario phasing out all direct transfer payments. We characterised habitat diversity with three indices: habitat richness, evenness, and rarity. The habitat pattern in 1995 served as reference for comparison. All scenarios predicted a general trend of homogenisation of the farmland habitat pattern, yet to a differing extent. Transfer payments coupled to production (Scenario 1) supported spatially segregated land use with fallow land primarily in low-productive areas and arable land use in the more productive sites. The scenario predicted intermediate values for habitat richness and habitat evenness. Decoupling transfer payments from production (Scenario 2) favoured grassland as the most profitable farming system. This led to a grassland-dominated landscape with low values of all habitat diversity indices. Phasing out transfer payments (Scenario 3) resulted in complete abandonment or afforestation of agricultural land and extremely low values in all habitat diversity indices. Scenario results revealed that the payment of subsidies may prevent cessation of agricultural production, but may not fully counteract the homogenisation in marginal landscapes.

Zusammenfassung

Seit Ende des Zweiten Weltkriegs ist in agrarstrukturell und standörtlich benachteiligten, marginalen Kulturlandschaften ein tief greifender Nutzungswandel zu beobachten, bei dem meist der Ackerbau zunehmend an Bedeutung verliert und durch Grünlandnutzung oder Brachland ersetzt wird. Dieser allgemeine Trend zur Marginalisierung wirkte sich in der Folge nachhaltig auf Landschaftsfunktionen und Prozesse mit weit reichenden Konsequenzen für Biodiversität und natürliche Ressourcen aus. Im Zuge der EU-Agrarpolitik sind auch künftig weitere Veränderungen der Nutzungsmuster in marginalen Kulturlandschaften zu erwarten (**Kapitel 1**). Vor diesem Hintergrund wurden in dieser kumulativen Arbeit (A) der Agrarlandschaftswandel in einer marginalen Kulturlandschaft untersucht und (B) Methoden zur Untersuchung des Landschaftswandels auf multiplen raum-zeitlichen Skalen entwickelt. Beide Arbeitsschwerpunkte wurden in drei Teilstudien (**Kapitel 4, 5 und 6**) behandelt und in der zusammenfassenden Diskussion (**Kapitel 7**) getrennt voneinander diskutiert. Als Untersuchungsregion diente das Lahn-Dill-Bergland (1270 km²), eine marginale Kulturlandschaft in Hessen, welche in den vergangenen Jahrzehnten einem erheblichen Nutzungswandel unterlag (**Kapitel 2**). Die in den hierzu vorgestellten drei Studien angewandten Methoden wurden in **Kapitel 3** zusammengefasst.

In der ersten Studie (**Kapitel 4**) wurde eine Methodik zur Identifizierung von Typen der Nutzungsmuster und der Nutzungsdynamik (TLPDs) auf Gemarkungsebene entwickelt. Mit Hilfe von aktuellen Satellitendaten und historischen Daten aus der Agrarstatistik sowie einer K-Means Clusteranalyse konnten sechs TLPDs identifiziert und hinsichtlich ihrer physischen Eigenschaften charakterisiert werden. Auf Landschaftsebene konnte ein allgemeiner Trend zur Aufgabe der ackerbaulichen Nutzung nachgewiesen werden. Signifikante Unterschiede im gegenwärtigen Nutzungsmuster und in der Richtung des Nutzungswandels zeigten sich hingegen auf Gemarkungsebene: Im östlichen Teil des Lahn-Dill-Berglandes, wo die Höhenlage, der Anteil steiler Hänge und trockener Böden gering sind, blieb die Landnutzung nahezu unverändert. Geringe bis stark ausgeprägte Veränderungen jedoch traten in den übrigen Gemarkungen mit vergleichsweise ungünstigen Bedingungen für die Kultivierung auf.

In der zweiten Studie (**Kapitel 5**) wurde eine 3-stufige Methodik zur systematischen Abschätzung der räumlichen Verteilung von Grünlandalter in marginalen Kulturlandschaften entwickelt. Die Methodik basiert auf einer vorausgehenden zweistufig geschichteten

Landschaftsstratifizierung, einer Luftbildinterpretation ausgewählter Grünlandflächen und einer nachfolgenden räumlichen Extrapolation des bestimmten Grünlandalters. Die Ergebnisse zeigten, dass die entwickelte Methode für eine realistische Abschätzung des Grünlandalters auf Gemarkungsebene und über einen Zeitraum von fünf Jahrzehnten geeignet ist. Die abgeleiteten Wahrscheinlichkeiten der Grünlandaltersklassen sind spezifisch für Grünlandtypen in Gebieten mit einheitlichem Muster des Nutzungswandels. Die Ergebnisse zeigten außerdem ein Vorherrschen von altem Grünland (>47 Jahre). Vorkommen von mittelaltem Grünland (18-47 Jahre) konzentrieren sich auf Gemarkungen mit starker Nutzungsdynamik in diesem Zeitraum, während junges Grünland (<18 Jahre) offensichtlich gleichmäßig über das Untersuchungsgebiet verteilt ist.

In der dritten Studie (**Kapitel 6**) wurden die potentiellen Auswirkungen von drei alternativen Transferzahlungen auf das landwirtschaftliche Habitatmuster einer marginalen Kulturlandschaft abgeschätzt. Für die Untersuchungen wurden drei Szenarien definiert: (1) Ein Szenario mit direkten Transferzahlungen gekoppelt an die Produktion, (2) ein Szenario mit direkten Transferzahlungen entkoppelt von der Produktion und (3) ein Szenario ohne Zahlung von direkten Transferzahlungen. Habitatdiversität wurde mit drei Indizes charakterisiert: Habitatreichtum, Habitat-Evenness und Habitat-Rarität. Das Habitatmuster von 1995 diente als Vergleichsgrundlage. Alle Szenarien sagen einen allgemeinen Trend zur Homogenisierung des Habitatmusters voraus, jedoch in unterschiedlichem Ausmaß. An die Produktion gekoppelte Transferzahlungen (Szenario 1) begünstigten eine räumliche Trennung der Landnutzung mit einer Aufgabe der landwirtschaftlichen Nutzung insbesondere in schwach produktiven Gebieten und Ackernutzung in stark produktiven Gebieten. Entsprechend wies das Szenario mittlere Werte für den Habitatreichtum und die Habitat-Evenness auf. Eine Entkoppelung der Transferzahlungen von der Produktion (Szenario 2) begünstigte Grünland als profitabelstes Landnutzungssystem. Dies führte zu einer von Grünland dominierten Landschaft mit geringen Werten aller Habitatindizes. Bei Einstellung der Zahlung von Transferzahlungen (Szenario 3) ist von einer völligen Aufgabe der landwirtschaftlichen Nutzung bzw. einer Wiederaufforstung und extrem niedrigen Werten aller Habitatindizes auszugehen. Diese Ergebnisse zeigen, dass die Zahlung von Prämien eine Aufgabe der landwirtschaftlichen Produktion verhindern kann. Sie kann jedoch nicht einer Homogenisierung marginaler Kulturlandschaften entgegenwirken.

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