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Characterizing social–ecological units to inform biodiversity conservation in cultural landscapes

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ABSTRACT

Aim Cultural landscapes and their biodiversity are threatened by land use changes and the abandonment of traditional farming techniques. Conceptualizing cultural landscapes as social–ecological systems can be useful to develop strategies for biodiversity conservation. First, this study aimed to develop a typology of social–ecological units based on land use patterns. Second, we sought to relate this typology to biophysical and socio-demographic drivers as well as to biodiversity outcomes.

Location Southern Transylvania (Romania).

Methods We developed a typology of villages in Southern Transylvania based on land use data. We collected species richness data for plants, butterflies and birds, modelled local richness data for each village and related these values to the village typology. Also, we related village typology to biophysical and socio-demographic variables.

Results We identified four types of villages that showed distinct species richness patterns. Bird richness was highest in forest-dominated and mixed-land use villages; plant richness was highest in pasture-dominated villages; and butterfly richness was high in arable-dominated, mixed-land use and pasture-dominated villages. The four types of villages had distinct topographic characteristics and also differed in terms of ethnic composition, migration patterns and geographic location. Drawing on a combined understanding of social–ecological variables, different conservation actions could be prioritized for each of the four village types.

Main conclusions Applying social–ecological approaches has the potential to inform biodiversity conservation in cultural landscapes. Social–ecological typologies can improve our understanding of complex systems and provide useful input for the development of effective strategies for biodiversity conservation.

Keywords
farmland biodiversity, human–environment systems, landscape sustainability science, traditional farming landscapes.

INTRODUCTION

Cultural landscapes often contain semi-natural ecosystems and provide habitat for native species (Bignal & McCracken, 2000; Ramankutty *et al.*, 2008; Liu *et al.*, 2013). Therefore, they are an important priority for biodiversity conservation (Bignal & McCracken, 2000; Tscharntke *et al.*, 2005). In particular in many European landscapes, the long history of low-intensity land use has led to a remarkable cultural and biological diversity (Barthel *et al.*, 2013). However, the loss of (semi-)natural vegetation, increasing inputs of fertilizers and pesticides, and the cessation of traditional farming methods have caused the loss of biodiversity throughout Europe (Weibull *et al.*, 2000; Benton *et al.*, 2003; Geiger *et al.*, 2010).

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While the general pattern of agricultural intensification driving biodiversity loss has been widely documented (Foley *et al.*, 2005), the locally specific outcomes are shaped by complex interactions between biophysical conditions, socio-economic conditions and historical legacies (Wilbanks & Kates, 1999). One promising avenue to better understand such interactions is to conceptualize cultural landscapes as social–ecological systems, that is dynamic systems with interacting social and ecological components (Berkes *et al.*, 2003). Given historically tight interconnections between ecological and social system components in cultural landscapes (Farina, 2000; Fischer *et al.*, 2012), the relative dearth of simultaneous consideration of ecological and social system components via integrated analyses is a critical shortcoming in research to date (but see Alessa *et al.*, 2009; Cumming *et al.*, 2015). Methods to do so need to simplify the complex nature of social–ecological systems, but at the same time preserve locally important contextual information (Lüdeke *et al.*, 2004). To this end, we employed the approach of developing social–ecological typologies. This approach has been applied previously at the scale of Europe (Levers *et al.*, 2016), for patterns of global land use (Václavík *et al.*, 2013) and global change (Lüdeke *et al.*, 2004) and to assess the vulnerability of social–ecological systems (Kok *et al.*, 2016). Here, we based our typology on land use patterns, because land use represents the most important interaction between the humans and their environment in farming landscapes (Wu, 2013) and land use intensity and change have been identified as the main drivers of biodiversity loss in agricultural systems (Stoate *et al.*, 2001). In our study, we focused on the region of Southern Transylvania (Central Romania), which harbours some of the most notable cultural landscapes in Europe (Akeroyd & Page, 2006; Palang *et al.*, 2006). In this region, both land abandonment and land use intensification pose threats to biodiversity (Hanspach *et al.*, 2014; Sutcliffe *et al.*, 2015). Specifically, we developed a typology of villages as social–ecological units, and investigated village-level biodiversity outcomes in relation to underlying biophysical and socio-demographic drivers (Fig. 1). The resulting typology contributes to a better understanding of social–ecological relationships and can help to inform conservation policy in our study system.

METHODS

Study area

The study area comprised an area of 7440 km^2 in Southern Transylvania (Romania; Fig. 2a). It is characterized by a hilly terrain at altitudes between 230 and 1100 m. The main land cover types are arable land (37%), forest (28%), grassland (24%) and settlements (4%) (EEA, 2006). Semi-subsistence small-scale farming is still common in the region. This low-intensity farming type has maintained a cultural landscape that is rich in biodiversity. The study area is populated by Romanian (65%), Hungarian (28%), Roma (6%) and Saxon (1%) ethnicities (INS, 2012).

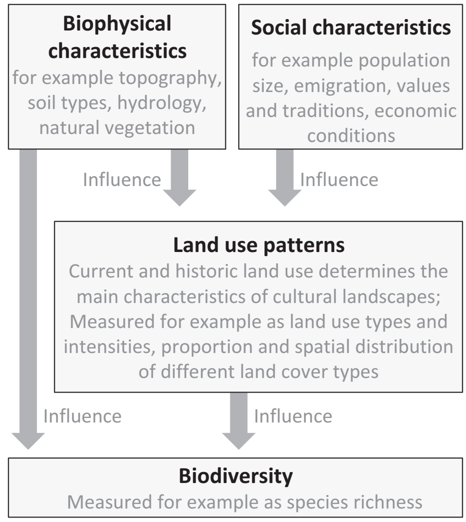

Figure 1 Biophysical and socio-demographic variables are key determinants of land use patterns in cultural landscapes. Land use patterns, in turn, strongly influence patterns of biodiversity. This paper first developed a typology of social–ecological units based on land use patterns. Second, it investigated species richness in the villages and how village-specific richness profiles relate to the land use typology. Third, it investigated which biophysical and socio-demographic variables are related to the land use typology.

We used villages and their surrounding landscapes as the basic social–ecological unit of analysis because this represents 'the basic unit of European landscapes' (Angelstam *et al.*, 2003) – meaning it is typically the smallest administrative entity within which land is being governed and land use realized. Thus, villages represent miniature social–ecological systems that traditionally exhibited tight coupling of humans and the environment. The study area contained 448 villages and their surrounding land (mean area = 16.1 km^2; standard deviation = 10.5 km^2). We delineated the area belonging to a given village using a cost–distance algorithm that allocated each pixel to the village with the lowest travel cost to this pixel (horizontal distance penalized by slope; implemented in ArcGIS, ESRI, Redlands, California, USA). For the sake of simplicity, we henceforth use the term 'village' when we refer to a given village including its surrounding landscape.

While most analyses were completed for all 448 villages, we performed in-depth studies on biodiversity and socio-demographic characteristics for a subset of 30 villages (Fig. 2a; for details on the steps of analysis, see Fig. S1). These 30 villages covered the main biophysical and social gradients in the study area (see Fig. S2 for comparison of socio-demographic variables). They were selected randomly from all 448 villages, but stratified to cover: (1) the full gradient in terrain ruggedness (measured as the variation in altitude within a given village) and (2) different levels of conservation status (no protection, protection under the EU Birds Directive, protection under the EU Habitats Directive). Notably, conservation status has only recently been imposed, and management plans have not been implemented to date.

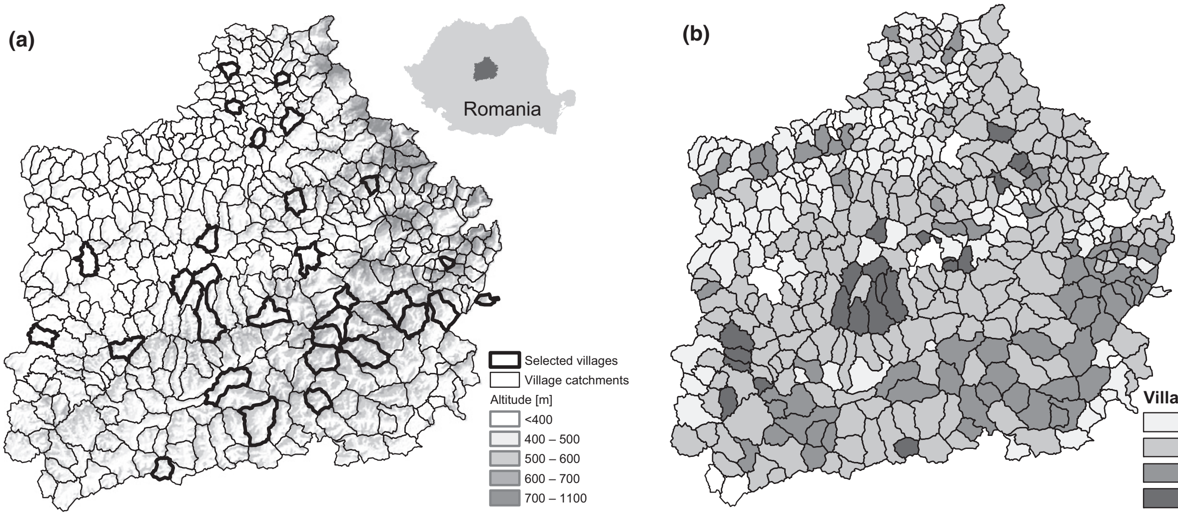

Figure 2 Overview over the study area. Panel (a) shows village catchments including the subset of 30 villages that were used for biodiversity sampling and socio-demographic analysis. Panel (b) shows the classification of villages into four land cover clusters.

Biophysical, socio-demographic and land use data

To describe individual villages, we noted key variables that are potentially important drivers influencing biodiversity conservation. First, we considered biophysical variables that described the gradient in topography, and second, a range of socio-demographic characteristics. Third, we quantified the land cover characteristics of an individual village as a main feature of interaction between the social and the ecological systems. Topographic and land use variables were chosen because these were identified to be important drivers of species distribution and community composition in the region (Dorresteijn *et al.*, 2015; Loos *et al.*, 2015). Socio-demographic variables were informed by stakeholder input and previous research that investigated regional social–ecological system dynamics (Hanspach *et al.*, 2014).

Topographic data (calculated for all 448 villages) included altitude, terrain ruggedness (standard deviation of altitude), proportion of village area with an inclination of < 5°, between 5° and 10° and above 20°, terrain wetness index (TWI) (Fischer *et al.*, 2010) and heatload [cos (aspect −225) *tan (inclination)]. All topographic variables were based on a digital elevation model with a 30 m resolution (ASTER GDEM v2, NASA, Washington D.C., USA).

Socio-demographic data (for the subset of 30 villages) were derived from commune-level statistics for the year 2010 from the National Institute for Statistics (INS, 2012) – village-level data were not available for the whole study area. Communes are the smallest administrative units in the study region and consist of groups of adjacent villages. The communes studied here contained, on average, four villages. For approximately one-quarter of the study area, we had data on ethnic composition at both the village and the commune level, and these data suggested that commune-level data appeared to be a good proxy for village-level characteristics (Pearson's correlation village and commune level = 0.83, $P < 0.001$). Socio-demographic data ultimately used included total population size, proportions of the main ethnic groups (Romanians, Hungarians, Roma and Saxons), unemployment rate, net migration, as well as the proportion of pupils in the population. In addition, we added information on the remoteness of a given village (calculated as travel time by car to the next town with > 20,000 inhabitants based on speed estimates for different road types), and a dummy variable for whether a given village contained the town hall of the commune it belonged to (or not).

Finally, land cover data (considered for all 448 villages) considered land cover types, the amount of woody vegetation and landscape heterogeneity. First, to describe differences in dominant land use, we calculated the proportions of the main land cover types for each village [forests, arable land including heterogeneous agricultural areas, pastures, orchards (i.e. permanent crops)] based on the CORINE land cover map 2006 (EEA, 2006). Second, we calculated the proportion of woody vegetation separately for (1) the whole village, (2) within arable land and (3) within pasture. This was based on a map of woody vegetation derived by a supervised classification of the monochromatic channels of SPOT 5 data at a 10 m × 10 m resolution using a support vector machine algorithm (Hanspach *et al.*, 2014). Third, we calculated three measures of landscape heterogeneity. These were (1) the standard deviation in the panchromatic channel of SPOT 5 satellite imagery at a 2.5 m × 2.5 m resolution (CNES 2007, Distribution Spot Image SA) across the entire area, arable land and pasture within a given village, (2) the Simpson diversity index (SIDI); and (3) edge density (ED). The latter two were both based on land cover data from CORINE 2006 (level 2) and were calculated for each village in FRAGSTATS 4.0 (McGarigal *et al.*, 2012). Unlike CORINE-derived measures, the heterogeneity measure based on SPOT data effectively captured small-scale patterns of heterogeneity (e.g. from smaller field sizes, field margins, unused land or scattered trees). Thus, heterogeneity links to different levels of land use intensity in the study area. Similarly, generating a woody vegetation layer at a very high resolution was guided by the desire to capture the presence of shrubs and trees at a local scale. Low-intensity farming in Southern Transylvania is characterized by a high frequency of woody vegetation along field margins or even within fields, whereas this is lacking in more intensively farmed areas. At the same time, the amount of woody vegetation can be an indicator of farmland abandonment as woody vegetation increases through shrub encroachment.

Biodiversity data

In the subset of 30 villages, we conducted detailed biodiversity surveys. We surveyed the number of bird, butterfly and plant species in 150 circular 1-ha sites (60 in pastures, 60 in arable land and 30 in forest) during spring and summer of 2012. These taxa were chosen because they represent a wide range of ecological requirements and functions, and are sensitive to environmental change (Thomas *et al.*, 2004).

Within a given village, survey sites in farmland (i.e. arable land and pasture) were placed using stratified random selection. Stratification was performed by fully covering gradients in landscape heterogeneity (measured as the variation in the panchromatic channel of SPOT 5 satellite imagery in a 1-ha circle) and amount of woody vegetation cover (Hanspach *et al.*, 2014), separately for pasture and arable land. Selection of forest sites was random. Typically, in a given village, we selected two sites in arable land, two sites in pasture and one site in the forest.

Bird richness was estimated by conducting three 10-min point counts of singing males within each site (Loos *et al.*, 2014b). Butterfly richness was assessed by conducting standard Pollard walks (Pollard & Yates, 1993) of 200 m length within a given site, repeated at four different times (Loos *et al.*, 2014a). Plant richness was determined using eight randomly selected 1-m^2 squares within each 1-ha site (Loos *et al.*, 2015).

Species richness modelling

We separately modelled the site-level (i.e. local scale, 1 ha) species richness of birds, butterflies and plants using generalized linear mixed-effects models (GLMER) with Poisson's error distributions using the LME4 package in the R environment (Bates *et al.*, 2014). For each model, we included village as a random effect, as well as an observation level random effect to account for overdispersion. The full models contained the following fixed effects (at the site level): heterogeneity (linear and quadratic terms), woody vegetation cover (linear and quadratic terms), TWI, heatload, main land cover type (forest, pasture, arable), as well as interaction terms between land cover type and heterogeneity, and land cover type and woody vegetation cover. Models were simplified to minimal adequate models using backwards selection using likelihood-ratio tests (Zuur *et al.*, 2009).

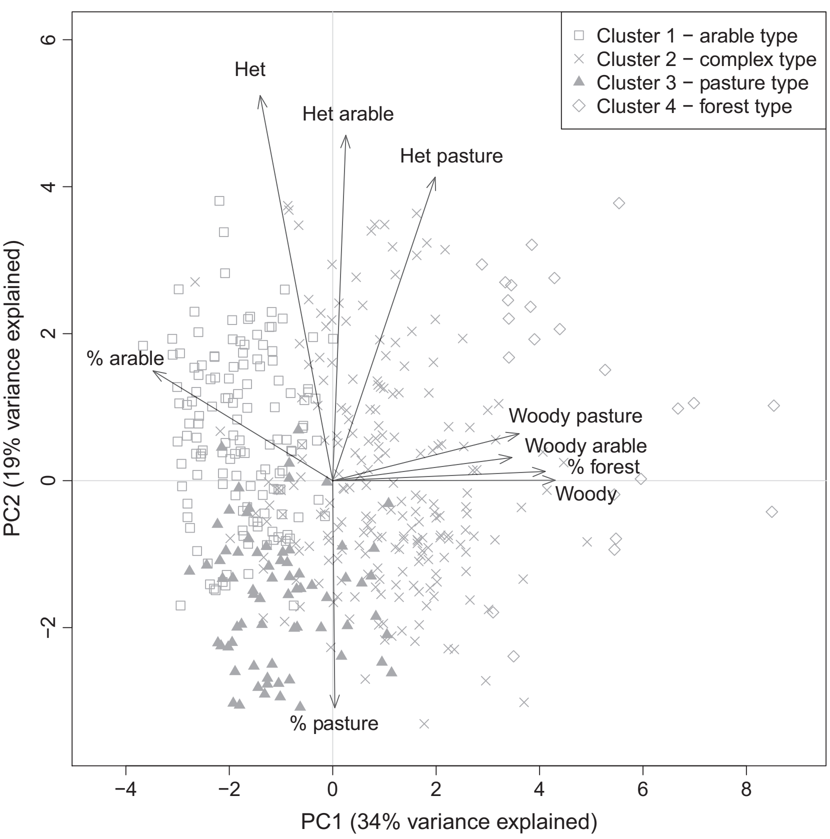

Figure 3 Typology of social–ecological units as derived from a cluster analysis of land cover variables. The typology is presented in an ordination plot based on a principal component analysis of the variables underpinning the typology (woody pasture – woody vegetation cover in pastures; woody arable – woody vegetation cover in arable land; woody – woody vegetation cover in the whole village; het past – heterogeneity in pastures; het arable – heterogeneity in arable land; het – heterogeneity in the whole village).

We validated our models by comparing predicted values against an independent-test data set. Test data consisted of 35 randomly selected sites (17 in arable, 13 in pasture and 5 in forests) distributed across five villages, at which richness data on all three groups had been collected in 2011 (Loos *et al.*, 2014b). Agreement between modelled and observed richness values was assessed using Pearson's correlation coefficients.

Based on the estimated fixed effects, we predicted species richness to the whole study area, excluding areas that were outside of the calibration range (i.e. having environmental conditions beyond the range of what was surveyed). We visualized the modelled richness per hectare by plotting the cumulative density distribution of modelled species richness for each of the villages in the study area. For a given village and taxonomic group, we defined each of these density distribution curves as a 'species richness profile'. Our design did not allow for the meaningful estimation of beta-diversity or other biodiversity measures at the village level, although we acknowledge that investigating such patterns would be interesting.

Creating a village typology and relating it to biodiversity and biophysical and socio-demographic variables

Our objective was to develop a typology of social–ecological units and identify context-specific threats and opportunities for biodiversity conservation. This typology was intentionally based only on variables that describe current land use, which we considered to represent the immediate, or proximate, interface between the social and the ecological systems (Wu, 2013). In contrast, we considered topographic and socio-economic variables as being more distal in their relationship to biodiversity (Austin, 2007). To work through the complexity of different types of potentially interesting relationships, we divided further analyses into four steps (Fig. 1).

Table 1 Results of the species richness models. Estimates are given for fixed and random effects of the minimum adequate models.

Fixed effects	Estimate	SE	*P*-value	Random effects	SD
Bird richness in farmland ($N = 120$)					
Intercept	1.5	0.070	***	Observation	0.017
Woody	0.59	0.072	***	Village	0.000
Land cover type pasture	0.13	0.092			
Woody: land cover type pasture†	−0.26	0.098	**		
Bird richness in forest ($N = 30$)					
Intercept	2.5	0.074	***	Observation	0.000
Heterogeneity	0.14	0.066	*	Village	0.000
Heterogeneity^2	−0.11	0.056	*		
Butterfly richness in farmland ($N = 119$)					
Intercept	2.7	0.057	***	Observation	0.20
Woody	0.13	0.036	***	Village	0.15
Land cover type pasture	0.12	0.062			
Heterogeneity	0.12	0.053	*		
Heterogeneity^2	−0.057	0.028	*		
Land cover type pasture: Heterogeneity	−0.21	0.074	**		
Butterfly richness in forest ($N = 15$)					
Intercept	1.4	0.15	***	Observation	0.21
				Village	0.000
Plant richness in farmland ($N = 115$)					
Intercept	3.6	0.057	***	Observation	0.29
Woody	0.16	0.050	**	Village	0.11
Land cover type pasture	0.48	0.067	***		
Heterogeneity	−0.041	0.036			
Heterogeneity^2	−0.070	0.025	**		
Terrain wetness index (TWI)	−0.077	0.035	*		
Woody:† Land cover type pasture	−0.10	0.064			
Plant richness in forest ($N = 23$)					
Intercept	2.2	0.16	***	Observation	0.000
Heterogeneity	0.054	0.093		Village	0.28
Heterogeneity^2	0.23	0.10	*		
TWI	−0.55	0.18	**		

SE, standard error; SD, standard deviation; *N*, sample size; woody, proportion of woody vegetation.

†Colons indicate interaction effects.

P-values: *< 0.05, **< 0.01, ***< 0.001.

First, we developed a village typology by means of a cluster analysis (agglomerative clustering using Euclidian distances and Wards method; Legendre & Legendre, 1998) on land cover characteristics of the villages (i.e. proportions of main land cover types, the amount of woody vegetation and landscape heterogeneity; for description see above; all variables scaled). We derived four main clusters representing four village types. To aid interpretation, we visualized these clusters along the first two axes of a principal components analysis on the same set of land cover variables.

Second, we related species richness in a given village to this land cover typology. To this end, we calculated the modelled median (per 1 ha) species richness for each village and related it to the village's land cover characteristics. The median was chosen to describe the central tendency in species richness for a given village and was more meaningful than alternative measures (e.g. the mean) because many richness profiles did not follow a normal distribution. The relationship between median richness and village types was then visualized using box-and-whisker plots.

Third, we assessed the relationship of land cover variables with topographic and socio-demographic variables, drawing on the subset of 30 villages. We used a redundancy analysis (RDA; Legendre & Legendre, 1998) with all land cover variables as response variables and all topographic and socio-demographic variables as explanatory variables (all variables scaled). RDA was performed using the function *rda* with default settings in the VEGAN package in R (Oksanen *et al.*, 2014).

RESULTS

We identified two main gradients of land cover characteristics in the study area. The first gradient described the amount of forest cover and woody vegetation in arable land

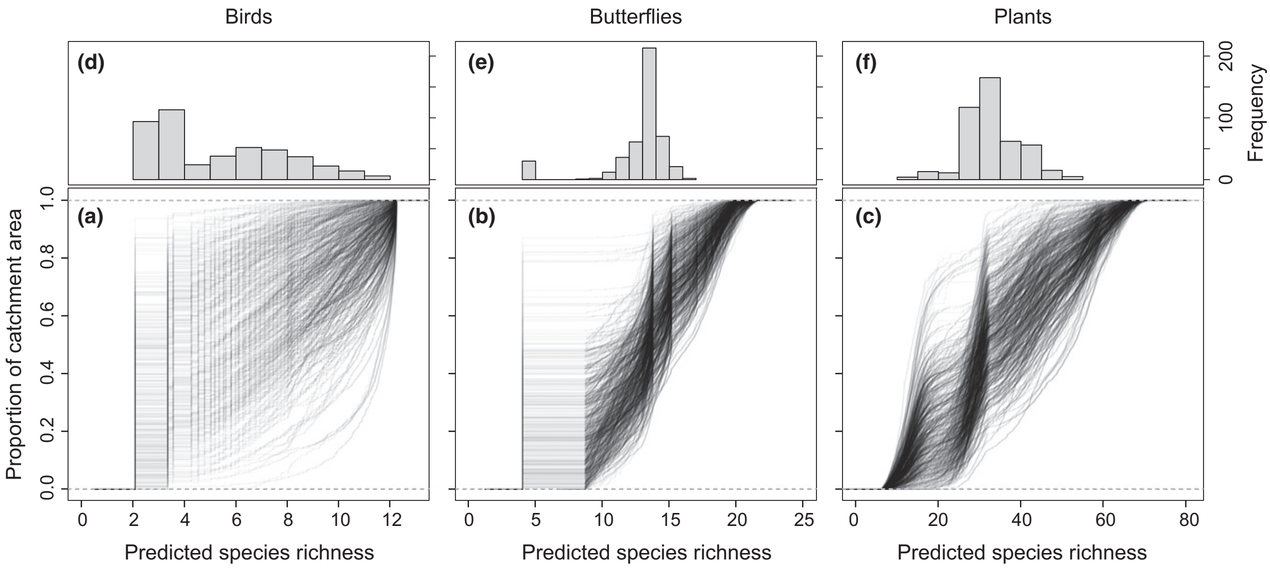

Figure 4 Species richness profiles (a–c) and histograms of median richness (d–f) for all 448 villages in the study area. Each line in the species richness profiles is the cumulative density distribution for a given village and species group. Some villages exhibited high median richness values for a given species group, while others exhibited low median richness values. Median values of richness, in turn, were related to different cumulative density plots (or species richness profiles).

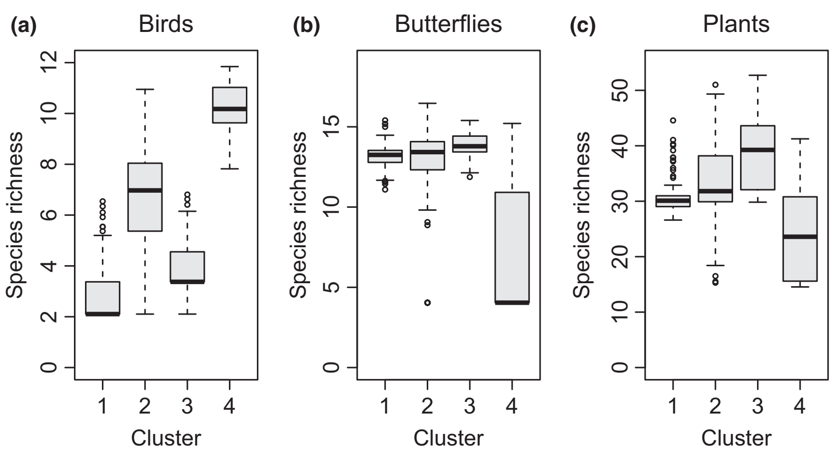

Figure 5 Relationship between village type (i.e. cluster; see also Fig. 3) and modelled median species richness for birds (a), butterflies (b) and plants (c) per village. (Cluster 1 – arable type; cluster 2 – complex type; cluster 3 – pasture type; cluster 4 – forest type.)

and pastures in a given village (Fig. 3). The second gradient was defined by the amount of pasture and land cover heterogeneity in a given village. Based on the cluster analysis on land cover variables, we derived a village typology with four clusters (Figs 2b & 3). Cluster 1 ('arable type') contained villages with large proportions of arable land and relatively low amounts of woody vegetation. Cluster 2 ('complex type') contained villages with a lower amount of arable land and intermediate-to-high amounts of woody vegetation. Cluster 3 ('pasture type') included villages with high proportions of pasture, low heterogeneity and low amounts of woody vegetation. Cluster 4 ('forest type') included villages with very large amounts of woody vegetation and low proportions of arable land.

Species richness modelling resulted in three minimum adequate models (Table 1). A validation against our independent data set showed a high predictive ability of the minimum adequate models (Pearson's *r* of 0.79 for birds, 0.74 for butterflies and 0.63 for plants; see Fig. S3). We thus concluded that our models worked reliably to predict site-level (i.e. 1 ha) richness patterns of birds, butterflies and plant across the entire study area (see Fig. S4).

Predicting richness of the three taxonomic groups across the region resulted in a wide range of richness profiles. Richness profiles varied both between villages and species groups (Fig. 3a–c). In particular for bird richness, the richness profiles of villages were widely spread; that is, some villages had large proportions of land with relatively low bird species richness (and only a few locations with high richness), whereas other villages had high bird species richness throughout most of the village. For butterflies and plants, the distribution of richness profiles and the associated distribution of median richness per village (Fig. 4d–f) were less widely spread.

The median richness per village clearly mapped onto the land cover typology (Fig. 5). Median bird richness was particularly high in villages with a high amount of woody vegetation (complex and forest types in Fig. 3). Median plant and butterfly richness, in contrast, were highest in villages with large proportions of arable land or pastures (arable, complex and pasture types in Fig. 3).

The village typology developed above was also mirrored closely in the RDA (Fig. 6a). Land cover in a given village was strongly related to topographic and socio-demographic characteristics (Fig. 6b), with topography explaining 43% and socio-demographic variables explaining 23% of variation in land cover variables. In particular, villages with high amounts of woody vegetation (complex and forest types) were characterized by steep terrain, larger proportions of Romanians, Saxons and Roma, and higher unemployment rates and emigration. Villages with low amounts of woody vegetation (arable and pasture types), in contrast, had a flatter terrain, high proportions of Hungarians and tended to have higher rates of immigration. Further, villages with high proportions of pasture (pasture type) tended to be at high

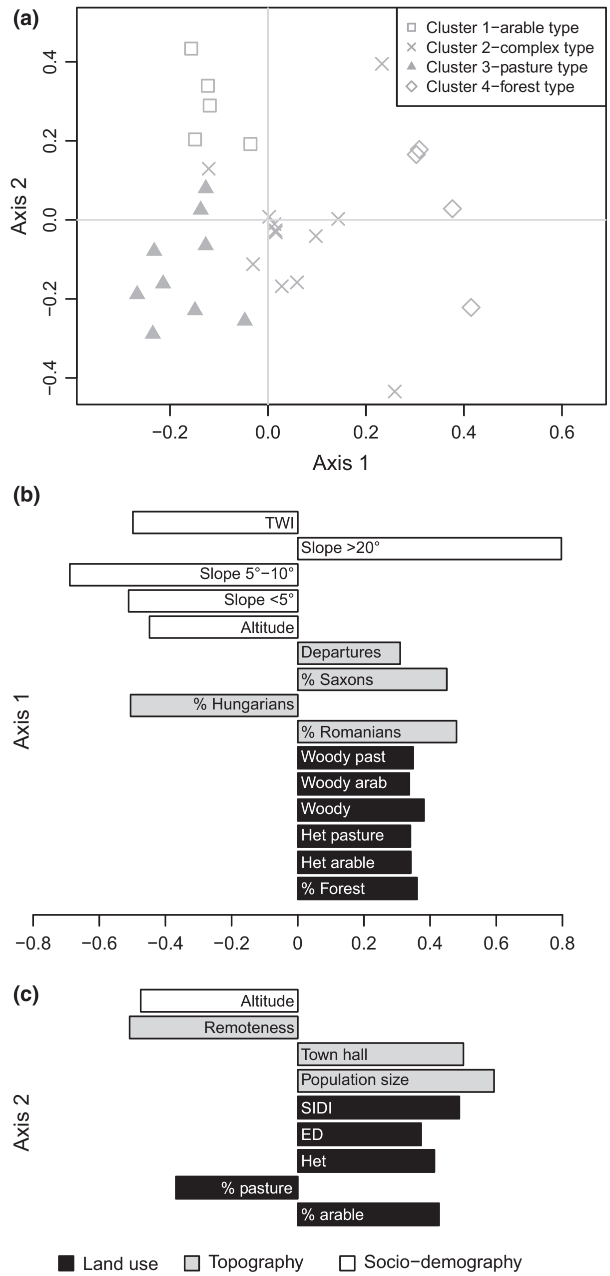

Figure 6 Relationship between village type, land cover, topography and socio-demographic variables as derived from a redundancy analysis (RDA) for a subset of 30 villages. The graphs show the scores for the first two axes of the RDA. First (a), the villages belonging to a certain cluster (from Fig. 3) are displayed along the first two RDA axes. Second (b and c), for both axes the loadings of the variables are displayed. Only variables with scores larger than 0.3 were plotted in panel b. (TWI – terrain wetness index; woody past – woody vegetation cover in pastures; woody arab – woody vegetation cover in arable land; woody – woody vegetation cover in the whole village; het past – heterogeneity in pastures; het arable – heterogeneity in arable land; SIDI – Simpson diversity of land cover types; ED – edge density of land cover types; het – heterogeneity in the whole village).

elevations as well as relatively far from bigger towns. Villages with high proportions of arable land and high heterogeneity (arable type) tended to be large, well connected to towns and often contained the town hall of the commune.

DISCUSSION

Understanding the underlying processes that lead to biodiversity loss in cultural landscapes is an important challenge for biodiversity conservation (Norton *et al.*, 2013; Martín-López & Montes, 2015). Drawing on a rich array of empirical data, we identified linkages between social and ecological characteristics of villages, and on this basis developed a typology of social–ecological units. This typology captures how humans influence biodiversity via patterns of land use, which in turn are mediated by topography and socio-economic characteristics (Fig. 1). Our typology thus helps to extract key patterns of how social–ecological dynamics play out locally in terms of biodiversity outcomes (Lüdeke *et al.*, 2004). Understanding such patterns, in turn, may be a useful input to inform conservation strategies that take into account local peculiarities while still considering the wider landscape context (Table 2; Bengtsson *et al.*, 2005; Gabriel *et al.*, 2010).

With respect to biodiversity conservation, different threats and opportunities result from the interplay of biophysical conditions, socio-demographic factors and geographic location (Table 2). First, *biophysical conditions* and especially topography frame the setting of cultural landscapes as a whole. Although they are not a direct driver of biodiversity loss (Norton *et al.*, 2013), biophysical conditions are relevant determinants that structure the spatial arrangement of settlements and choices of land use types. For example, in Switzerland, flat lowlands are dominated by intensive agriculture with relatively low plant species richness, whereas plant species richness peaks at mid-elevations supporting low-

Table 2 Summary of the different characteristics of four village types (see Fig. 3) and summary of key threats and opportunities for biodiversity conservation.

Cluster	Topography	Socio-demographics	Land cover	Species richness	Threats and opportunities for biodiversity conservation
1 (arable type)	Lower altitudes, relatively flat	Large, well connected and often Hungarian, with town hall	Large proportion of arable land, low amount of woody vegetation and forest	Low in birds, high in butterflies and intermediate in plants	Risks to lose butterfly and plant richness through intensification; opportunities to conserve open-area arable specialists and species within farmland
2 (complex type)	Undulating	Tends to have many Romanians and to be remote from larger towns	Intermediate amounts of woody vegetation and arable land	Intermediate to high in birds, plants and butterflies	In some villages tendency to abandonment, in others to intensification; general trend towards forest exploitation; threats and opportunities for all groups
3 (pasture type)	Higher altitudes, relatively hilly	Remote, small, no town hall, tends to have more Hungarians	Large proportions of pasture, low heterogeneity in farmland, low amount of woody vegetation	Low in birds, high in butterflies and high in plants	Intensification of pastures; degradation of pastures due to overgrazing; opportunities for conservation of open-area pasture species
4 (forest type)	Steep slopes	Departures high, many Romanians, Roma and Saxons	High amounts of woody vegetation, low amounts of arable	High in birds, low in butterflies and plants	Tendency of abandonment of arable land and pastures; risk of forest exploitation; opportunities to conserve birds of semi-open farmland and large forests; conservation strategies need to be cognizant of often poor socio-economic conditions

intensity agriculture (Wohlgemuth *et al.*, 2008). Similarly in Transylvania, larger scale (and often more intensive) agriculture is more pronounced in flat valley bottoms, whereas pastures typically occupy the higher slopes, and forests dominate in rugged areas and on hilltops (Hanspach *et al.*, 2014).

Second, we were able to identify linkages of *socio-demographic factors* with land cover characteristics. Emigration of young villagers, for example, has been shown to be driven by the search for income opportunities which small-scale farming no longer provides (Hanspach *et al.*, 2014; Mikulcak *et al.*, 2015). Emigration, in turn, causes the abandonment of traditional, labour-intensive farming methods that sustained high species richness over centuries. Land use intensification, in contrast, is the other economically viable response to the lack of profitability of traditional farming. Intensification has been fostered by national and international policies such as the Common Agricultural Policy in large parts of Western Europe and is widely acknowledged to have contributed to the loss of farmland biodiversity (e.g. Donald *et al.*, 2001). Rather than considering only biophysical conditions, conservation management thus needs to be cognizant of potentially important socio-demographic factors.

Third, *geographic location* was found to relate to social–ecological conditions and therefore land use and biodiversity. Well-connected (i.e. less remote) villages tend to be larger and contain the town hall. Consequently, economic conditions are better in such villages, and land use intensification is more likely, while, in contrast, emigration and land abandonment have been shown to be more likely in small, remote villages (Hanspach *et al.*, 2014). Also, larger communities may be more resilient against shocks than smaller ones (Alessa *et al.*, 2009). The impact of roads on resource use and land use change is probably most evident in frontier landscapes such as the Amazon (Southworth *et al.*, 2011), but has also been found to influence rural development in Transylvania (Mikulcak *et al.*, 2013). It is, however, just one factor among many to influence land use change (Beilin *et al.*, 2014).

Importantly, threats and opportunities for biodiversity conservation in agricultural landscapes such as Southern Transylvania are the result not only of single variables, but also of their complex interactions (Norton *et al.*, 2013). For example, the current high species richness in Southern Transylvania (Akeroyd & Page, 2006) is the result of a long history of low-intensity small-scale farming (interrupted only briefly by a more intensive period during communism) and the maintenance of traditional values and knowledge (Palang *et al.*, 2006; Barthel *et al.*, 2013). Among other reasons, the rural exodus through Saxon emigration and (temporal) migration of Romanians and Hungarians for work to Western Europe has contributed strongly to the loss of this cultural identity and the discontinuation of traditional farming.

Clearly, typologies, such as that developed in this paper, provide only a snapshot and cannot capture the temporal dynamics of systems. Romania has, just as many eastern European countries, gone through major socio-economic changes in the last decades with profound and ongoing effects on land use patterns (Munteanu *et al.*, 2014). Moreover, future research could address in more detail how certain social characteristics influence land use and biodiversity. In addition to the variables we considered, social values, technology or knowledge have been identified as drivers of land use decisions elsewhere (Norton *et al.*, 2013). By focusing on villages as the units of analysis, we aggregated across differences within villages, although such within-village variability could be very important both ecologically (e.g. for species turnover) (e.g. Dorresteijn *et al.*, 2015) and socially (e.g. for diversity of values and livelihood strategies) (e.g. Milcu *et al.*, 2016). Finally, land use patterns result from an interplay between factors from outside and within any given landscape (Hanspach *et al.*, 2014). In the above typology, we assumed that external drivers would have similar effects on the whole region, but we are acutely aware that various external drivers may play out differently in different locations within the study area (Hanspach *et al.*, 2014). Notwithstanding these limitations, incorporating both social and ecological knowledge is a promising strategy to develop more effective strategies for biodiversity conservation (Knight *et al.*, 2006; Ban *et al.*, 2013).

CONCLUSIONS

Ecological patterns in farming landscapes, such as the distribution of local levels of species richness, are driven by human land use, which itself results from the interaction of spatially heterogeneous topographic and socio-demographic conditions. The development of typologies of social–ecological units can help to understand how the interplay of these conditions leads to different biodiversity outcomes in different places. This information, in turn, can be used to help inform locally relevant conservation strategies.

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SUPPORTING INFORMATION

Additional Supporting Information may be found in the online version of this article:

Figure S1 Analysis steps.
Figure S2 Distribution of socio-demographic variables.
Figure S3 Model validation.
Figure S4 Maps of predicted species richness.

BIOSKETCH

Jan Hanspach is a post-doctoral researcher at Leuphana University Lüneburg. He is interested in combining ecological and social perspectives to achieve better outcomes for biodiversity conservation and sustainability in farming landscapes.

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